BNL-48777
(Rev. 5/93)

PILOT STUDY RISK ASSESSMENT FOR SELECTED PROBLEMS AT THE FERNALD ENVIRONMENTAL MANAGEMENT PROJECT (FEMP)

L. D. Hamilton, A. F. Meinhold, S. L. Baxter, S. Holtzman,
S. C. Morris, R. Pardi, M. D. Rowe, and C. Sun

March 1993

Prepared for Office of Environmental
Restoration and Waste Management
Office of Technology Development

BIOMEDICAL AND ENVIRONMENTAL
ASSESSMENT GROUP

ANALYTICAL SCIENCES DIVISION

DEPARTMENT OF APPLIED SCIENCE

BROOKHAVEN NATIONAL LABORATORY
UPTON, LONG ISLAND, NEW YORK 11973

PILOT STUDY RISK ASSESSMENT FOR SELECTED PROBLEMS AT THE FERNALD ENVIRONMENTAL MANAGEMENT PROJECT (FEMP)

L.D. Hamilton, A.F. Meinhold, S.L. Baxter, S. Holtzman,
S.C. Morris, R. Pardi, M.D. Rowe and C. Sun

March 1993

Prepared for Office of Environmental Restoration
and Waste Management
Office of Technology Development

BIOMEDICAL AND ENVIRONMENTAL ASSESSMENT GROUP
ANALYTICAL SCIENCES DIVISION
DEPARTMENT OF APPLIED SCIENCE
BROOKHAVEN NATIONAL LABORATORY
UPTON, NY 11973

Under Contract No. DE-AC02-76CH00016 with the
U.S. Department of Energy

DISCLAIMER

ACKNOWLEDGEMENTS

This study was funded by the United States Department of Energy, Office of Environmental Restoration and Waste Management, Office of Technology Development. The Project Officer for the effort was Mr. Michael Barainca. This pilot study was a collaborative effort between Brookhaven National Laboratory (BNL) and Lawrence Livermore National Laboratory (LLNL). The principal investigator for the BNL effort was Dr. L. D. Hamilton. The principal investigators for the LLNL work were Dr. L. Anspaugh and Dr. D. Layton. We thank the staff of the Westinghouse Company of Ohio and United States Department of Energy personnel associated with the Fernald Environmental Management Project for their cooperation, and Mr. Leo Duffy and Dr. Clyde Frank for their encouragement and support.

ABSTRACT

Two important environmental problems at the USDOE Fernald Environmental Management Project (FEMP) facility in Fernald, Ohio were studied in this human health risk assessment. The problems studied were radon emissions from the K-65 waste silos, and offsite contamination of ground water with uranium.

Waste from the processing of pitchblende ore is stored in the K-65 silos at the FEMP. Radium-226 in the waste decays to radon gas which escapes to the outside atmosphere. The concern is for an increase in lung cancer risk for nearby residents associated with radon exposure. Monitoring data and a gaussian plume transport model were used to develop a source term and predict exposure and risk to fenceline residents, residents within 1 and 5 miles of the silos, and residents of Hamilton and Cincinnati, Ohio. Two release scenarios were studied: the routine release of radon from the silos and an accidental loss of one silo dome integrity. Exposure parameters and risk factors were described as distributions. Risks associated with natural background radon concentrations were also estimated.

Exposure to radon associated with the K-65 silos was estimated to result in individual lifetime risks greater than 1×10^{-4} only for indoor workers (mean: 4.3×10^{-4}) and fenceline residents (median: 1.8×10^{-4}) under the routine release scenario. Population risks associated with the routine and accidental release scenarios were less than 1.0 for all identified receptor populations. Individual and population risks associated with background radon concentrations (median individual risk: 7.3×10^{-3}) were 1 to 4 orders of magnitude larger than the risks associated with radon from the silos.

Historical releases of uranium at the FEMP have resulted in a plume of uranium in ground water south of the facility. The concern is for toxic effects and or an increased risk of cancer to people using ground water. Two scenarios were considered: 1) a continuing source of uranium and 2) no additional source. Model predictions were performed by IT Corporation, using a calibrated, three dimensional ground water transport model. Exposures and risks were estimated 70 years into the future for currently located residential wells, potential wells located along the center of the developing plume, and all possible future well locations in the impacted area. Exposure routes included in the assessment were: water ingestion, intake of homegrown food and intake of homeproduced milk. Intake rates were based on distributions derived from published data.

A threshold distribution for effects in the kidney was developed, based on a pharmacokinetic model, and the probability of a toxic effect defined as the probability of kidney uranium concentration exceeding the threshold. The cancer risk assessment was based on ALI values (annual limits on intake) published by the International Commission of Radiological Protection (ICRP), modified to reflect a distribution of gut uptake factors and allowing dose to be committed only for 70 years.

No toxic effects were predicted for any individual well. An assessment incorporating additional model uncertainties resulted in a predicted probability of toxic effects of 4×10^{-6} for any well located south of the FEMP in the next 70 years. All estimated cancer risks were small. The largest predicted individual lifetime risk was for a specific residential well (1.3×10^{-5}). The predicted individual lifetime cancer fatality risks for wells located anywhere south of the facility were small (always less than 2.2×10^{-5}). Predicted risks for both the stop and continue source scenarios were similar because most exposure is associated with uranium discharged before 1989.

Contents

ABSTRACT ii

ACRONYMS, ABBREVIATIONS AND SYMBOLS xiii

1 INTRODUCTION 1

2 RISK ASSESSMENTS FOR U.S. DEPARTMENT OF ENERGY SITES 2
2.1 Risk Assessment 2
2.2 Approaches to Risk Assessment 3
2.3 Pilot Study Approach to Realistic Risk Assessment for USDOE Facilities 6
2.3.1 Specific Steps in the Analysis 6
2.3.2 Methods and Assumptions 8

3 OVERVIEW OF THE FERNALD ENVIRONMENTAL MANAGEMENT PROJECT 12
3.1 The Fernald Environmental Management Project 12
3.2 Regional Setting and Physical Environment 12
3.2.1 Location 12
3.2.2 Climate 12
3.2.3 Hydrology 13
3.2.4 Hydrogeology 14
3.2.5 Topography 15
3.2.6 Population 16
3.2.7 Land Use 16
3.2.8 Use of Surface and Ground Water 16
3.3 Site Description and Site Operations 16
3.4 Environmental Problems 18
3.4.1 Releases From Site Operations 18
3.4.2 Waste Site Releases 18
3.5 Environmental Monitoring and Environmental Restoration 18
3.5.1 Environmental Monitoring Program 18
3.5.2 Environmental Restoration 19

4 PROBLEMS TO BE STUDIED 20
4.1 Criteria for Problem Selection 20
4.2 Problem 1: Radon Emissions From the K-65 Silos 20
4.2.1 Description of Problem 20
4.2.2 Reasons for Selecting This Problem 20
4.2.3 Transport Pathways and Potential Receptors 21
4.3 Problem 2: Uranium Contamination of Ground Water 21
4.3.1 Description of Problem 21
4.3.2 Reasons for Selecting This Problem 21
4.3.3 Transport Pathways and Potential Receptors 21
4.4 Problems Identified for Future Assessments 21

5 RISK ASSESSMENT FOR RADON EMISSIONS FROM THE K-65 SILOS 22
5.1 Problem Summary and Outline of Steps in the Assessment 22
5.1.1 Steps in the Assessment 23
5.1.2 Problem Summary 23
5.2 The K-65 Silos 23
5.2.1 Physical Description 23
5.2.2 Silo Contents and Emissions 24
5.2.3 Current Status 25
5.3 Radon 26
5.3.1 Physical Properties 26
5.3.2 Indoor vs. Outdoor Concentrations 28
5.3.3 Radon and Radon Progeny in a Plume 29
5.3.4 Exposures and Dose-Conversion Factors 30
5.3.5 Dose-response Relationships 31
5.4 Identification of Scenarios and Receptors 33
5.4.1 Release and Exposure Scenarios 33
5.4.2 Receptors 34
5.4.3 Exposure Periods 37
5.5 Radon and Radon Progeny Concentrations 39
5.5.1 Background Concentrations 39
5.5.2 Prediction of Radon and Radon Progeny From K-65 Silos 40
5.5.2.1 Atmospheric Dispersion 40
5.5.2.2 Radon Source Term 41
5.5.2.3 Radon Production Rate 46
5.5.2.4 Routine Concentrations On and Offsite 46
5.5.2.5 Accidental Exposures 51
5.6 Exposures to Radon and Radon Progeny 52
5.6.1 Calculation of Exposure 53
5.6.1.1 Routine Release Scenario and Natural Background Exposures 53
5.6.1.2 Accidental Release Scenario 54
5.6.2 Background -- Input Parameters and Exposure Estimates 54
5.6.3 Routine Release -- Input Parameters and Exposure Estimates 55
5.6.4 Accidental Release -- Input Parameters and Exposure Estimates 58
5.7 Risk Characterization 59
5.7.1 Calculation of Risk 59
5.7.2 Natural Background -- Risk Estimates 60
5.7.3 Routine Release -- Risk Estimates 62
5.7.4 Accidental Release -- Risk Estimates 65
5.8 Assumptions and Uncertainties 66
5.8.1 Indoor and Outdoor Background Concentrations 66
5.8.2 Prediction of Radon and Radon Progeny Concentrations 67
5.8.3 Exposure Periods 69
5.8.4 Risk Factor for Radon 69
5.9 Summary -- Risk Assessment for Radon Released From the K-65 Silos 70

6 RISK ASSESSMENT FOR URANIUM IN GROUND WATER ("SOUTH PLUME") 72
6.1 Problem Summary and Outline of Steps in the Assessment 72
6.1.1 Steps in the Assessment 72
6.1.2 Problem Summary 73
6.1.3 Current Status 73

6.2 Uranium - Physical Properties and Environmental Fate 74
6.2.1 Physical and Chemical Properties 74
6.2.2 Geochemistry of Uranium 77
6.2.2.1 Occurrence and Distribution of Natural Uranium 77
6.2.2.2 Solid Earth 77
6.2.2.3 Ground Water 78
6.2.2.4 Factors Affecting Uranium Mobility 79
6.2.3 Biological Uptake 82
6.3 Human Health Effects 83
6.3.1 Introduction 83
6.3.2 Toxicity 83
6.3.2.1 Toxic Effects in Humans 83
6.3.2.2 Realistic Risk Analysis for Uranium Toxicity 84
6.3.2.3 Method 84
6.3.2.4 Characterization of the Probability Distribution 85
6.3.3 Carcinogenicity 92
6.3.3.1 Evidence for Uranium Cancer Risk in Humans 92
6.3.3.2 Retention Function 92
6.3.3.3 Effective Dose 93
6.3.3.4 Dose per Unit Intake 94
6.3.3.5 Risk Factor 97
6.4 Identification of Scenarios and Receptors 98
6.4.1 Release Scenarios 98
6.4.2 Receptors 98
6.4.3 Exposure Scenarios 98
6.4.4 Incremental vs. Total Exposure and Risk 99
6.5 Uranium Concentrations in Ground Water 100
6.5.1 Natural Background Concentrations 100
6.5.2 Prediction of Uranium Concentrations in the South Plume 101
6.5.2.1 Introduction 101
6.5.2.2 Ground Water Flow and Transport Model 102
6.5.2.3 Simulations and Model Predictions 105
6.5.2.4 Predicted Concentrations at Receptor Locations 107
6.5.3 Model Uncertainty 112
6.6 Uranium Concentrations in Food and Milk 112
6.6.1 Calculation of Uranium Concentrations in Homegrown Food 112
6.6.2 Calculation of Uranium Concentrations in Homeproduced Milk 113
6.6.3 Values Used in Calculation of Uranium Concentrations in Homeproduced Food and Milk 114
6.6.4 Estimated Uranium Concentrations in Homeproduced Food and Milk at Identified Receptors 118
6.6.5 Model Uncertainty 119
6.7 Uranium Intake 119
6.7.1 Calculation of Intake 120
6.7.2 Parameter Distributions 121
6.7.3 Intake Calculations: Background, Toxicity Assessment and Cancer Risk 125
6.7.3.1 Background 125
6.7.3.2 Toxicity Assessment 125
6.7.3.3 Cancer Risk 125

6.8 Risk Characterization 126
6.8.1 Risk of Toxic Effects 126
6.8.2 Cancer Risk 128
6.9 Assumptions and Uncertainties 131
6.9.1 Toxicity Assessment 131
6.9.2 Cancer Risk Assessment 131
6.10 Summary -- Risk Assessment for Uranium in Ground Water 132

7 SUMMARY AND CONCLUSIONS 133

8 REFERENCES 135

Appendix A: Comparison of Uranium Toxicity Assessment to USEPA Approach 148

Appendix B: Predicted Uranium Concentrations in Water, Food and Milk 154

Figures

2-1 Spectrum of approaches to risk assessment 4

2-2 Steps in the assessment 7

2-3 Probabilistic risk assessment example: estimation of individual lifetime risk of cancer mortality from ingestion of tritium in drinking water 11

3-1 Location of the Fernald Environmental Management Project 13

3-2 Surface water features at the FEMP 14

3-3 Buried valley aquifer underlying the FEMP 15

3-4 Fernald Environmental Management Project 27

5-1 Steps in the risk assessment for radon emissions from the K-65 silos 22

5-2 The radon decay chain 27

5-3 Decay of radon and development of radon progeny with time in the absence of deposition to environmental surfaces (working levels per 100 pCi/L radon-222 in a plume) 27

5-4 Indoor and outdoor equilibrium factors in a radon plume as a function of travel time 30

5-5 Individual receptors located along the western and southern boundaries of the FEMP 35

5-6 1990 Population distribution within 5 miles of the FEMP (data from SAIC, 1987) 35

5-7 Populations near the FEMP 37

5-8 Fraction of rural households living in the same residence for t years or more 39

5-9 Location of site radon monitors 43

5-10 Source term coefficient 43

5-11 Predicted source term from data at each monitor 44

5-12 Measured and predicted radon concentrations 45

5-13 Locations of FEMP residential monitors 47

5-14 Predicted radon concentrations near the source 49

5-15 Estimated radon concentrations from routine emissions of 1150 Ci/yr (4.3×10^4 GBq/yr), indoor and outdoor concentrations in units of pCi/L 50

5-16 Average individual exposures within 5 miles of the K-65 silos 57

5-17 Average population exposures within 5 miles of the K-65 silos 58

5-18 Distribution of individual lifetime risks associated with exposure to background concentrations of radon and radon progeny 61

5-19 Population (collective) risks due to natural background radon (predicted # of cancers x 10^{-2}) within 5 miles of the silos 62

5-20 Routine release scenario -- distribution of individual lifetime risks for most highly exposed fenceline residents 63

5-21 Routine release scenario -- average individual lifetime risks within 5 miles of the silos 64

5-22 Population (collective) risks based on the routine release scenario (predicted # of cancers x 10^{-2}) within 5 miles of the silos 65

6-1 Steps in the risk assessment for uranium in ground water at the FEMP 72

6-2 Current extent of uranium contamination in ground water (as predicted by a ground water model, calibrated against monitoring data, Section 6.5.2). Concentration range is 1 - 450 μg/L, contour interval shown is 50 mg/L 74

6-3 Uranium-238 and uranium-235 decay series 75

6-4 pH-Eh diagrams for uranium and thorium (after Brookins, 1984) 81

6-5 Distribution of threshold of effect of uranium in kidney 91

6-6 Integrated retention function for uranium in humans 93

6-7 Currently located potential receptors for uranium in ground water 99

6-8 Offsite wells 101

6-9 Flow and transport model boundaries 103

6-10 Predicted uranium concentrations (μg/L), source removed, initial conditions and 70 years 106

6-11 Predicted uranium concentrations (μg/L), source continued, initial conditions and 70 years 106

6-12 Representative and centerline receptor wells 108

6-13 Predicted uranium concentrations (μg/L) over time at five representative well locations 109

6-14 Predicted uranium concentrations (μg/L) over time at four centerline wells 109

6-15 Grid cells in model domain representing potentially impacted wells 110

6-16 Model predictions: probability of locating a well in ground water with specified range of uranium concentrations (μg/L).....111

6-17 Locations sampled for uranium in soil and vegetation in 1989.....116

6-18 Mean individual lifetime cancer risk for potential future wells in the impacted area, stop source and continue source scenarios 130

Tables

5-1 Typical uranium concentrates.....24

5-2 Dose-response function for exposure to radon and radon progeny.....32

5-3 Receptors for radon risk assessment - individual and population risk.....34

5-4 Natural background measurements of radon near the FEMP 40

5-5 Natural background radon concentrations near the FEMP 40

5-6 Measured radon concentrations at three residential monitors (pCi/L, (Bq/m^3)) ...47

5-7 Calculated radon concentrations at three residential monitors (WL) 48

5-8 Routine release scenario - models and parameters for predicting concentrations...49

5-9 Routine release scenario -- predicted concentrations at individual receptors.....50

5-10 Accidental release scenario - models and parameters for predicting concentrations and calculating exposure.....52

5-11 Estimated range of accidental exposures to 1 mile 52

5-12 Lifetime individual exposure estimates - background concentrations (WLM) 54

5-13 Lifetime population exposure estimates - background concentrations.....55

5-14 Routine release scenario -- parameters for calculating exposure.....56

5-15 Routine release scenario - lifetime individual exposure estimates.....56

5-16 Routine release scenario -- lifetime population exposure estimates.....57

5-17 Accidental release -- individual exposure estimates 59

5-18 Population exposure estimates -- accidental release.....59

5-19 Lifetime individual risks - background concentrations 60

5-20 Population (collective) risks due to natural background radon 61

5-21 Routine release scenario --lifetime individual risk estimates 63

5-22 Routine release scenario -- lifetime population risks 64

5-23 Accidental release -- individual risk estimates 66

5-24 Accidental release - population risks 66

5-25 Uncertainty analysis for radon source term -- effect on individual lifetime risks for individuals living within 1-mile of the silos 68

5-26 Individual lifetime risks 71

5-27 Population risks 71

6-1 Data used to estimate uranium uptake through the gut 87

6-2 Comparison of absorption of uranium through gut to the bloodstream per unit intake from three classes of study 88

6-3 Blood to kidney fractional transfer rates with half-times 89

6-4 ALIs and calculated effective doses, E_{50}, E_{70} for uranium isotopes 94

6-5 Uranium retention function, percentage of effective dose delivered over time for intake at year 1 and effective dose (Sv) for a chronic unit intake of uranium (1 Bq/day) in each year of a 70-year lifetime. (based on f1 = 0.05) 95

6-6 Concentration of uranium in offsite wells, 1989 101

6-7 Parameters used to calculate the concentration of uranium in homeproduced food and milk 114

6-8 Concentration of uranium in soil and vegetation samples near the FEMP (USDOE 1990c) 117

6-9 Potential wells-- concentrations of uranium in homegrown food and homeproduced milk for ground water concentration ranges 119

6-10 Parameter distributions used in exposure calculations 122

6-11 Estimated values and distributions for parameters used in calculating the amount of uranium in the kidney 126

6-12 Representative residential and centerline wells -- individual lifetime cancer risk distributions 129

6-13 Uncertainty analysis for cancer risk assessment -- dose estimates and transport model uncertainty 132

ACRONYMS, ABBREVIATIONS AND SYMBOLS

Term	Definition
α	fractional transfer blood to kidney
α_i	fractional transfer blood to kidney, term i
a_s	specific activity
A_∞	amount of uranium in kidney at equilibrium
ALI	Allowable Limit on Intake
AR	activity ratio, $^{234}U/^{238}U$
AV	Avogadro number
B_{iv}	concentration factor for uptake of uranium from soil by crops
BEIR	Committee on the Biological Effects of Ionizing Radiation
Bq	becquerel
C_F	concentration of uranium in feed
Ci	Curie
C_{im}	uranium concentration in milk from ingestion in water and feed
C_{imc}	uranium concentration in milk (Bq/l)
C_{imt}	uranium concentration in milk (mg/l)
C_{iv}	concentration of uranium in the edible portion of crop
C_{ivc}	concentration of uranium in food from homegrown sources (Bq/kg)
C_{ivt}	concentration of uranium in food from homegrown sources (mg/kg)
C_S	concentration of uranium in soil
C_W	concentration of uranium in water
C_{wc}	uranium concentration in water (Bq/l)
C_{wt}	uranium concentration in water (μg/l)
CAA	Clean Air Act
CERCLA	Comprehensive Environmental Response, Compensation and Liability Act
CF	correction factor: $0.05/f_1$ distribution
CL	confidence limit
d	day
d_b	bulk density
d_j	deposition rate
dis	disintegrations
DOE	United States Department of Energy
DREF	dose rate effectiveness factor
E_{50}	50 year committed effective dose
E_{70}	70 year committed effective dose
$E_{70}(Y)$	effective dose integrated over a 70-year lifetime, for intake in year y
$E_{70}L$	sum of effective doses over a 70 year lifetime
EE/CA	Engineering Evaluation/Cost Appraisal
EF	equilibrium factor
EPA	United States Environmental Protection Agency
FEMP	Fernald Environmental Management Project
f_1	fractional transfer gut to blood
f_j	fraction of the year crops are irrigated
f_{im},F_{im}	transfer factor to milk
f_{if}	transfer factor to meat
F	amount of fruits and vegetables consumed
FH	fraction of fruits and vegetables that are homegrown
FN	fraction of fruits and vegetables not grown at home (1-FH)
FFCA	Federal Facilities Cooperative Agreement

FMPC	Feed Materials Production Center
G	giga, 10^9
GBq	gigabecquerel
GSD	geometric standard deviation
I	daily intake
IC	indoor radon concentration
ICRP	International Commission on Radiological Protection
IEXP	indoor radon exposure
IF	fraction of year exposed indoors
ILR	individual lifetime fatal cancer risk
IR	average irrigation rate
K_d	distribution coefficient
λ	decay constant
λ_{Ei}	effective removal rate constant for the crop
λ_i	uranium radioactive decay constant
λ_w	removal rate constant for loss by weathering
L	liter
LET	linear energy transfer
LOAEL	lowest adverse effect level
μ	micro, 10^{-6}
m	milli, 10^{-3}
M	homeproduced milk intake
MEPAS	Multimedia Environmental Pollution Assessment System
mgd	million gallons per day
MI_c	uranium intake in milk (Bq/day)
MI_t	uranium intake in milk (μg/day)
n_e	effective porosity
NAS	National Academy of Sciences
NCP	National Contingency Plan
NCRP	National Council on Radiological Protection and Measurements
NIRS	National Inorganics and Radionuclides Survey
NLO	National Lead of Ohio
NOAA	National Oceanic and Atmospheric Administration
NOAEL	no adverse effect level
NPDES	National Pollution Discharge Elimination System
NRC	National Research Council
NTS	Nevada Test Site
OC	outdoor radon concentration
OEXP	outdoor radon exposure
OF	fraction of year exposed outdoors
OI_t	uranium consumed in fruits and vegetables from other sources
OF_t	uranium consumed in fruits and vegetables from other sources
p	pico, 10^{-12}
P	effective surface density for soil
pCi	picocurie
P-WLM	person-working level month
Q_{mW}	consumption rate of contaminated water by an animal
Q_{mF}	consumption rate of contaminated feed by an animal
Q_{mS}	consumption rate of soil by livestock
r	fraction of deposited activity retained on crops
R	retardation coefficient
R(t)	whole body retention function

$Rk(t)$	retention function in kidney, t days after ingestion
$R_{bone}(t)$	retention function in bone at t days
$R_{kidney}(t)$	retention function in kidney at t days
$R_{other}(t)$	retention function in other tissues at t days
RCRA	Resource Conservation and Recovery Act
RI/FS	Remedial Investigation/Feasibility Study
Rn	radon
SRS	Savannah River Site
Sv	sievert
SWIFT	Sandia Waste Isolation Flow and Transport Model
t_b	period of time soil is exposed to contaminated water
t_e	period of crop exposure during growing season
t_h	holdup time - interval between harvest and consumption
T_i	biological half-life in kidney, term i
TEXP	total exposure to radon
TI_t	total uranium intake rate (μg/day)
TI_c	total uranium intake rate (Bq/day)
$TI_c(Y)$	daily intake in year y (Bq/day)
U	uranium
UNSCEAR	United Nations Scientific Committee on the Effects of Atomic Radiation
USDA	United States Department of Agriculture
USDOE	United States Department of Energy
USEPA	United States Environmental Protection Agency
USGS	United States Geological Survey
USNRC	United States Nuclear Regulatory Commission
WMCO	Westinghouse Materials Company of Ohio
VI_c	uranium intake in homegrown food (Bq/day)
VI_t	uranium intake in homegrown food (μg/day)
W	water intake
WI_c	uranium intake (Bq/day)
WI_t	uranium intake (μg/day)
WL	Working Level
WLM	Working Level Month
Y_v	agricultural productivity
YE	years exposed

1 INTRODUCTION

Widespread environmental contamination has been documented at United States Department of Energy (USDOE) facilities. Human health risk assessments are increasingly being used to support decisions concerning remediation at these sites.

This study assessed health risks for potential problems at three USDOE facilities: (1) the Savannah River Site (SRS), near Aiken, South Carolina; (2) the Fernald Environmental Management Project (FEMP) in Fernald, Ohio; and (3) the Nevada Test Site (NTS), near Las Vegas, Nevada. The pilot study was a collaborative effort between the Biomedical and Environmental Assessment Group (BEAG) of Brookhaven National Laboratory (BNL) and the Health and Ecological Assessment Division (HEAD) of Lawrence Livermore National Laboratory (LLNL). BNL had primary responsibility for the risk assessments of the SRS and the FEMP, while LLNL took the lead in the assessment of the NTS. This volume of the report, titled *Pilot Study Risk Assessment for Selected Problems at the Fernald Environmental Management Project*, presents risk assessments of an impact on human health for two potentially important problems identified at the FEMP.

Current methods for assessing risk at USDOE facilities are generally excessively conservative or simplistic. Generic models, default parameter values and conservative assumptions are often used, and unrealistic exposure and land use scenarios are embedded in the analyses. These approaches are appropriate only as first level screening analyses to identify contaminants or pathways that are not important in terms of risk to human health.

Site and problem specific decisions about cleanup at USDOE facilities should be based on sophisticated state-of-the-science approaches to human health risk assessment. A more detailed, site-specific approach, based on realistic exposure scenarios and up-to-date dose-response relationships would provide better information to USDOE, to regulatory agencies and to the public. Decisions concerning the need for remediation, the choice between remediation options, definition of cleanup goals, and funding and research priorities could then be assured of protecting the public health in a scientifically based way.

This pilot project has three principal objectives:

(1) to develop scholarly, realistic, and quantitative health-risk assessments, based on state-of-the-art scientific knowledge and methods, for several of the more important environmental and hazardous-waste problems at three USDOE facilities;

(2) to provide and demonstrate methods for realistic risk analyses that can be adapted for application at other sites in the USDOE complex;

(3) to improve significantly the scientific and technical foundation upon which remediation actions at USDOE sites are based, so that such risk-management decisions are scientifically defensible and fiscally sound.

We define a realistic and objective risk assessment to be one that:

(1) avoids unrealistic and conservative exposure scenarios, and focuses on the development of reasonable and sensible scenarios;

(2) replaces generic or inappropriate default assumptions with site-specific data;

(3) explicitly characterizes uncertainties in parameters and does not depend on conservative assumptions;

(4) uses site-specific transport and exposure models and depends on monitoring data for model calibration or input parameters;

(5) uses the latest scientific information in describing dose-response relationships;

(6) acknowledges that situations with very little data available to describe source terms or exposure routes cannot be assessed in a realistic way without additional data collection. These situations can be addressed credibly and practically in screening level assessments.

Section 2 of this document summarizes the approaches to human health risk assessment commonly used at USDOE facilities, and outlines the method of realistic and objective risk assessment developed and demonstrated in the pilot study. Section 3 presents an overview of the Fernald Environmental Management Project and develops much of the background data and information needed in the risk assessment. The bases for the choice of problems and contaminants analyzed in the pilot study are discussed in Section 4. The remainder of the document gives the risk assessments for the two problems studied at the FEMP: (1) radon released from the K-65 silos and (2) uranium in offsite ground water.

2 RISK ASSESSMENTS FOR U.S. DEPARTMENT OF ENERGY SITES

2.1 Risk Assessment

Risk can be defined in different ways. Cohrssen and Covello (1989) technically define risk as the possibility of suffering harm from a risk agent (i.e., chemical substance, organism, radioactive material, or other potential hazard). Consequently, the analysis of risk must describe the discharge of the risk agent, its transport and fate in an environmental media (i.e., air, soil, food, water) and any associated human exposure. Human-health risks are then calculated based on data and models that relate exposures to risk (Till and Meyer, 1983). The scope of such a risk analysis can range from a qualitative discussion to a careful, realistic and quantitative assessment of the likelihood of adverse human health effects (Cohrssen and Covello, 1989).

Several measures can be used to describe the probability that harm will result from exposure to a risk agent (adapted from Cohrssen and Covello, 1989):

o Individual lifetime risk: the estimated increase in probability that an individual will experience a specific adverse health effect over a lifetime as a result of exposure to a specified concentration of a risk agent; and

o Population risk: the number of deaths or adversely affected individuals in the exposed population.

The U.S. Environmental Protection Agency (USEPA) currently considers excess individual lifetime cancer risks (assumed here to mean fatal cancer risks) within the range of 1×10^{-4} (one-in-ten-thousand) to 1×10^{-6} (one-in-a-million) to be

acceptable (Federal Register, 1991). A 10^{-6} excess individual lifetime cancer-mortality risk for the entire United States population (240 million in 1987, DHHS, 1990) would result in 240 premature deaths from cancer. Based on a lifetime expectancy of 70 years, this represents 3.4 deaths in addition to the 2.1 million annual deaths from all causes, including the 477,000 cancer deaths that occurred in the United States in 1987 (DHHS, 1990).

2.2 Approaches to Risk Assessment

Risk assessments cover a spectrum of complexity and accuracy, depending on the aims of the assessment and the level of certainty required of the results. Figure 2-1 outlines this spectrum of approaches to risk assessment that can be used to address environmental problems at USDOE facilities. The accuracy of the results of a risk assessment depends on the transport and exposure models used in the analysis, the variables used as input to the models, and the realism of the assumptions made concerning exposure scenarios, dose-response and risk factors, and receptors. Categorization and screening assessments are the approaches most frequently used at USDOE facilities.

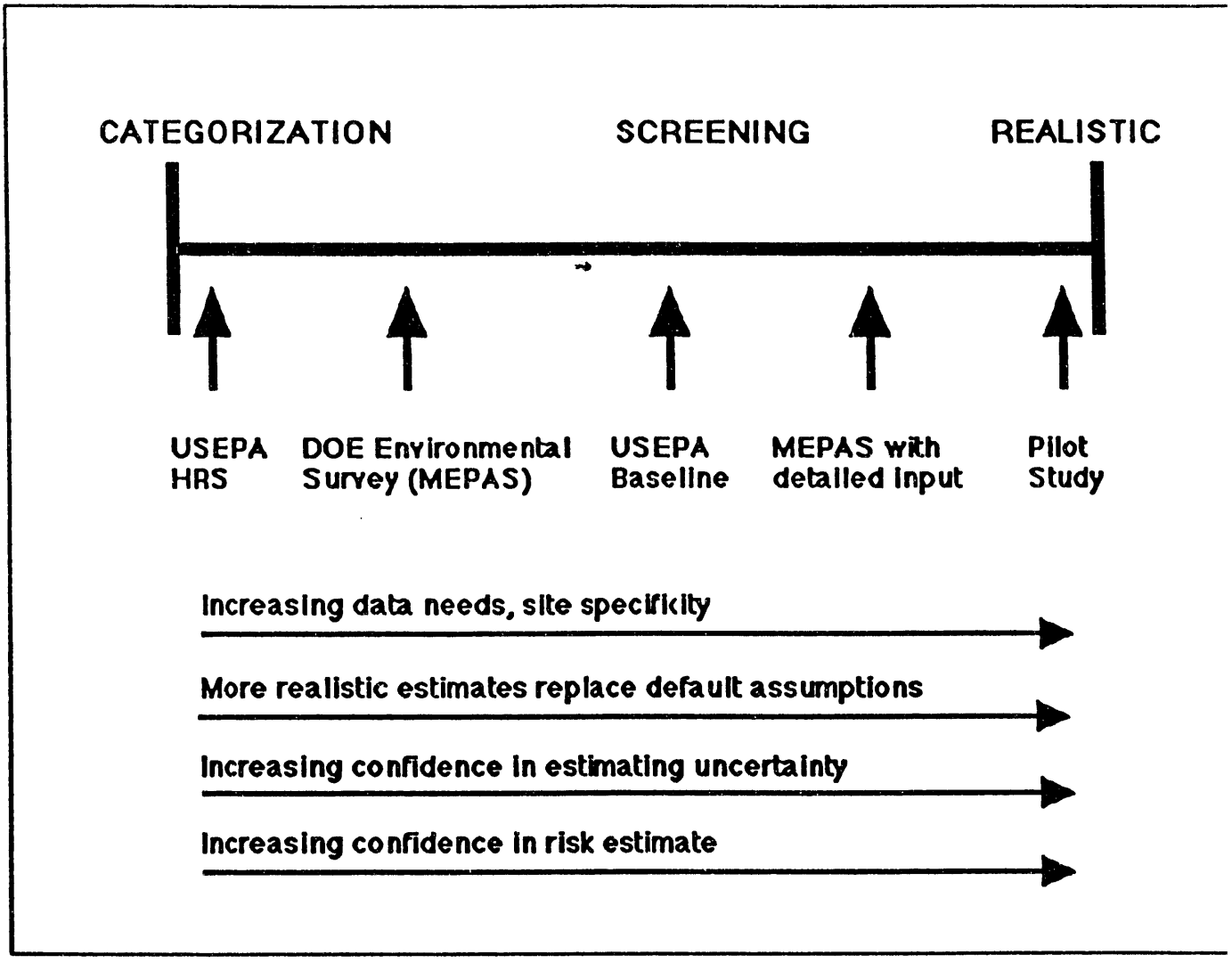
Categorization Assessment

The least complex and accurate approach is aimed simply at identifying sites or problems of potential concern. In this report this approach is called a categorization assessment. The most widely used categorization assessment is the USEPA Hazard Ranking System (HRS: NCP, 1990) which generates numerical scores, based on general site information and a simple ranking model. Sites scoring above a certain level are included on the National Priorities List (NPL) under the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA). This categorization system gives no information on the risks associated with environmental contamination at a site, and few site-specific data are required to generate an HRS score.

Screening Assessment

Screening assessments are the next level of complexity and accuracy in risk assessment. Assessments of this type can range from simple conservative assessments of risk, assuming worst-case conditions, to more sophisticated analyses, based on multiple pathway models. The range of accuracy of the risk estimates produced by these approaches is driven by two major factors: the sophistication of the transport models and other algorithms used; and the accuracy of the input data and other assumptions contained in the analysis. These assessments give approximate estimates of individual and population risks but often rely on simplistic transport models. Conservative and unrealistic default assumptions are usually used to deal with uncertainty. The results of screening assessments can be used as a first step in the identification of environmental problems requiring further assessment.

Figure 2-1 Spectrum of approaches to risk assessment.



Two major kinds of screening risk assessments are commonly performed at USDOE facilities. These are assessments using a Multimedia Environmental Pollution Assessment System (MEPAS) model (Droppo et al., 1990; USDOE, 1988), and baseline risk assessments performed as part of Remedial Investigation/Feasibility Studies (RI/FS) required for Superfund sites (USEPA, 1989).

The MEPAS model was designed to use the results of an environmental survey performed by USDOE for its facilities to rank environmental problems in terms of risk. The model is a multiple-pathway model that follows a contaminant from the source through various environmental media to man. The MEPAS model was designed specifically to produce relative rankings among environmental problems -- the Hazard Potential Index (HPI) produced by the model does not represent absolute estimates of health risk. There are a number of limitations to the model that make it inappropriate for use in a more detailed and accurate approach to risk assessment. These limitations include the simple groundwater model included in the package, treatment of both cancer and toxicity risk in the same way (assuming no threshold), and difficulty in describing the time associated with exposure (Morris, 1990; Morris and Meinhold, 1988). The MEPAS model also uses toxicity values recommended by the USEPA that contain "safety factors" ranging from 10 to 1000, and cancer risk factors that are based on a number of conservative assumptions.

The USDOE environmental survey implemented the MEPAS model by using data and assumptions derived from initial investigations of potential problems at USDOE facilities. The detailed data and analyses required for an accurate estimation of risk were not available across all USDOE sites. Therefore, results of the survey were to be used simply to rank problems in terms of risk, and to help identify areas where further investigation and characterization were needed. For situations in which better data and more accurate assumptions are available, the MEPAS model can generate better estimates than those resulting from the USDOE environmental survey. However, the model is limited by design to producing screening assessments only.

Baseline risk assessments are performed in accordance with USEPA guidelines, and the specifics of the analyses are usually dictated by the USEPA regional office responsible for the facility. Baseline risk assessments represent screening-level assessments because the conservative assumptions embedded in the analyses and the common use of simplistic transport models produce approximate estimates of risk. A baseline risk assessment is conducted to estimate the risks associated with an identified operable unit prior to any remediation of a USDOE site that is listed on the NPL. The assumptions used in the assessments are designed to protect public health and are generally conservative, and tend to overestimate risks. The USEPA has identified a number of environmental transport models that it considers appropriate for use in these assessments. They range from simple 1-dimensional models to more complex computer codes.

The conservative assumptions contained in these baseline assessments include the previously described toxicity values and cancer risk factors, and the use of worst-case conditions for land use, predicted environmental concentrations and exposure rates. The identified receptor for a baseline risk assessment is an individual receiving the "reasonable maximum exposure" [sic] (USEPA, 1989), whether or not she or he exists.

Realistic Assessment

The highest level of accuracy and complexity in a risk assessment process is a realistic assessment. Realistic assessments are usually performed for environmental problems identified by a screening assessment as presenting potentially significant risks. Conservative assumptions and safety factors are replaced in a realistic assessment by best estimates and reasonable exposure scenarios. Environmental concentrations, exposures and risks are presented in the form of distributions, where appropriate. More sophisticated transport models may be used, and predicted concentrations are compared to environmental monitoring data. Uncertainties and conservative assumptions are explicitly described. When data are not available for a realistic assessment, a screening analysis is performed instead.

The screening assessments commonly used at USDOE facilities are appropriate for use as initial tools, or to satisfy USEPA RI/FS requirements. Realistic assessments are the only way to produce reasonable, scientifically-defensible estimates of risk. Realistic assessments: provide better risk estimates to the USEPA, USDOE and to the public; form a better basis for remedial action decisions; and support cost-effective clean-up planning.

2.3 Pilot Study Approach to Realistic Risk Assessment for USDOE Facilities

The risk assessments in this document and in the companion documents for the SRS and NTS are based on a realistic and objective approach to risk assessment. The results of the assessments of the health risks associated with important environmental problems at the three USDOE facilities are well-documented, and scientifically-defensible. Where realistic assessments were not appropriate because of a lack of site- and problem-specific information, detailed credible screening assessments were performed to estimate the magnitude of potential health risks, and to determine what additional data collection is needed. The next section outlines the methods used for developing realistic risk estimates for problems at USDOE sites, and subsequent sections provide additional descriptions of the constituents of a "realistic" risk assessment.

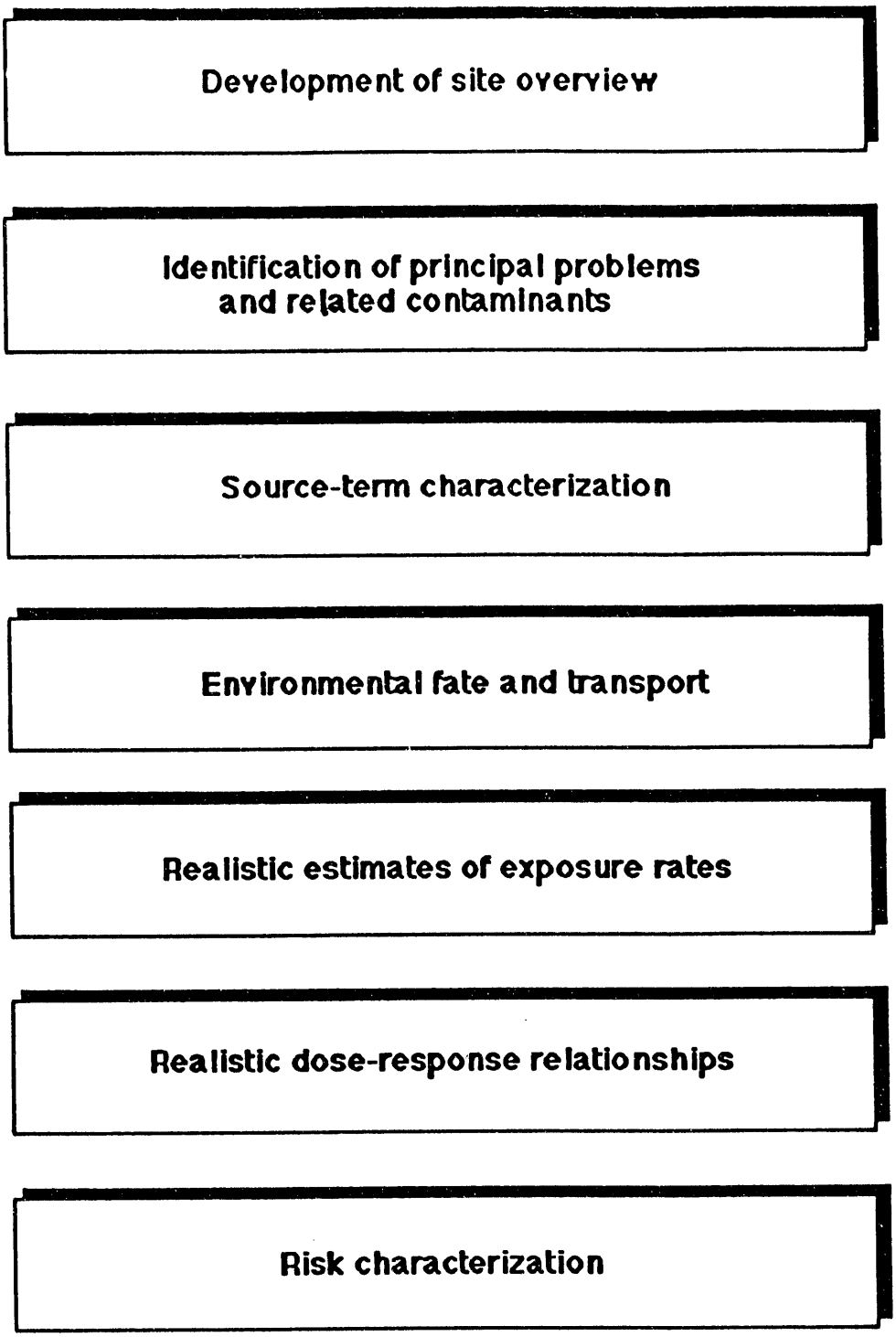
2.3.1 Specific Steps in the Analysis

Figure 2-2 outlines the procedure developed in this pilot study for assessing human health risk at USDOE sites. The steps in this procedure are described in more detail below.

1. *Development of Site Overview.* The first step is to develop a site description that includes relevant background information on the physical setting of the site, nearby populations and land uses, local meteorology, the direction of groundwater flow, and the uses of ground and surface water.

2. *Problem and Contaminant Identification.* The important potential problems and related contaminants in terms of potential human exposures and risks are identified.

Figure 2-2. Steps in the assessment.



3. *Source Term Characterization/ Environmental Fate and Transport.* The major chemical fate and transport pathways are described for the contaminants of concern. The temporal and/or spatial variations of a contaminant's concentration in environmental media are predicted. Monitoring data and other available information are used to estimate the source term. Environmental transport models, adapted to the specific site, are used to predict contaminant concentrations.

4. *Realistic Estimates of Exposure Rates.* Predicted concentrations of a particular contaminant in environmental media are used to estimate potential human exposure to the substance of concern. These estimates are based on reasonable estimates of exposure parameters such as water intake, food intake, and residence time. Site-specific data are used when available. This step also includes the development of reasonable land-use scenarios.

5. *Realistic Dose-response Relationships.* Dose-response relationships, which relate toxic endpoints to exposures or doses are constructed or are identified in the literature. The safety factor approach used by the USEPA is replaced in the estimated relationship between dose and effect for toxicants by an explicit consideration of uncertainty. Cancer risk factors are described as distributions about the best estimate available.

6. *Risk Characterization.* Finally, the potential individual and or population health risks and/or hazards for a specific environmental contaminant are quantified. Where possible, these risks are described as probability distributions.

2.3.2 Methods and Assumptions

Identification of Receptors

Receptors are either people who are currently exposed to the pollutant of interest, or people who have a potential for such exposure. Currently exposed people include onsite populations (*i.e.*, workers), and local or regional populations that are exposed because of environmental transport processes (*e.g.*, wind, surface and ground water movement). Potentially exposed people include people that reside in the path of a plume that is expected to reach them in the future, people that gain entry to the site as a result of a loss of institutional control, and additional people who will be exposed as a result of population growth.

Development of Exposure and Land Use Scenarios

It is assumed that land uses in the future conform to current activities, or to current activities on nearby land, unless there is compelling evidence to the contrary. Either ground water or nearby surface water is assumed to be the source of drinking water, unless there is evidence against such usage.

The assumption of the maintenance or loss of institutional control at a USDOE facility can be critical to the results of a risk assessment when the bulk of the identified contamination is onsite in soil or water. A loss of institutional control may require the use an intruder scenario, and a scenario assuming the possibility of farming or a family living onsite. For problems for which institutional control affects the risk assessment, two analyses are done. The first assumes a maintenance of institutional control in perpetuity. The second assumes loss of institutional control after 100 years.

Environmental Transport

Contaminant concentrations in air, ground water, surface water and food are predicted by using environmental transport models appropriate to each site, each problem and the available data. Where complex site-specific transport models were developed by the USDOE or site contractors, these models are considered for use in the assessments. When a large amount of historical data is available, empirical models are used in place of physics-based models to relate site contaminant discharges to environmental concentrations at receptors.

When sites and source terms are poorly characterized, more elementary models are used. Such models provide appropriate estimates of future environmental concentrations of the principle contaminants of concern for use in a screening analysis.

Exposure Routes and Parameters

Each route of exposure is analyzed for its contribution to the total exposure of a person to a contaminant. The analysis relates the concentration of contaminant in a medium (*e.g.*, pCi/L_{H_2O}, mg/kg_{soil}) to the total amount of the medium to which an individual is exposed. The data used in the analyses include distributions of personal exposure factors (*e.g.*, rates of breathing, ingestion of fish and other activities). These distributions reflect population characteristics in the United Sates, either obtained from published sources or developed from available data. Specific pathways of exposure are not analyzed further when documented evidence shows that they are not appropriate to the specific problem.

Dose-response Relationships and Risk Characterization

Dose-response functions per unit of contaminant exposure are obtained from the literature, or are developed from the available data. For each contaminant, a dose response function is applied to the total exposure to determine the total dose.

Hazards are characterized qualitatively and quantitatively for carcinogenicity and/or toxicity. Risk functions (factors) are also obtained from the literature, or are developed from the available data. These risk factors, expressed as risk per unit dose, or risk per unit intake, are applied to the total doses (or intakes) of contaminants to estimate the risks. In most cases the risks are estimated for cancer mortality, including individual lifetime risks and population risks. These risks are expressed as central values (average and/or median) and uncertainties (standard deviation and cumulative upper 95% probability value). Graphic examples of the ranges of risks are also reported.

For toxic effects other than cancer, the risks are expressed as the probability of an individual incurring the effects, including both a central value and uncertainty in the expression.

Probabilistic Assessment and Uncertainty

A realistic and probabilistic assessment uses the range of possible values for parameters and variables, rather than relying on single values. When appropriate, the risk assessments use distributions of values, rather than single estimates, to produce probabilistic analyses of risk (as demonstrated simplistically in Figure 2-3).

For example, the rate of drinking water intake by individual receptors is represented by a distribution of values, instead of the average (or maximum) rates commonly used in USEPA and USDOE risk calculations. This approach to risk assessment allows an estimate of the uncertainty associated with an "average" or "worst-case" risk estimate. As described above, the result of such an assessment for cancer mortality risk is a distribution that can be presented graphically (Figure 2-3), in addition to presenting the mean and 95% Upper Confidence Limit (*ie.*, cumulative upper 95% value).

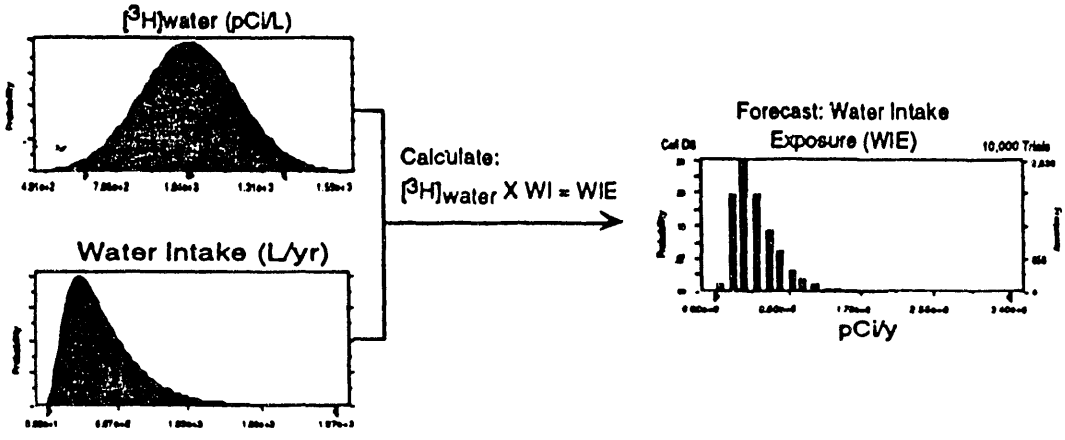
Many of the probabilistic analyses in the pilot study were produced with a Monte Carlo analysis, using Crystal Ball@ (Decisioneering, Boulder, CO) software. In a Monte Carlo analysis, a sample from the distribution of a variable is placed into a simulation run (iteration) to interact with samples from other variables. The frequency of sampling within a distribution of a variable depends upon the relative frequency of a value in that distribution (Paustenbach *et al.*, 1990).

Inadequate Data and Screening Assessment

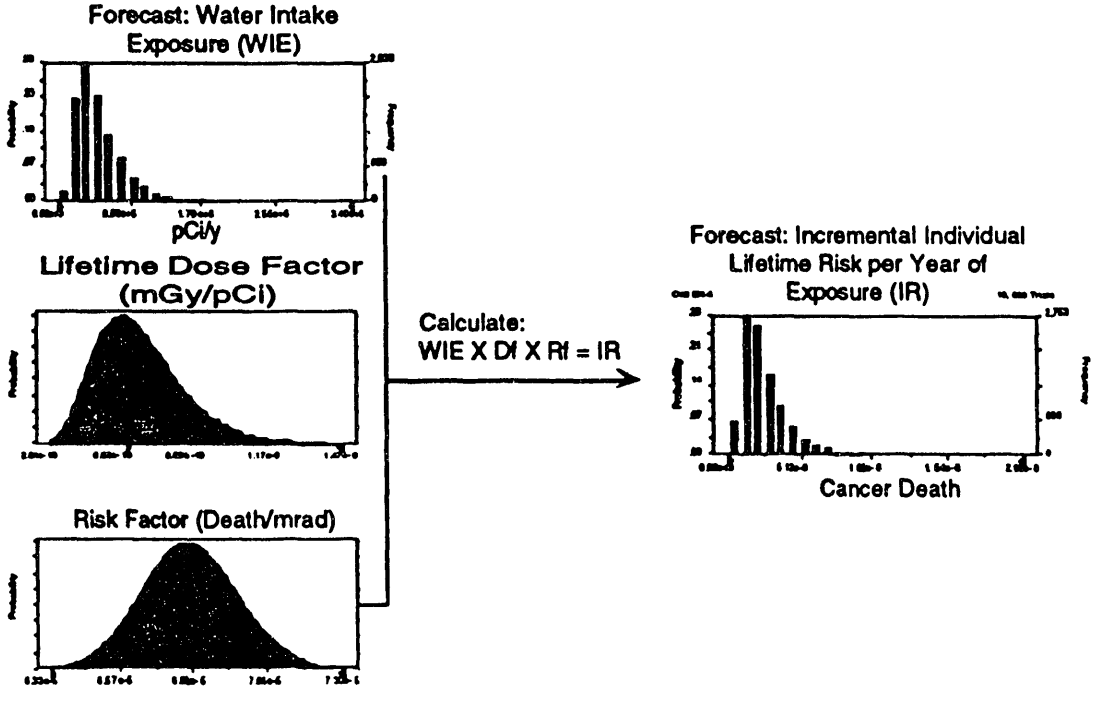
Part of the risk assessment process developed in this pilot study is the exercise of professional judgment concerning the amount (and quality) of site characterization and source term data needed to justify the application of the sophisticated models and probabilistic methods required for a "realistic" assessment. When source term or site characterization data are inadequate, a screening level assessment is the only feasible approach to a problem. The approach used in this pilot study was: to perform such screening assessments when the available data were limited (eliminating unnecessary conservatism where possible); to document explicitly all assumptions used in the analysis; and to draw conclusions on the need for additional data collection.

Figure 2-3. Probabilistic risk assessment example: estimation of individual lifetime risk of cancer mortality from ingestion of tritium in drinking water.

Step 1. a). Develop an assumption about the distribution of tritium concentations in drinking water ($[^3H]_{water}$).
b). Develop an assumption about the distribution of drinking water intake (WI).
c). Calculate exposure to tritium in drinking water (WIE).



Step 2. a). Develop an assumption about the dose factor for tritium ingestion (Df).
b). Develop an assumption about the risk factor for tritium ingestion (Rf).
c). Calculate incremental individual lifetime risk for cancer mortality (IR).



3 OVERVIEW OF THE FERNALD ENVIRONMENTAL MANAGEMENT PROJECT

3.1 The Fernald Environmental Management Project

The Fernald Environmental Management Project (FEMP) is a contractor-operated federal facility that was used for the production of high purity uranium metal for the United States Department of Energy (USDOE). From 1951 to 1986 the FEMP was operated by National Lead of Ohio, Inc. (NLO), and from 1986 to 1992 by Westinghouse Materials Company of Ohio (WMCO). The facility is currently operated by Fluor Daniels Inc. Production ceased in 1989, and resources are now focused on environmental restoration. Prior to 1992, the facility was known as the Feed Materials Production Center (FMPC).

Plant operations resulted in airborne contaminant releases and discharges to surface waters and in the generation of radioactive and nonradioactive wastes. Wastes were stored in pits, silos and drums. Operational and waste site releases resulted in the contamination of air, soil, ground water and surface water on and near the facility. The potential health and environmental effects associated with these releases are of concern to the site contractor, the Department of Energy, regulatory agencies and the public.

A Remedial Investigation and Feasibility Study (RI/FS) is in progress pursuant to CERCLA, as required by a Federal Facilities Consent Agreement signed by the United States Department of Energy and the United States Environmental Protection Agency. Much of the background information used in this report is based on documents prepared as part of the RI/FS process (e.g. USDOE, 1987), and on FEMP Annual Environmental Monitoring Reports (e.g. USDOE, 1990c).

3.2 Regional Setting and Physical Environment

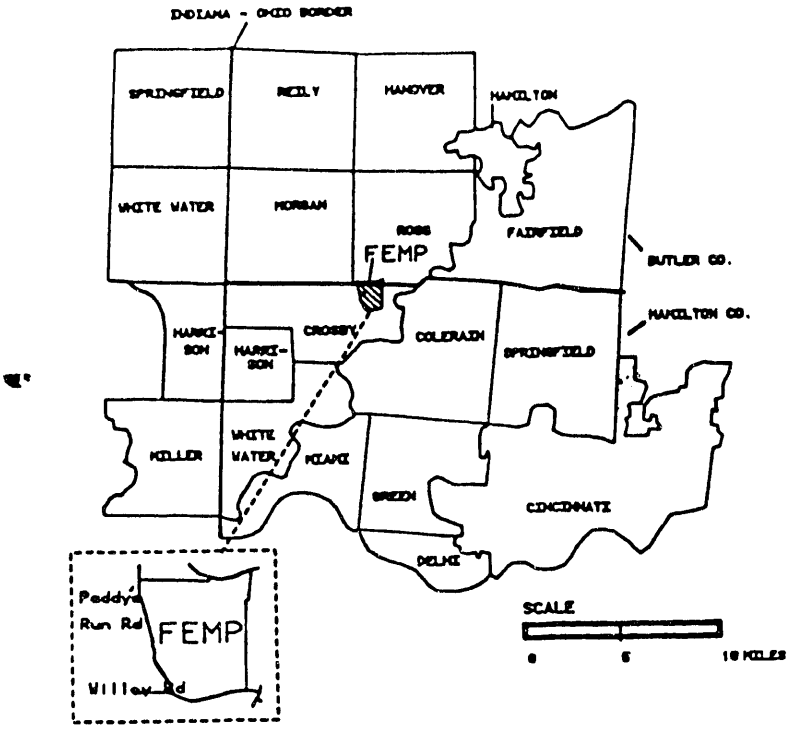
3.2.1 Location

The Fernald Environmental Management Project (FEMP) is located in rural southwestern Ohio, approximately 18 miles northwest of Cincinnati. Hamilton, Ohio is located approximately 10 miles to the northeast (Figure 3-1), and Ross, Ohio is located approximately two miles northeast of the facility. The FEMP comprises 1050 acres, and the production area covers approximately 136 acres near the center of the plant. The facility is located in Hamilton and Butler counties and is bounded by Ohio Route 126 to the north, a dairy farm to the east, Willey Road to the south and Paddy's Run Road to the west.

3.2.2 Climate

The regional climate is continental, with temperatures ranging from an average of 29 ^{o}F in January to 75.5 ^{o}F in July (NOAA, 1984). The average annual precipitation for the period 1975 through 1984 was 37.75 inches and ranged from 29.22 to 40.64 inches per year (NOAA, 1984).

Figure 3-1. Location of the Fernald Environmental Management Project.



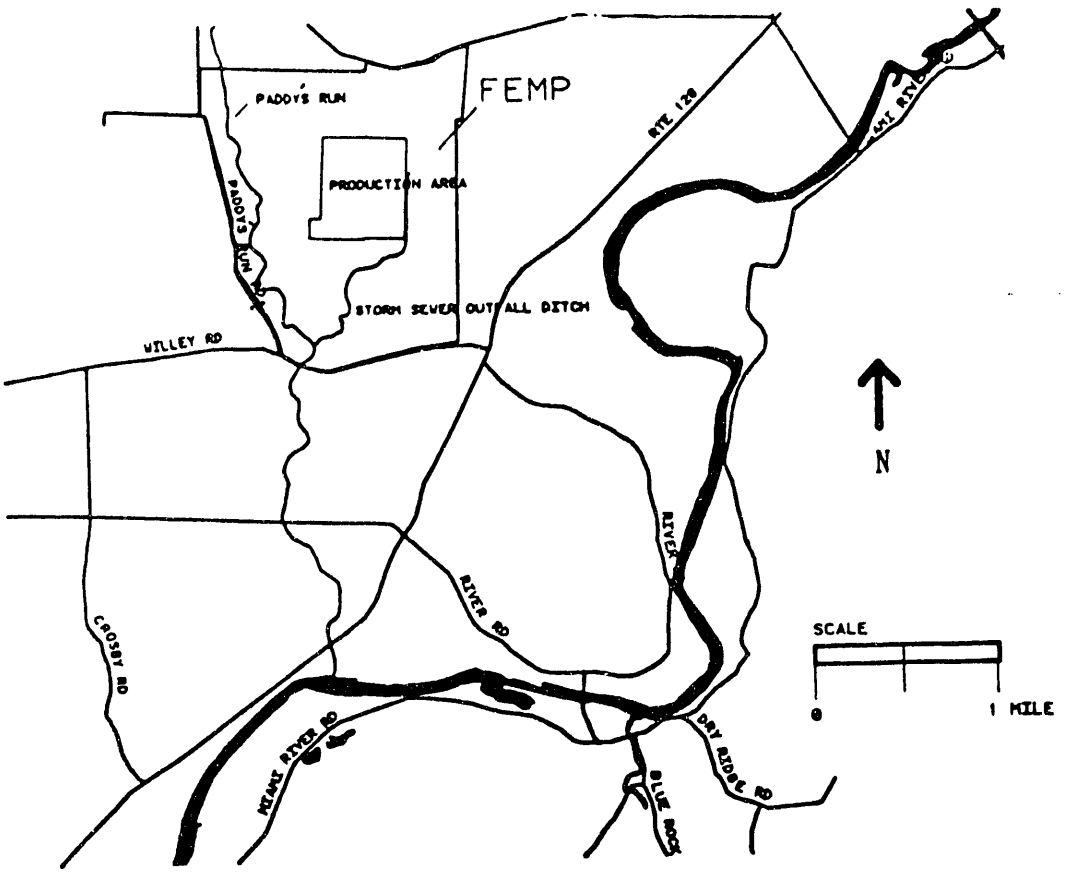
The prevailing winds are from the south-southwest. Data are available for both the Dayton and Cincinnati Airports, and the two data sources are similar in terms of wind direction and wind speed. The average monthly wind speed for the Greater Cincinnati Airport ranged from 6.7 mph in August to 11.1 mph in March (NOAA, 1984). Data for several years are also available from a site meteorological tower.

3.2.3 Hydrolcgy

The main surface water feature in the vicinity of the FEMP is the Great Miami River located about 0.6 mile southeast of the facility (Figure 3-2). The Great Miami River flows into the Ohio River about 18 miles south of the FEMP. Most drainage is to Paddy's Run and then to the Miami River. Paddy's Run flows south along the west side of the waste storage facility (Figure 3-2). This stream has its highest flows between January and May (Dames & Moore, 1985). Additional drainage from the site is to the storm sewer outfall ditch which flows south of the production area and into Paddy's Run in the southwest corner of the facility (Figure 3-2) (Dames & Moore, 1985).

Near the confluence of Paddy's Run and the Storm Sewer Outfall Ditch the relatively impermeable glacial till grades into permeable sand and gravel; surface water in these two streams percolates to ground water (Dames & Moore, 1985).

Figure 3-2. Surface water features at the FEMP.



3.2.4 Hydrogeology

At the FEMP, approximately 49 feet of clay rich till overlie a sand and gravel aquifer. North of the site, till deposits directly overlie bedrock. South of the site, Paddy's Run has eroded some of the till, exposing the sand and gravel aquifer.

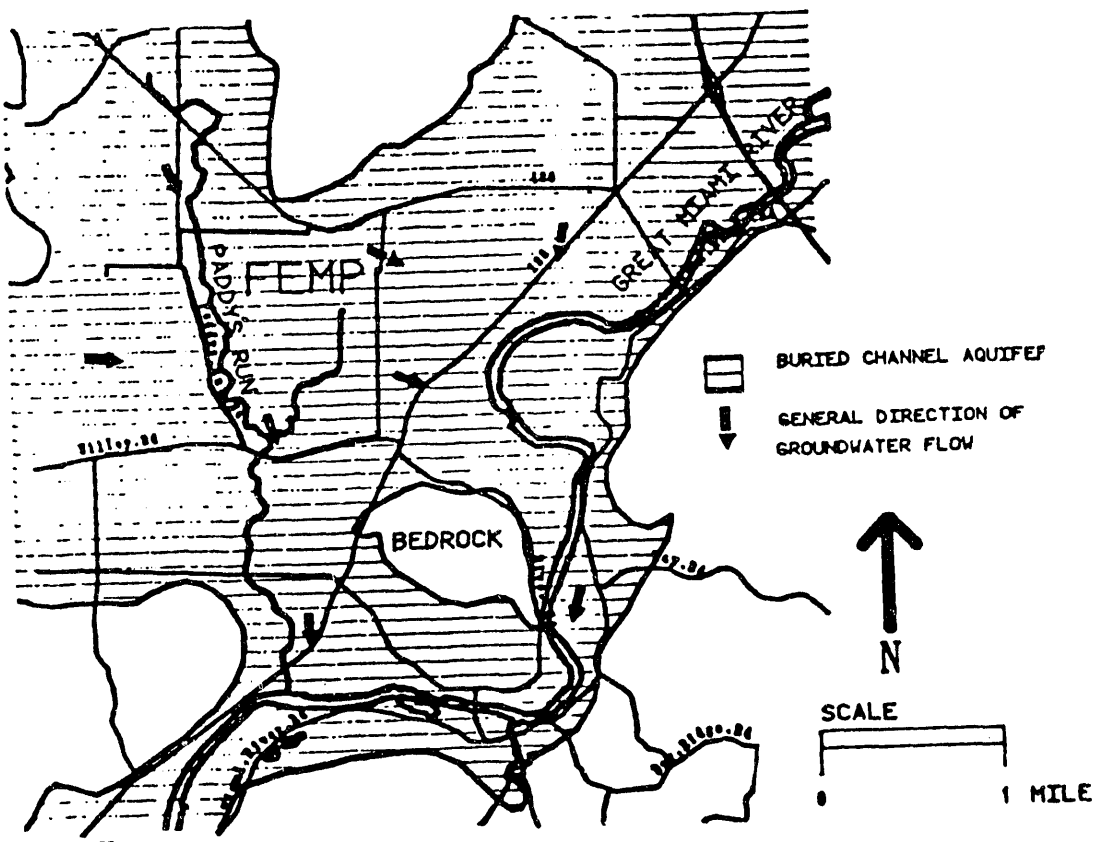
The sand and gravel deposits in the area of the FEMP are part of a large aquifer system. These deposits are about 3 miles wide and 151 feet deep, and fill the remains of an ancient river valley cut into bedrock (Figure 3-3). The Great Miami River flows through the sand and gravel deposits.

Bedrock below the FEMP lies at about 200 feet in the southern portion of the site, and at about 60 feet in the northern portion. The sand and gravel deposits underlying the FEMP site are approximately 150 to 200 feet thick. In some areas of the site a 10 to 20 foot layer of silty clay (referred to as blue clay) occurs at approximately 100 to 125 feet below grade. The blue clay layer does not act as an

aquitard, and no significant head differences exist between wells completed above and below this layer (Dames and Moore, 1985).

Water in the aquifer beneath the FEMP occurs 60-90 feet below the land surface. The upper 20 to 30 feet of the sand and gravel aquifer are not saturated. In some areas of the site, a saturated zone occurs in the silty clay till that overlies the sand and gravel aquifer at 4-9 feet below the land surface.

Figure 3-3. Buried valley aquifer underlying the FEMP.



Ground water movement rates in the sand and gravel aquifer are about 90 to 200 feet per year. Ground water in the sand and gravel aquifer moves east under the waste pit and production areas, and to the south on the southern edge of the facility. Figure 3-3 shows the direction of ground water flow in the sand and gravel aquifer (USDOE, 1990c).

Movement rates in the till near the waste pit area and in the northwest corner of the site are about 14 to 22 feet per year. The flow directions in the till are not uniform, and is affected by local topography and the composition of the till (USDOE, 1990c).

3.2.5 Topography

The FEMP is located on a relatively level plain, about 580 feet above mean sea level. The land rises to 698 feet at the northern boundary and slopes downward to about 551 feet on the western boundary.

3.2.6 Population

The population within 50 miles of the FEMP is approximately 2,577,000 (USDOE, 1987). The population within 5 miles of the site is approximately 14,300 (SAIC, 1987). There are a number of small towns within 5 miles of the site, with populations ranging from 30 to 3,000 (USDOE, 1987). Cincinnati (population 442,000) is located approximately 18 miles to the southeast.

3.2.7 Land Use

Areas immediately surrounding the facility are primarily rural, with some light industry and a few residences. Parts of Hamilton and Butler counties are urbanized, and land uses are primarily residential, commercial and light industrial. The area north of the FEMP and south of State Route 126 is zoned for agricultural use. Land immediately south of the facility is zoned industrial, and land to the east is zoned agricultural (USDOE, 1987).

The principal land use near the FEMP is agriculture. The major agriculture in the area is dairy and beef cattle, and corn and soybean production. There are three dairy operations within three miles of the FEMP, and land on the FEMP site is leased for dairy and beef cattle grazing.

A few heavy industries, including Miami Valley Ready-Mix, Delta Steel, Albright Wilson and Rutgers-Neese are located just south of the FEMP. There are a few residences near the facility, and a larger number of residences in the unincorporated villages of Fernald, New Baltimore, Ross and Shandon (USDOE, 1990d).

3.2.8 Use of Surface and Ground Water

Ground water is a major source of water supply in the area near the FEMP. Nearby water users include FEMP wells (0.42 mgd), Southwestern Ohio Water Company (Industrial Supply, 17.38 mgd), Delta Steel (Industrial Supply, 1 factory), Albright & Wilson Chemical Company (0.14 mgd) and Ruetgers-Nease Chemical Company (0.1 mgd) (USDOE, 1987). There are also many other smaller industrial, commercial, agricultural and private ground water users in the area.

The Great Miami River is not a source of drinking water between the FEMP and the Ohio River. There is some fishing in the river, but it is considered unsafe for swimming due to turbulence (USDOE, 1990c).

3.3 Site Description and Site Operations

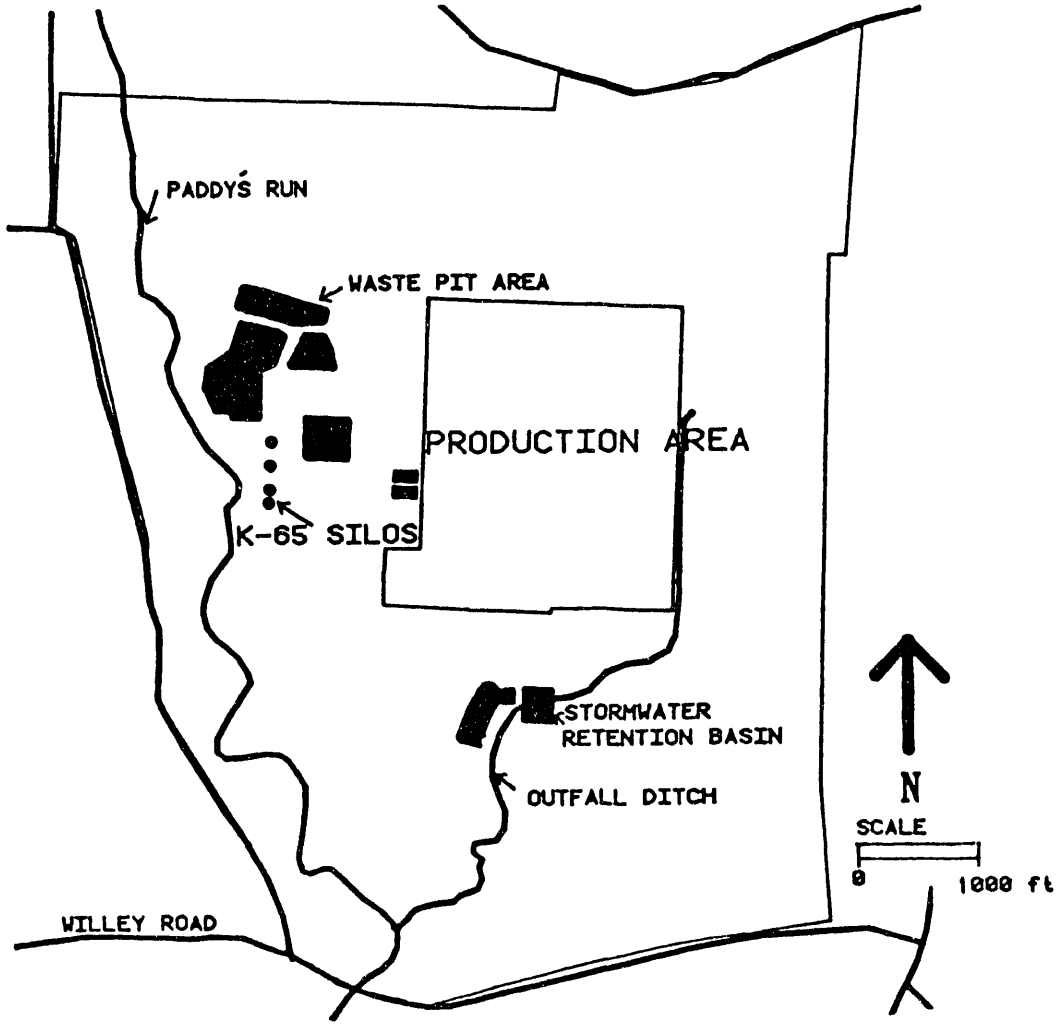
The production area consists of several processing plants and waste pit areas, and is located on approximately 136 acres near the center of the facility (Figure 3-4). The waste storage area is located on the western side of the facility (Figure 3-4).

A wide variety of chemical and metallurgical processes were used at the FEMP. Uranium ore concentrates and recycled materials were converted to either uranium oxides or uranium ingots and billets. These were machined or extruded into tubular form for production reactor fuel cores and target fuel element fabrication. During this process, other chemicals and compounds were utilized.

The most recent large-scale chemical operations consisted of processing enriched uranium scrap residues to produce a uranyl nitrate feed solution. The uranium received had been through one or more chemical separations at other sites, which removed most of the decay products. Purified uranyl nitrate solution was concentrated and then denitrated to uranium trioxide (UO_3) which was then reduced to UO_2 and uranium tetrafluoride (UF_4) for reduction to metal. Uranium-235 enrichment in raw materials ranged up to 2%. The maximum product enrichment was 1.25%, and the average product was slightly depleted (Weston, 1987).

From 1953 through 1955 the FEMP refinery processed pitchblende ore from the Belgian Congo. The pitchblende ore contained the full range of products in the uranium decay chains. Beginning in 1956, the feed consisted of yellowcake (uranium concentrates) from Canada and the United States. Most of the uranium progeny are removed in the production of yellowcake. Radium-226 remains in the yellowcake in amounts that vary with the process used for the yellowcake production. Canadian yellowcake contained higher levels of thorium than yellowcake from the United States and was not processed after 1960. Small amounts of thorium were produced at the FEMP on several occasions from 1954 through 1975.

Figure 3-4. Fernald Environmental Management Project.



3.4 Environmental Problems

3.4.1 Releases From Site Operations

Plant operations resulted in airborne releases and liquid effluent discharges, and have resulted in releases of uranium to the atmosphere. Liquid waste streams are treated and discharged in compliance with NPDES (National Pollution Discharge Elimination System) permits. Untreated storm water runoff from the production area was routinely discharged to the Great Miami River, and excess storm flows were periodically discharged to Paddy's Run.

3.4.2 Waste Site Releases

Both radioactive and nonradioactive wastes were generated as a result of activities conducted at the facility. Wastes were stored in pits, silos and drums. The waste storage area includes six low-level radioactive waste storage pits, two earthen-bermed concrete silos containing K-65 residues (residues resulting from the pitchblende refining process), one concrete silo containing metal oxides, two lime sludge ponds, a sanitary landfill, two fly ash piles and a burn pit. No solid waste materials have been disposed of in the Waste Storage Area since 1985.

Documented waste site releases include radon emissions from the K-65 silos, uranium in runoff from the production and waste storage areas and a plume of uranium in ground water south of the facility.

3.5 Environmental Monitoring and Environmental Restoration

3.5.1 Environmental Monitoring Program

At the FEMP samples of air, water, soil and other media are collected to assess the site's impact on the surrounding area. Descriptions of the samples taken each year and the results of the environmental monitoring program are available in Annual Environmental Monitoring Reports (e.g. USDOE, 1990c)

Sixteen continuous air monitoring stations are operated to measure particulate and radionuclide concentrations in air. Seven of these stations are offsite. Radon monitors are located around the K-65 silos, along the FEMP boundary fence, at other locations onsite, and at locations offsite. These monitors give quarterly estimates of radon concentrations at these sampling locations.

Soil and grass samples are taken onsite and offsite and analyzed for uranium and fluoride. Offsite produce and milk samples are also analyzed for uranium.

Liquid effluents to the Great Miami River are sampled and analyzed. Surface water and sediment samples are taken along the Great Miami River and Paddy's Run and are analyzed for radionuclides. Fish are collected along the Great Miami River and analyzed for uranium.

Ground water samples are taken in both onsite and offsite wells and analyzed for both radioactive and nonradioactive constituents. Onsite wells are screened at different depths to monitor ground water quality. Many wells have been installed at the facility to support the RCRA ground water assessment program and the RI/FS characterization effort.

3.5.2 Environmental Restoration

In July, 1986 a Federal Facilities Compliance Agreement (FFCA) pertaining to environmental impacts associated with FEMP operations was jointly signed by the United States Department of Energy and the United States Environmental Protection Agency. The FFCA was entered into to ensure compliance with existing environmental statutes and regulations, including the Clean Air Act (CAA), the Resource Conservation and Recovery Act (RCRA), and the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA). The FFCA was amended by a Consent Agreement under section 120 and 106(a) of CERCLA which became effective in June 1990. A Remedial Investigation and Feasibility Study (RI/FS) is in progress pursuant to CERCLA, as required by the FFCA.

Thirty-nine separate units that require investigation have been identified at the FEMP. These units have been segregated into logical groupings of units (called operable units) that are similar based on physical features, contaminant sources or types and schedules or likely responses (USDOE, 1990a). The five identified operable units at the FEMP are: 1) waste storage area, including the six waste pits; 2) solid waste units including the sanitary landfill, lime sludge ponds and fly ash piles; 3) facilities and suspect areas including the production area; 4) special facilities including the K-65 silos and 5) environmental media including regional ground water and soils.

4 PROBLEMS TO BE STUDIED

4.1 Criteria for Problem Selection

Problems to be assessed in this pilot study were chosen using a number of factors and criteria, and were selected based on a qualitative critical review, rather than a formal ranking system. Criteria included identification by USDOE and contractor site managers, results of screening assessments, and professional experience with the sites being studied. For example, problems identified by the MEPAS (Multimedia Environmental Pollution Assessment System) model as implemented by the USDOE Environmental Survey were considered for assessment. Professional knowledge and experience was also applied to identify potentially important problems that were scored low by the MEPAS model as implemented by the USDOE Environmental Survey. Public and USDOE concern were important factors in choosing problems to analyze, and emphasis was placed on problems where offsite transport and exposure has already occurred.

4.2 Problem 1: Radon Emissions from the K-65 Silos

4.2.1 Description of Problem

Two silos (known as the K-65 silos) at FEMP contain waste from the refining of pitchblende ore from the Belgian Congo. The pitchblende ore contained the full range of decay products within the uranium decay chains. The waste stored in the silos contains approximately 1,600 Ci (5.9×10^4 GBq) of Ra-226, which decays to radon gas. The silos are each 30 feet tall, 80 feet in diameter and are surrounded by an earthen embankment. The silos are located in the waste storage area on the western side of the facility (Figure 3-4).

The silos are not air-tight, and radon gas escapes to the outside atmosphere. Radon concentrations above natural background have been measured at the site boundary. Exposure to radon and radon progeny is associated with an increased risk of lung cancer.

4.2.2 Reasons for Selecting This Problem

This problem was selected for analysis for several reasons. There is a potential for offsite exposures because the K-65 silos are located near the western site boundary, and elevated radon concentrations have been detected at the boundary and at nearby residences.

A major reason for assessing the risk associated with the K-65 silos is that this environmental problem was the highest scoring ranking unit for the FEMP under the MEPAS scoring system as implemented by the USDOE Environmental Survey. The MEPAS modeling of this problem contained a number of conservative assumptions which resulted in this large score. One of the more important conservative assumptions was the large emission rate derived from radon measurements taken at near cracks on the silo domes (NUS, 1990).

4.2.3 Transport Pathways and Potential Receptors

The transport pathway of concern for this problem is air. Radon from the K-65 silos can be transported offsite in air and expose nearby residents to the radon gas and its decay products through inhalation. People living in population centers at greater distances from the site may also be exposed to radon and radon progeny from the K-65 silos, but concentrations are diluted to background levels within a few miles.

4.3 Problem 2: Uranium Contamination of Ground Water

4.3.1 Description of Problem

Uranium was detected in concentrations above background at several locations offsite. Storm water runoff from the production area and the waste storage area may be a continuing source of uranium contamination in surface and ground water at the facility.

4.3.2 Reasons for Selecting This Problem

The major reason for selecting this problem for analysis is the potential for offsite exposures. Elevated concentrations of uranium were detected in downgradient, offsite wells. Uranium ingested in water and in food is of concern for two reasons: 1) the potential for toxic effects and; 2) the potential for an increased risk of cancer.

Environmental problems associated with uranium contamination of ground water scored high in the USDOE Environmental Survey MEPAS ranking (contamination from liquid discharges, contaminant releases from waste pits) which supports the decision to include this problem in the pilot risk assessment for the FEMP.

4.3.3 Transport Pathways and Potential Receptors

The principal transport pathway for uranium is through ground water which can move offsite and expose nearby residents through ingestion in drinking water. Other potential transport and exposure pathways include uptake by plants and animals and subsequent ingestion by people. Ground water south of the facility eventually discharges to the Great Miami River, and there is a potential for exposure to uranium in fish in the future.

4.4 Problems Identified for Future Assessments

The analysis presented here considered only the exposures and risk associated with the ingestion of uranium associated with ground water contamination south of the facility. Problems identified for future assessments include an integrated assessment of uranium released by the facility to air, soil, ground water and surface water.

5 RISK ASSESSMENT FOR RADON EMISSIONS FROM THE K-65 SILOS

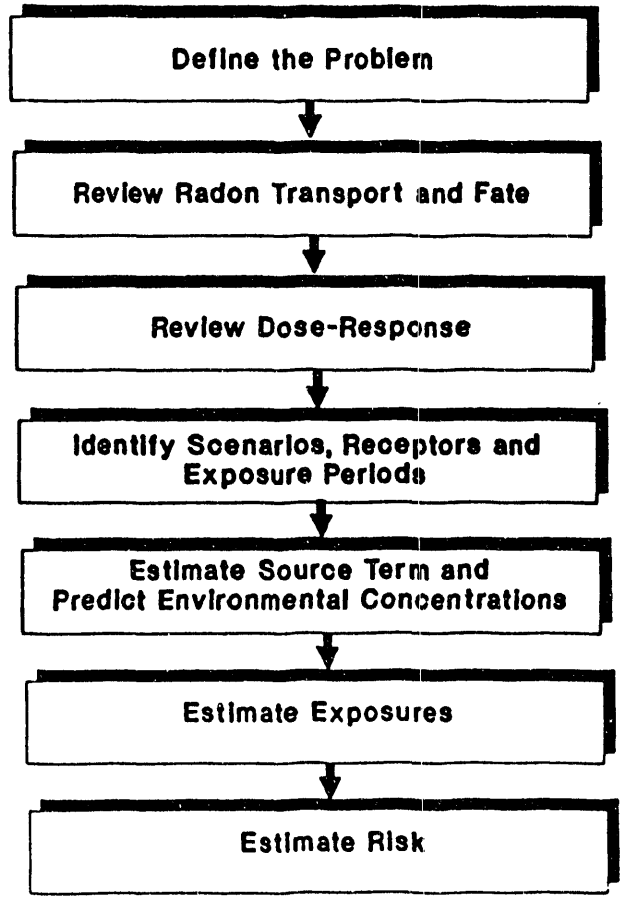
5.1 Problem Summary and Outline of Steps in the Assessment

5.1.1 Steps in the Assessment

Major steps in the risk assessment for radon released by the K-65 silos are listed below, and are summarized in Figure 5-1.

1. Define the Problem (section 5.1.2).
2. Review information on radon and radon daughter transport and fate (sections 5.3.1, 5.3.3).
3. Review dose-response information, and identify appropriate risk factors (section 5.3.5).
4. Identify release scenarios, exposure scenarios, receptors and exposure periods (section 5.4).
5. Develop estimates of source term and predict environmental radon (and radon progeny) concentrations (section 5.5).
6. Estimate individual and population exposures (section 5.6).
7. Estimate lifetime individual and population risks (section 5.7).

Figure 5-1. Steps in the risk assessment for radon emissions from the K-65 silos.



5.1.2 Problem Summary

Two concrete silos, known as the "K-65" silos, contain residues from the processing of pitchblende ore. The residue contains high concentrations of radium-226. Radon gas generated by the decay of the radium in the waste escapes to the outside atmosphere.

The major transport pathway of concern is airborne transport. It is possible that material in the silos has migrated into the soil and ground water, but available data do not demonstrate this conclusively. This possibility is being investigated as part of the ongoing RI/FS studies at the FEMP.

The exposure pathway of concern for radon emitted by the silos is the inhalation of radon progeny. The two radon release scenarios of concern are 1) the documented, routine release of radon gas from the silos and; 2) a potential release of all radon available in a silo dome head space in the event of a silo dome failure.

Important receptors in the area of the FEMP include onsite workers, and residents in houses located along the western and southern site boundaries. The population within 5 miles of the facility represents an important receptor, and population centers within 5-10 miles of the facility may also be of concern in terms of total population risk.

The K-65 silos are being characterized in the site RI/FS study currently underway at the FEMP. Corrective steps were recently taken to eliminate routine radon releases and to prevent an accidental loss of radon in the event of a silo dome failure.

5.2 The K-65 Silos

5.2.1 Physical description

The K-65 silos were built in 1951 and 1952 to store residues from the processing of pitchblende ore to extract uranium. The term "K-65" refers to radium bearing wastes generated during the extraction step. Table 5-1 shows typical compositions of the original pitchblende ore. Most of the residue stored in the silos was generated at the Mallinckrodt Chemical Works in St. Louis Missouri, using ore from the Belgian Congo. During 1957 and 1958, some residue from the processing of Australian ore at FMPC was added to one of the silos.

The K-65 silos are located south of the waste pit area on the west side of the FEMP site (Figure 3-4). The silos are each 80 feet in diameter with vertical walls approximately 26 feet high. The silo domes are approximately 36 feet high. The walls and silo domes are constructed of reinforced concrete, and the floors are constructed of 4 inches of concrete overlaying layers of gravel, asphaltic concrete and compacted clay.

The exterior surfaces of the silos exhibited significant deterioration and were repaired in 1964. After the repair work was completed, an earthen berm was built to the top of the walls to reduce the stress on the walls. In 1986 a temporary dome cover, composed of structural steel members and plywood sheeting covered with a weatherproof membrane was installed across the silo domes. In 1987 a rigid polyurethane foam coating was applied to the exterior surfaces of the two silo

domes to reduce weathering, temperature changes within the silos and the associated pumping action that promotes radon gas emissions.

Table 5-1. Typical uranium concentrates (Harrington and Ruehle, 1959 as cited in USDOE 1990a).

Constituent	Acid Leach, Ion Exchange	
	Belgian Congo Ore	Australian Ore
U_3O_8 (%)	48 - 73	55 - 70
H_20 (%)	0.1 - 0.5	0.2 - 0.5
As (%)	<0.02	0.02 - 0.2
B (ppm)	80 - 600	4 - 40
Cu (%)	0.4 - 2	0.01 - 1
Halides (%)	0.02 - 0.03	0.05 - 0.15
Fe (%)	0.4 - 4	>4
MoO_3 (%)	0.2 - 0.6	0.01 - 0.08
P_2O_5 (%)	0.25 - 0.7	0.1 - 1
Na (%)	0.07 - 2	2
So_4^{-2} (%)	1 - 37	0.5 - 2.5
V_2O_5 (%)	0.002 - 0.004	<0.02

A structural analysis of the silos was completed in 1986 (Camargo Associates, 1986). The results indicated that the center 20 foot diameter portion of the dome tops are structurally unsound for a load greater than the existing static dead load. The silo domes were estimated to have no life expectancy (Camargo Associates, 1986).

5.2.2 Silo Contents and Emissions

The silo residues are not homogeneous, and consist of silicates, hydroxides, carbonates and sulfates. Inorganic constituents include magnesium and lead. The radionuclide inventory has been estimated to include a total of 7 Ci (260 GBq) of uranium (0.71% U-235), 3,300 Ci (1.2×10^5 GBq) of radium-226 and 1,180 Ci (4.4×10^4 GBq) of thorium-230 (BNI, 1990).

The silos were filled to a height of approximately 23 feet -- leaving a head space above the residues. Some of the radon gas generated by the radium in the residues collects in this head space. However, most of the radon generated by the residues is trapped in the waste and does not reach the head space.

Based on gas samples taken from the silo head spaces, it has been estimated that there are about 33 Ci (1221 GBq) of radon available in the head space of each silo (Silo 1: 34 Ci; Silo 2: 32.5 Ci; BNI, 1990). Using the 33 Ci value, and assuming no loss to the outside environment, it has been estimated that about 6 Ci/day (222 GBq/day) of radon are added to each silo head space each day (BNI, 1990).

No measurements of the radon flux from the surface of the silo domes have been made since the foam coating was applied in 1987. Measurements made before 1987 range from 3 pCi/m^2-s to 13 x 10^7 pCi/m^2-s (0.4 Ci/year - 8.9 x 10^5 Ci/year; 14.8 GBq/year - 3.3 x 10^7 GBq/year) Hagee et al (1985)). The larger fluxes were measured near cracks on the silo domes.

Two mechanisms for radon release from the silos have been suggested (IT, 1989). These mechanisms are diffusion through the silo domes and free air exchange between the pore volume inside the domes and the outside atmosphere (IT, 1989). Borack (1985) and Boback et al., (1987) calculated the emission rate due to diffusion through the silo domes using one-dimensional steady state diffusion equations (Colle et al., 1981). The resulting emissions estimate for both silos was 60 Ci/year (2.2 x 10^3 GBq/year).

Temperature and pressure changes in the silos were studied in 1987 (Grumski, 1987). The pressure data gathered in this study indicated that both of the silos exchange gas freely with the surrounding atmosphere (IT, 1989; Grumski, 1987). Grumski (1987) calculated the rate of radon release by free air exchange, assuming that each silo exchanges an average of 1000 ft^3 of gas per day with the outside atmosphere, a radon concentration of 3 x 10^4 pCi/cm^3, and a dome void volume of 25,000 ft^3. This calculation yielded an estimated radon release rate of 600 Ci/year (2.22 x 10^4 GBq/yr) for both silos.

IT (1989) updated the analysis by Grumski (1987) by incorporating other estimates of the dome void volume (average 43,758 ft^3 ; IT, 1989). The resulting estimate for the radon release rate due to free air exchange was 512 Ci/yr (1.9 x 10^4 GBq/yr) per silo. IT (1989) suggested a total release rate of 1084 Ci/yr (4.01 x 10^4 GBq/yr)-- 512 Ci/yr from each silo (due to free air exchange) plus 60 Ci/yr from diffusion through the domes.

Previous modeling and risk assessment studies have used a wide range of values as estimates of the emissions rate. These estimates include 60 Ci/yr (2.22 x 10^3 GBq/yr) (Boback et al., 1987; based on estimates of diffusion through the domes); 600 Ci/yr (2.22 x 10^4 GBq/yr) (Grumski, 1987); 650 Ci/year (2.4 x 10^4 GBq/yr) (Ijaz et al, 1990); 1084 Ci/yr (4.01 x 10^4 GBq/yr) (IT, 1989); and 10,960 Ci/year (4.06 x 10^5 GBq/yr) (NUS, 1990).

5.2.3 Current Status

The K-65 silos (silos 1 and 2) are part of Operable Unit 4, which also includes two metal oxide silos (silos 3 and 4), piping and tanks that lie beneath the silos, and the earthen berm surrounding the K-65 silos. A RI/FS study is underway for Operable Unit 4 and sampling and analysis programs are ongoing.

As required in the Consent Agreement under CERCLA, an engineering evaluation and cost analysis (EE/CA) was performed for the K-65 silos to evaluate the feasibility of taking corrective action in response to short-term threats to public health and the environment (BNI, 1990).

Based on the analysis presented in the EE/CA (BNI, 1990) the preferred alternative for the corrective action was to cover the residues in the K-65 silos with a bentonite slurry. This step was completed in 1992. The addition of a layer of bentonite to the surface of the waste in the silos should substantially reduce or even

eliminate routine radon emissions, and is expected to contain radon gas in the event of a silo dome collapse.

5.3 Radon

5.3.1 Physical Properties

Radon gas (radon-222) is a natural decay product of radium-226, the fifth daughter in the decay chain of uranium-238. Uranium-238 and radium-226 are present in most rocks and soils, and radon gas easily diffuses through soil, air, and water, so radon is ubiquitous in the environment. Natural background concentrations vary widely depending on local conditions.

Radon-222 decays with a half-life of 3.82 days through a series of solid, short-lived radioactive decay products called "radon progeny" or "radon daughters" (Figure 5-2). In addition to radon, itself, two of its decay products, polonium-218 and polonium-214, also emit alpha particles which can damage sensitive cells, ultimately leading to cancer. Other progeny are also radioactive, but their relative contributions to the health effects of radon are small.

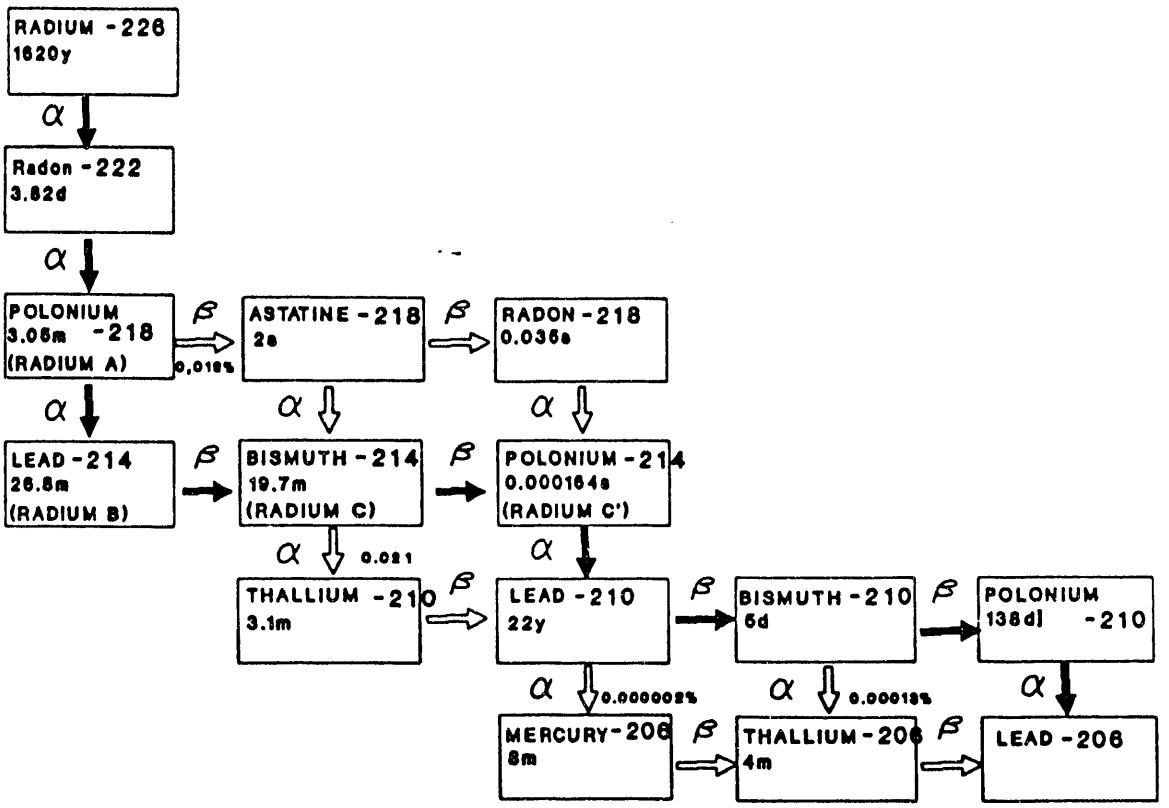
Because alpha radiation has low penetrating power (alpha particles can be stopped by a piece of tissue), they have little health significance unless their source is in close contact with sensitive cells. Only exposure to alpha radiation from radon and/or its decay products that have been inhaled or ingested into the body are of concern. For the situation being evaluated here, inhalation is by far the most important route of exposure and lung cancer is its primary health effect.

Although radon-222 is inert and diffuses readily into the air, its progeny are charged particles which are strongly attracted to any nearby surface, including other particles in the air. They are also short-lived, with half-lives of 3.05 minutes and 0.000164 seconds, for polonium-218 and polonium-214, respectively. They are therefore removed quickly from the air, either by direct plate-out on environmental surfaces, by deposition of larger and heavier atmospheric particles to which they attach, or by radioactive decay to less harmful nuclides. The mean lifespan of radon-222 in the air is 5.5 days and that of the important radon progeny is 4.4 minutes or less.

With respect to health effects, "radon" is really a complex mixture of the initial radon-222 plus its decay products which varies with local conditions, so an artificial unit of concentration has been developed to quantify the effects of the mixture as a whole. The unit was initially defined to quantify concentrations in mines and the conditions represented are occupational.

One "working level" (WL) of a mixture of radon-222 and its progeny is defined to be any combination in one liter of air that produces 1.3×10^5 MeV of alpha-particle energy. This is the amount of alpha energy produced by 100 pCi (3.7 Bq) of radon-222 in equilibrium with polonium-218, lead-214, bismuth-214, and polonium-214. Figure 5-3 shows the accumulation and decay of progeny with time. Equilibrium is reached in an hour or two, after which the total activity of the mixture remains constant at one WL per 100 pCi/L (3.7×10^3 Bq/m^3) of radon-222.

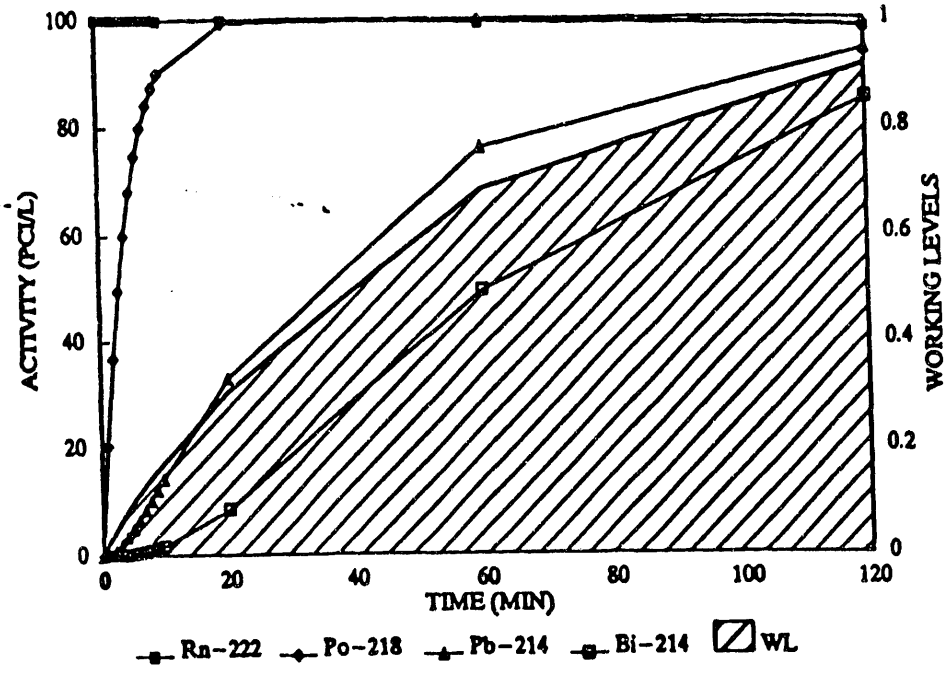
Figure 5-2. The radon decay chain.



To account for nonequilibrium conditions, an "equilibrium factor" (EF) is defined which quantifies the effect of disequilibrium on the total alpha energy of the mixture per unit activity of radon-222. The equilibrium factor is 1 for complete equilibrium (1 WL per 100 pCi/L radon-222) and less than 1 for younger mixtures and for mixtures in environments with high plate-out rates (e.g. mixtures containing high concentrations of particles from smoking).

Normal outdoor background radon mixtures have an equilibrium factor of about 0.7 and normal indoor background radon mixtures have an equilibrium factor of about 0.5 in houses with smokers and 0.3 to 0.4 in houses without (NRC, 1991).

Figure 5-3. Decay of radon and development of radon progeny with time in the absence of deposition to environmental surfaces (working levels per 100 pCi/L radon-222 in a plume).



5.3.2 Indoor vs. Outdoor Concentrations

There are significant differences between radon concentrations and exposures indoors and outdoors caused by physical differences in these environments.

- Buildings trap radon emitted from the ground. The tighter the building and the lower the exchange rate with outside air, the higher will be the concentration of radon inside.
- Many buildings are in direct contact with the ground through basements or slabs, which increases the rate of entry of radon from the ground many-fold.
- Buildings and contents provide high surface-to-volume ratios, which increase plate-out of radon progeny and reduce the equilibrium factor.

The combined effect of these differences is to produce on average a two-to-four-fold difference between indoor and outdoor radon concentrations. Annual average background radon concentrations outdoors ranged in one survey from 0.16 to 0.57 pCi/L with a median of 0.39 pCi/L (14 Bq/m^3) (Hopper et al., 1990). Annual average background concentrations in the living areas of single-family homes in one recent survey had a geometric mean of about 1.5 pCi/L (56 Bq/m^3) and a broad range extending past 20 pCi/L (740 Bq/m^3) in a few exceptional houses (Cohen and Shah, 1991). These authors estimated that their methodology produces an upward bias in these data of about 20%. Other surveys have produced similar results.

5.3.3 Radon and Radon Progeny in a Plume

Radon exposure studies are normally done for spaces having a continuous source of supply, such as mines and houses, and they normally do not include effects of deposition. Because the K-65 silos at the FEMP are point sources that produce a plume of radon, the mathematics of radon daughter formation and decay are somewhat different from the standard equations normally used (the classic paper in this area is Evans (1969)). We have also added to the equations the effects of deposition of radon progeny. The differences between results with and without a continuous source of supply are not significant, but the differences with and without deposition can be as high as 30%.

Rowe (1991) presents the necessary equations for decay without replenishment from background sources, and development of WL per unit radon-222 activity as a function of travel time in a plume. It is assumed that the effective deposition rate for radon progeny outdoors must be about 0.8% per minute to yield an outdoor equilibrium factor of 0.7. Applying this deposition rate to a radon plume yields the following summary equations expressing radon equilibrium factor as a function of travel time,t:

$$EF(0.8\%,t) = 0.716 - 0.0459\, e^{-t/4.25} - 1.70\, e^{-t/29.5} + 1.02 e^{-t/23.2}$$

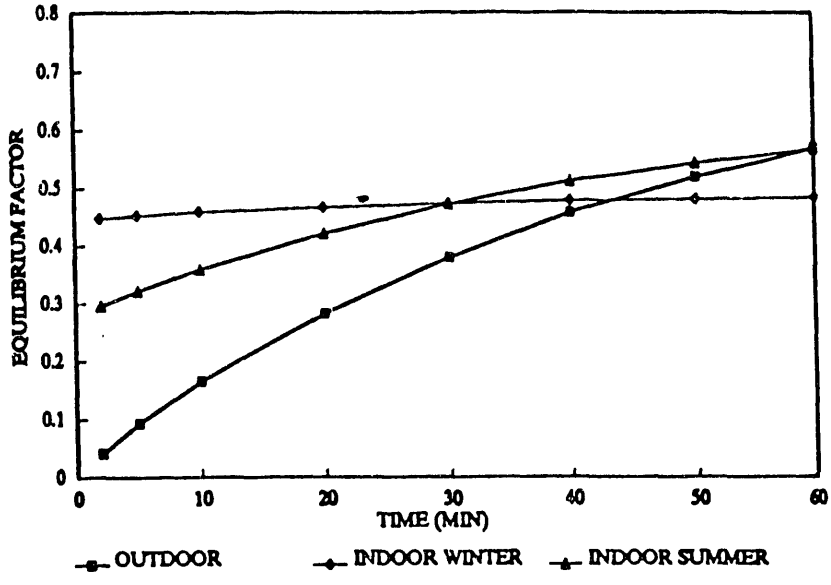
The working levels of radon progeny estimated by this equation are included in Figure 5-3. Note that at equilibrium, this equation yields an outdoor equilibrium factor of 0.716.

Estimated exposure to radon progeny as they move from outdoors to indoors depends on assumptions about the success of particles in entering the interior. At one extreme representing open windows, all outdoor radon progeny enter and leave freely, and the only effect of the house is to increase the age of the radon mix by the residence time of the air indoors and to increase the deposition rate of the progeny because of a higher surface to volume ratio indoors. At the other extreme, all particles are filtered out by the structure and only radon gas enters the interior. This extreme is indistinguishable from an indoor source of radon gas, except that indoor sources are normally in basements rather than higher portions of the structure. Reality is usually somewhere between these extremes.

Rowe (1991) has shown that the equilibrium factor produced by outdoor radon from a plume moving indoors remains much more stable than that for

outdoor radon over a broad range of conditions and travel times (Fig. 5-4). As a result, it is reasonable to use the standard assumption of an indoor equilibrium factor of 0.5 for radon from all sources.

Figure 5-4. Indoor and outdoor equilibrium factors in a radon plume as a function of travel time.



5.3.4 Exposures and Dose-Conversion Factors

The unit of exposure to mixtures of radon progeny is a "working level month" (WLM), which is defined to be cumulative exposure to a concentration of 1 WL for 170 hours (a month's worth of time in a mine for miners of the 1950's). A full, non-occupational month is 4.3 working months and a full year is 51.5 working months.

From the above, total exposure to radon progeny is expressed as:

(pCi/L Rn-222) * (EF of the mixture) * (hours exposure/170 hr/WLM) in WLM.

The internal absorbed dose of alpha-particle energy to lung tissues per unit external radon exposure depends on a broad range of environmental and physiological variables, including:

- Size distribution of atmospheric particles;
- Fraction of progeny attached to particles;
- Equilibrium factor;
- Lung morphology;
- Depth of target cells;
- Breathing volume;
- Particle deposition fraction;
- Thickness of mucus; and
- Mucus transport rate.

These variables are modeled in minute detail by specialists, but for risk assessment, the results are normally collected into a single dose-conversion factor that quantifies the ratio of the dose to the lung per WLM exposure to the general public in houses and to miners at work. This is done because all of the useful dose-response information available on effects of exposure to radon come from studies of miners (see below), so exposures in other environments must be compared with those in mines.

Various dose-conversion factors have been used for risk assessment. There is general agreement that the obvious differences between exposures to miners and to the general population tend to cancel out, so that the net effective difference in dose per unit exposure is within a factor of 2 (i.e, 0.5 to 2.0) (James, 1988). The earlier analyses yielded dose-conversion factors greater than one and the more recent yielded factors less than one. The most recent study estimates the dose-conversion factor to be 0.7 for adults and 0.8 for infants and children (NRC, 1991).

5.3.5 Dose-response Relationships

There is considerable controversy over the quantitative relationship between magnitude of exposure to alpha radiation from radon progeny and resulting risk of lung cancer (NRC, 1988). Much of the controversy arises from inadequacies of the data available from which the relationship must be estimated.

- All of the data used for quantitative studies are from miners exposed to doses many times higher than those in other environments;
- The data are badly confounded by cancers from smoking, which is almost universal among miners; and
- There are other less obvious but nonetheless serious methodological deficiencies that increase the uncertainty of the results (Hamilton, 1989).

Studies of miners have consistently shown an increased risk of lung cancer with increased cumulative exposure to radon (see, for example, Lubin et al., 1990). The problem with evaluating the health effects of radon exposures to the public arises not so much from the many inadequacies of these studies, themselves, but from a need to extrapolate results from the high doses received by miners to the much lower doses received by the public. Extrapolation to such different conditions and people requires a much higher level of understanding of the effects of all important variables than does extension to other, similar conditions.

Epidemiological studies of lung cancer risk for non-occupational exposures to radon have repeatedly failed to demonstrate a statistically significant relationship between dose and risk. Some have even shown lower lung cancer rates in areas with higher radon exposures (Cohen, 1991). This is partly from lack of understanding of the relative importance of the many variables involved, and partly because the risk, if any, is so very low, that the statistical power of the tests is correspondingly low, and huge sample sizes are required for any realistic study.

The most important question in low-dose extrapolation is a universal assumption that there is no threshold dose below which risk of lung cancer is zero. There are cellular repair mechanisms that could produce such a threshold, and epidemiologists' inability to demonstrate dose-related responses at low doses also

argues for one. But there are no consistent data that can provide a basis for estimating where this threshold might be.

Lacking the necessary data, we must assume the simplest relationship which, at the current state of knowledge, is a linear, nonthreshold function extrapolated from the high doses of miners to the much lower doses of the general public. This extrapolation has exceedingly high uncertainty and, given the high probability of an effects threshold, the true value of the risk factor for radon at very low doses could well be zero.

The BEIR IV Committee estimate of 3.5×10^{-4} deaths/WLM (NRC, 1988) is a value that is widely used to estimate cancer risks associated with radon and radon daughter exposure. An update to the BEIR IV study (NRC, 1991) estimated the differences between the exposure to dose relationship in mines and homes. Using a dosimetric model, NRC (1991) found that the dose of alpha energy per unit exposure delivered to target cells in the respiratory tract tends to be lower in homes than in mines. NRC (1991) estimated the reduction in dose to be about 30% for adults of both sexes and about 20% or less for infants and children. Puskin (1992) adjusted the BEIR IV (NRC, 1988) radon risk model to reflect these differences. The resulting risk for the general population is about 2.2×10^{-4} lung cancer deaths /WLM.

ICRP (1991a) reviewed the epidemiological data and lung cancer risk projection models that have been used to estimate lung cancer risk associated with exposure to radon and radon progeny. Table 5-2 lists the studies reviewed by ICRP (1991a) and the associated risk factors. ICRP (1991a) concludes that these studies have a range of $1\text{-}4 \times 10^{-4}$ lung cancer deaths /WLM, and that the range is due partly to the different projection models applied and to the different baseline cancer rates in the reference populations used.

Table 5-2. Dose-response function for exposure to radon and radon progeny (modified after ICRP, 1991a).

Source	Excess Lifetime Lung-Cancer Mortality (deaths/10^6 person WLM)
NCRP (1984)	130
ICRP (1987)*	150
ICRP (1987)**	230
USEPA (1986)***	115-400
UNSCEAR (1977)	200-450
BEIR IV (NRC, 1988)	350

* constant absolute projection
** constant relative projection
*** Puskin and Yang (1988), Puskin and Nelson (1989)

The geometric mean of the values in this range is 2.0×10^{-4} lung cancer deaths per WLM. If it is assumed that this range represents the 90% confidence interval of a lognormal distribution, the geometric standard deviation is 1.53. This

distribution was used to describe the risk factor for radon and radon daughter induced lung cancer in this assessment. This distribution is broadly consistent with that recently published by Puskin (1992) who derived a central estimate of lifetime risk of 2.24×10^{-4} deaths/WLM with a 90% range of 1.4×10^{-4} to 5.7×10^{-4} deaths/WLM.

5.4 Identification of Scenarios and Receptors

5.4.1 Release and Exposure Scenarios

Release Scenarios

Two major release scenarios are of concern for the K-65 silos. The first scenario is a continuing, routine release of radon from both silos. The second scenario is an accidental release of radon from one silo, resulting from a failure of the dome. Exposures and risks associated with background radon concentrations must also be taken into consideration. These scenarios are described below.

ROUTINE RELEASE. The first scenario is a continuing, documented release. Radon gas is generated by the radium-226 in the pitchblende waste in the silos, collects in the head space and escapes to the outside air. The recent addition of bentonite to reduce or eliminate radon emissions from the silos was not included in this assessment. A future analysis, using monitoring data collected after the bentonite cap was installed, will assess the risk associated with the silos after this corrective action.

ACCIDENTAL RELEASE. The second scenario is a loss of the integrity of one silo dome. This is a reasonable scenario because the silo dome covers have been found to be in poor condition, and to have no life expectancy. This event would result in a short term (less than 1 hour) release of the radon accumulated in the head space of one silo, and a continuing high release of radon for several days. As previously indicated, it has been estimated that there are about 33 Ci (1221 GBq) of radon available in the head space of each silo at any given time (BNI, 1990). In the event of a silo dome failure, this radon gas would be released. The facility expects to respond to such an event within 1 week by covering the exposed pitchblende waste with sand (BNI, 1990). During this week, there will be a continuing routine loss of the radon ordinarily generated to the head space.

BACKGROUND. Because radium and radon occur naturally in soil in Ohio, radon and its decay products are natural components of the air. Though the silos are significant sources of radon, the radon that occurs naturally in the air is far greater except for areas close to the silos. The concentrations of natural background radon, and the associated exposures and risks were estimated for comparison.

Exposure Scenarios

Radon is a gas and the important transport pathway is transport of the radon and radon progeny in air. The exposure pathway of concern is the inhalation of radon and radon progeny. The exposure scenarios considered include indoor and outdoor exposure to radon and radon progeny.

5.4.2 Receptors

In this assessment of the risks associated with the radon gas emitted from the K-65 silos, both individual and population (collective) risks were considered. Individual lifetime fatal cancer risks for people living close to the silos is an obvious concern in a health risk assessment. Because of the assumption that even a small exposure to radon and its decay products can lead to some cancer risk, the total societal impact of the release must also be assessed. This is accomplished the use of the population or collective risks to populations living close to the silos.

Important receptors for a risk assessment of the radon gas originating in the FEMP K-65 silos include:

1. onsite workers
2. fenceline residents
3. population within 1 mile of the silos
4. population within 5 miles of the silos
5. population centers within 10 miles of the silos

These receptor populations are summarized in Table 5-3 and are described in more detail below.

Table 5-3. Receptors for radon risk assessment - individual and population (collective) risk.

Receptor	Individual Risk	Population Risk
Onsite workers	individual 500 meters from silos	--
Fenceline residents	individual at fenceline residences	--
Population within 1 mile	individual living within 1 mile	population within 1 mile
Population within 5 miles	individual living within 5 miles	population within 5 miles
Hamilton and Cincinnati, Ohio	individual with average exposure in Hamilton and Cincinnati	population in Hamilton and Cincinnati

Onsite Workers

Individual workers at the FEMP may spend time close to the K-65 silos during a work day. The concentration of radon and radon progeny (WL) is predicted to be highest at 500 meters from the silos (Section 5.5.2.4). This point was used to estimate potential exposure to onsite workers. Because of the ingrowth of

radon progeny, the exposure to radon and radon progeny in units of WL will be different for people working indoors than for outdoor workers. The exposure and risk to both indoor and outdoor workers were estimated in this assessment.

Fenceline Residents

There are a number of houses located along the western and southern boundaries of the FEMP (Figure 5-5). These represent potentially important receptors because of their proximity to the K-65 silos.

Population Within 1 and 5 Miles

The area surrounding the FEMP is primarily rural, with some light industry and a few scattered residences. There are a few villages within 5 miles of the facility.

The 1990 population near the FEMP was estimated for a 16 sector (22.5 degrees) grid with annular rings every mile up to 5 miles from the FEMP (SAIC, 1987; Figure 5-6). Census block maps for 1980 were combined into a working base map onto which the polar grid was superimposed. Where necessary, populations in census blocks were split among sectors based on the percentage of the block's road mileage contained within the sector. The 1980 population was adjusted by an annual growth rate for each census tract (ranged from 1.0 to 2.5% per year) to obtain 1990 population estimates. These population estimates were done by SAIC (SAIC, 1987) in support of emergency response planning. This grid has its center at the physical center of the plant site. The K-65 silos are located approximately 1800 feet from this location. A qualitative assessment of the population data indicated that the distribution of the nearby population among sectors and rings (1-5 miles) would not be significantly affected by the change in the origin of the polar grid (required by the transport models with the emissions source at the silos, not the plant center). The SAIC population distribution for 1990 was used in this assessment. The total estimated 1990 population within 1 mile of the site is 175. Within 5 miles there are approximately 14,729 people (SAIC, 1987).

Population centers within 10 miles

Population centers within 10 miles of the K-65 silos were identified as potentially important receptors for radon originating in the silos. Population centers within 5 miles of the facility were distributed among the 16 sector, 1-5 mile grid cells of the polar grid described earlier. Population centers within 5-10 miles of the facility were identified using 1990 census data (U.S. Department of Commerce, 1992) and associated with a sector and distance for use in later modeling predictions. The two population centers identified for use in the assessment were Hamilton and Cincinnati, Ohio (Figure 5-7). Cincinnati was included as a receptor in this assessment because of its large population, and because its outer suburbs are within 10 miles of the silos.

Figure 5-5. Individual receptors located along the western and southern boundaries of the FEMP.

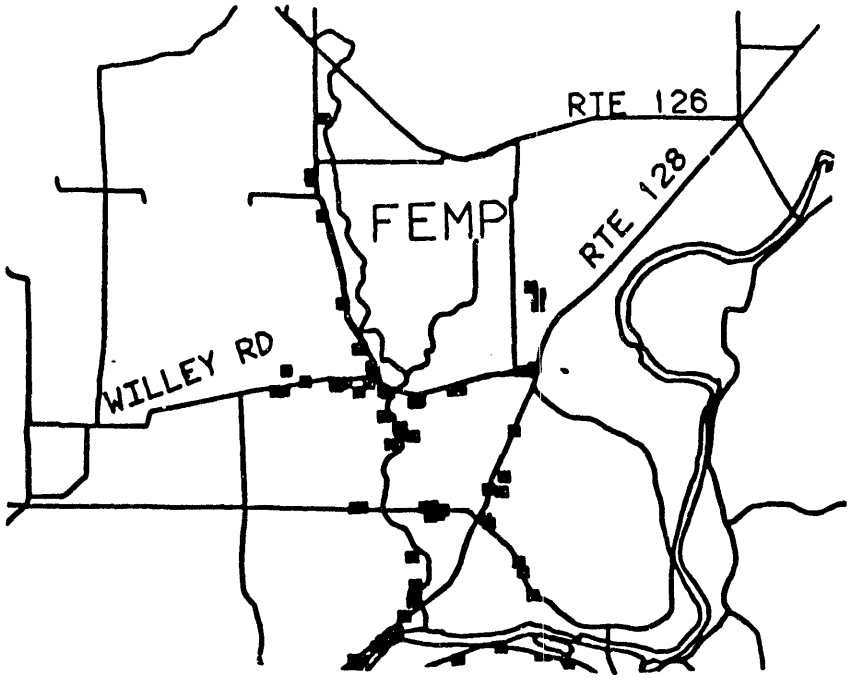


Figure 5-6. 1990 Population distribution within 5 miles of the FEMP. (data from SAIC, 1987; outline at center of polar grid is the FEMP site boundary).

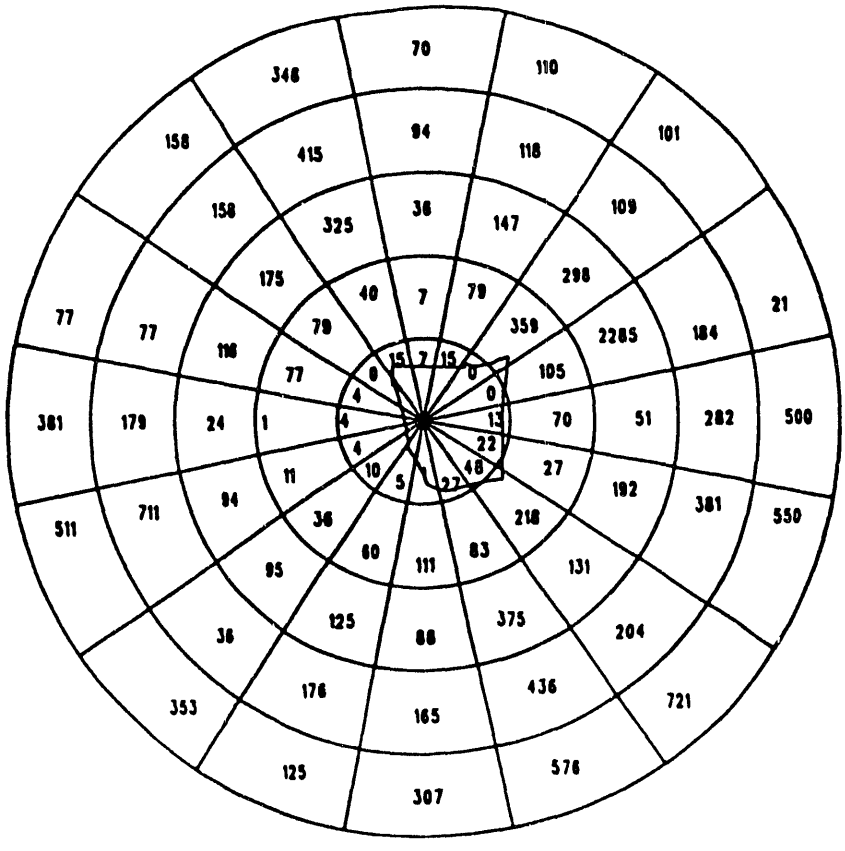
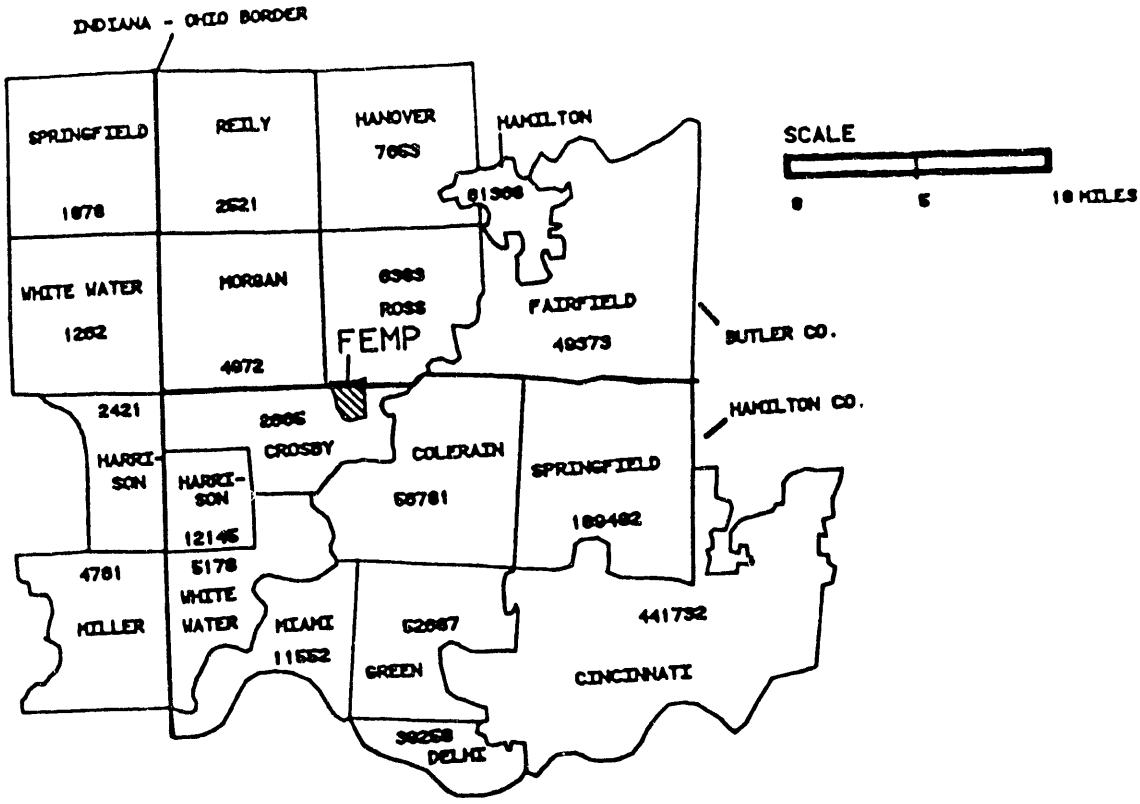


Figure 5-7. Population centers near the FEMP.



5.4.3 Exposure Periods

To estimate exposures to radon and the associated lifetime individual and population (collective) risks, some assumptions must be made concerning the length of time a person may be exposed. The information needed in this assessment includes the length of time a person will live in one place, the amount of time spent indoors and outdoors, and how much time workers spend at the facility. For some of these parameters simple assumptions have been made. Where possible, data available in the literature were used to derive reasonable estimates for parameters or parameter distributions.

Offsite Populations

Time at Home

A number of studies are available which can be used to estimate the amount of time people spend at home. USEPA (1990b) summarizes data from two studies which indicate that adults spend approximately 68 to 71% of their time at home (Chapin, 1974 and Szalai, 1972). Geomet (1981) reported values of 61.6%, 91.5% and 86.2% for a person employed away from home, a housewife and an elderly person, respectively (Geomet, 1991, cited in Puskin, 1992).

In this assessment, it was conservatively assumed that people receive all of their radon exposure where they live. This is not actually the case because most

people will spend part of their day away from home. This assumption was made because the area most impacted by radon from the silos is rural, and people are not likely to travel large distances during the day. For receptors within five miles of the FEMP, this assumption will probably result in an overestimate of exposure associated with the K-65 silos because time spent away from home is likely to be in an area with smaller radon concentrations. For receptors at large distances, the average concentrations of radon associated with the silos does not change significantly from sector to nearby sector where most people are likely to move during the day.

This approach differs from most assessments of indoor radon risks, which usually apply an "occupancy factor" of about 75% (Puskin, 1992). If a simple occupancy factor were applied, the exposure to farmers and other people who do not move far from their houses during the day would be underestimated.

In this analysis, it was assumed that people are exposed to the radon and radon daughter concentrations associated with the plume coming from the silos for the entire day. The differences between indoor and outdoor exposures were included by incorporating estimates of the time spent indoors and outdoors, described in the following section.

Time Spent Indoors

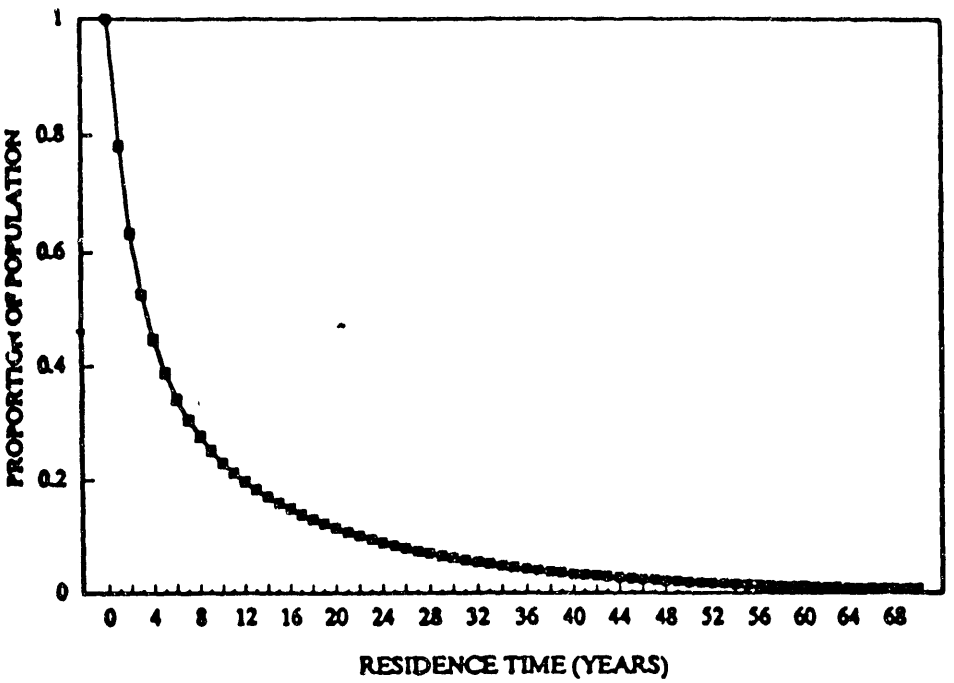
Because exposures to radon and radon progeny are different indoors and outdoors, estimates of exposure require assumptions about the length of time spent indoors and outdoors. Data are available from time-activity studies performed to assess the importance of indoor vs. outdoor pollution. These studies have all found that people spend about 90% of their time indoors USEPA (1990b). A reasonable distribution of times spend indoors and outdoors can be derived from this information and some simple assumptions. The maximum time spent indoors is 100% (24 hours) because some people may spend little or no time outdoors. The minimum time spent indoors is about 40% (9.6 hours) -- this represents farmers and other people who spend most of their waking time outdoors. The most probable amount of time spent indoors is 92.4% (USEPA 1990b). The frequency distribution used in this assessment to describe the fraction of time spent indoors is a simple triangular distribution, minimum at 0.4, maximum at 1.0, likeliest value of 0.924. The fraction of time spent outdoors is 1.0 minus the fraction of time spent indoors.

Average Residence Time

For calculation of routine exposure to radon this assessment assumed a distribution of average residence time based on a model developed by Israeli and Nelson (1992). The total residence time distribution was estimated from current residence time data (U.S. Department of Commerce, 1988; 1989) by modeling the moving process. The fraction of rural households living in the same residence for t years or more was described by the distribution shown in Figure 5-8. The average total residence time for rural households is 7.8 years.

The average total residence time for urban households was estimated to be 4.19 years (Israeli and Nelson, 1992). Urban receptors in Hamilton and Cincinnati are more likely than rural receptors to move to a nearby location, within the area impacted by the radon plume coming from the K-65 silos. A residence time of 70 years was assumed for the urban population because moving could not be assumed to remove the household from the impacted area.

Figure 5-8. Fraction of rural households living in the same residence for t years or more.



Onsite Exposures

Onsite workers at the FEMP were conservatively assumed to work at the facility for 50 years, and to spend 250 days per year at work. Outdoor workers were assumed to spend 8 hours per working day outdoors, and 1 hour indoors. Indoor workers were assumed to spend 9 hours per working day indoors. Background exposures for onsite workers assumed a 70-year exposure period and the time indoors distribution described above.

5.5 Radon and Radon Progeny Concentrations

5.5.1 Background Concentrations

Outdoor Background radon concentrations were based on monitoring data collected at 4 stations located more than 25 km from the silos. Measurements made at these stations in 1988, 1989 and 1990 are given in Table 5-4. These measurements have an arithmetic average of 0.53 pCi/L (19.4 Bq/m^3) and a standard deviation of 0.18. The average concentration in units of working levels (WL) is 2.5×10^{-3} WL (assuming an equilibrium factor of 0.7).

Indoor The distribution of indoor radon concentrations used in this assessment was based on a study of indoor radon concentrations for the United States reported by county (Cohen and Shah, 1991). Diffusion barrier charcoal adsorption collectors were deployed for seven days in homes responding to mail orders. Radon concentration measurements were seasonally corrected (Cohen, 1990) and bias reduction measures employed (Cohen, 1991). The authors believe that these values

may be biased upward by as much as 20% (Cohen and Shah, 1991). Indoor concentration distributions in living areas reported for Hamilton and Butler counties and for three adjacent counties represented in the data base were combined into a single distribution. The resulting distribution for indoor radon concentrations near the FEMP is lognormal, with a geometric mean of 2.45 pCi/L (90.7 Bq/m^3) and a geometric standard deviation of 2.78.

The estimates for indoor and outdoor natural background concentrations of radon and radon progeny are summarized in Table 5-5. The equilibrium factors used to calculate concentrations in working levels (WL) were 0.7 for outdoor concentrations and 0.5 for indoor concentrations.

Table 5-4. Natural background outdoor measurements of radon near the FEMP*.

STATION	Measured Radon Concentration pCi/L (Bq/m^3)		
	1988	1989	1990
BKGR1	0.3 (11.1)	0.4 (14.8)	0.4 (14.8)
BKGR2	0.9 (33.3)	0.6 (22.2)	0.4 (14.8)
AMS15	NS	NS	0.6 (22.2)
AMS16	NS	NS	0.6 (22.2)

* USDOE, 1990c; personnel communication, John Cartarelli.
NS: no sample

5.5.2 Prediction of Radon and Radon Progeny Concentrations From the K-65 Silos

This section describes the models and source terms used to predict the concentration of radon and radon progeny at identified receptors for the routine and accidental release scenarios. Radon and radon daughter concentrations were predicted in units of working levels (WL) for fenceline residents, onsite workers, Hamilton and Cincinnati, and for the center points of the polar grid cells used to describe the population distribution within 5 miles of the FEMP site.

Table 5-5. Natural background radon concentrations near the FEMP.

	OUTDOOR	INDOOR
Radon Conc. (pCi/L)	mean: 0.5 std dev: 0.18	geo mean: 2.45 geo sd: 2.78
Equilibrium Factor	0.7	0.5

5.5.2.1 Atmospheric Dispersion

Dispersion and resulting concentrations of radon and radon progeny from emissions of radon gas from the two silos were modeled using an adaptation of the

sector-averaged Gaussian plume model from CAP88-PC (Parks, 1992). This model predicts annual average concentrations from a point source, averaged over radial sectors corresponding to the aggregation level of the meteorological data used. We corrected the gaussian dispersion equations used in CAP88-PC to the current standard form, which uses a more up-to-date method of horizontal averaging, (Brenk et al., 1983) and derived equations for decay of radon and radon progeny with time for radon in plumes from a point source that include deposition of particulate radon progeny (Section 5.3.3). The equations of Section 5.5.3 estimate an outdoor exposure in WL about 30% lower than the standard equations derived by Evans (1969). This difference is attributable almost entirely to including deposition of radon progeny.

The basic model has 16 radial sectors with rings at one-mile intervals to five miles from the source. Concentrations of radon progeny were calculated at the center of each sector-ring area and applied to the entire population of that area. Meteorological data were entered as joint distributions of stability class (Pasquil A through G) and wind speed by sector (as measured at 10 meters). The model assumed wind speeds were in the middle of the reported ranges (i.e., 1, 3, 5, 7, 9, and 11 m/s). Receptor height, stack height, and mixing height were assumed to be 2, 10, and 1585 m, (Draxler and Hefter, 1981) respectively.

Concentrations and exposures at specific locations, such as near-by residences, monitoring stations, and distant cities, were modeled by inserting the appropriate radial sectors, distances, and populations in place of the standard five-mile modeling grid.

This model was implemented as a Lotus 1,2,3 spreadsheet, which makes it exceedingly quick and easy to use and provides direct access to the computing and graphics capabilities of the Lotus software. Results were validated against those of CAP88-PC (Parks, 1992) and found to agree within the one to two percent attributable to the small changes in equations.

5.5.2.2 Radon Source Term

A wide range of estimates has been made for the amount of radon emitted from the silos. In this analysis emissions were back-calculated from radon measurements at site monitors, measurements of site background, and results of radon dispersion calculations with the sector-average Gaussian plume model.

Figure 5-9 shows the locations of the monitoring stations used to estimate the source term. The model was run for a unit emission (1 Ci/yr, 3.7×10^{10} Bq/yr) with receptors at each site monitor, producing 16 source coefficients expressing the predicted concentrations at the monitors per unit emission from the source -- pCi/l per Ci/yr -- (Fig. 5-10). We then used least-squares regression analysis on the equation,

$$\text{Measured Concentration} = (\text{Source Term}) * (\text{Source Coefficient}),$$

to estimate the source term that minimizes the mean-square difference between the measured and the predicted concentrations over the 16 monitors. This yields an estimated source term of 1150 Ci/yr (4.0×10^4 GBq/yr). This value is similar to the estimate of 1084 Ci/yr (4.3×10^4 GBq/yr) made by IT (1989) for the years 1951 to 1984.

Other approaches to estimating the source term for the calculated source coefficients are possible. For example, the average source emission rate derived from all of the source coefficients can be used (this results in an estimate of about 2100 Ci/yr). Section 5.8.2 presents an uncertainty analysis for the source term estimate, using the range of source terms estimated from the monitoring data (575 to 4025 Ci/year). Including a range of possible source terms did not strongly influence the final risk estimates (see Section 5.8.2).

Figure 5-11 shows individual estimates of the source term from the data at each monitor. The spatial variability of this estimate is close to the factor of 2 that should be expected from predictions of annual averages using a Gaussian model (Harrison and McCartney, 1980). Figure 5-12 shows, as expected, that the Gaussian model over-predicts close to the source (the Gaussian equation goes to infinity at zero distance).

The model also under-predicts at the S and NNW monitors, both of which are located in a creekbed that runs past the silos and which have the highest radon concentration measurements (Fig. 5-12). These radon peaks are apparently caused by gravitational flow of cold, heavy air in the creekbed under low-wind conditions. This is called "the drainage effect" by site meteorologists (P. Spots, personnel communication September 1991). The effect appears to be especially strong in the measurements at the southern monitors in 1990. The nearby monitors in the eastern sectors are on the far side of the creekbed on a bluff and behind trees, so we assume that the channeling that increases concentrations in the creekbed also decreases concentrations at these monitors correspondingly. Thus there is no close-monitor peak in the measurements.

None of these conditions is included in the meteorological data for the site, which were taken from a tower not subject to those local, small-scale effects.

It is clear from Figure 5-12 that the site and monitors have some characteristics that are not well handled by the sector-average Gaussian plume model and that can explain the outliers in the estimated source term. If we (semi-arbitrarily) eliminate those estimates produced by conditions we know are outside the capabilities of the model -- the two highest values from the closest monitors (WSW and W) and three extreme low values from monitors in the creekbed (S, NNW, and SSE) -- we obtain an estimated source term of about 1500 Ci/yr. The uncertainty of this estimate is equivalent to that of the Gaussian dispersion model, about a factor of 2 plus the uncertainty of the estimated background radon concentration.

Figure 5-9. Location of site radon monitors.

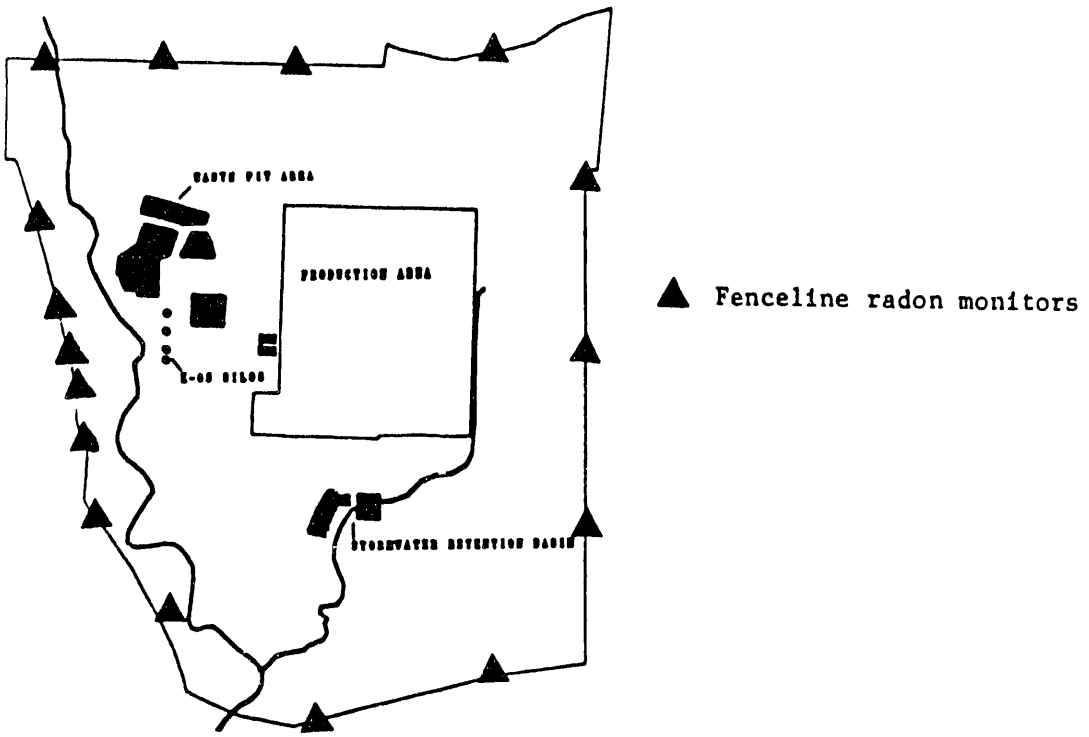


Figure 5-10. Source term coefficient.

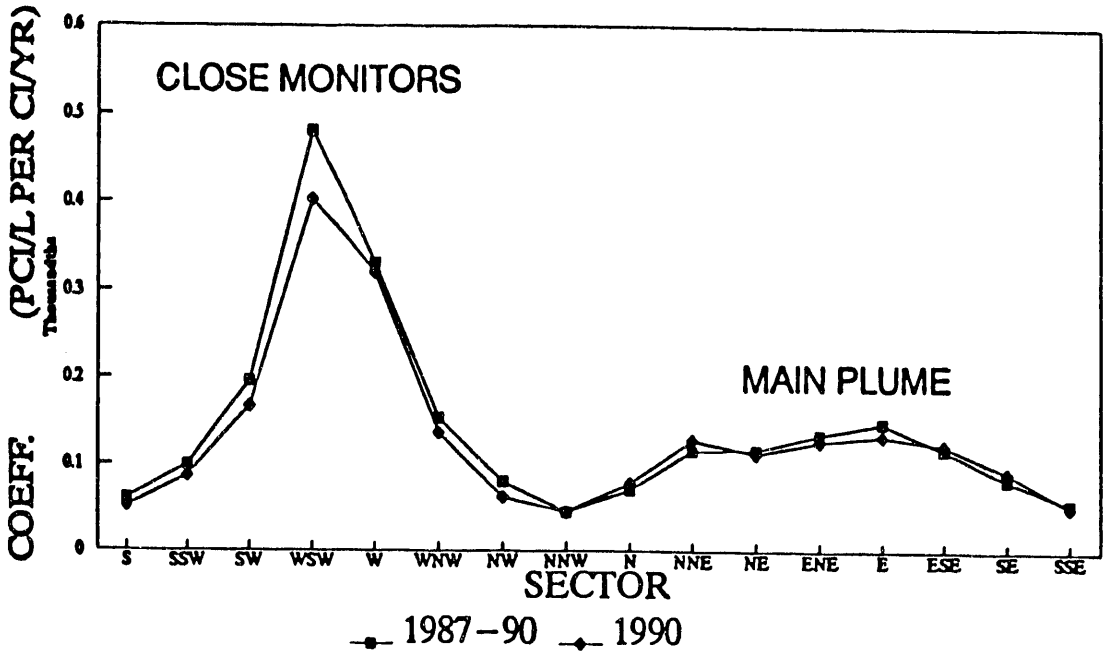


Figure 5-11. Predicted source term from data at each monitor.

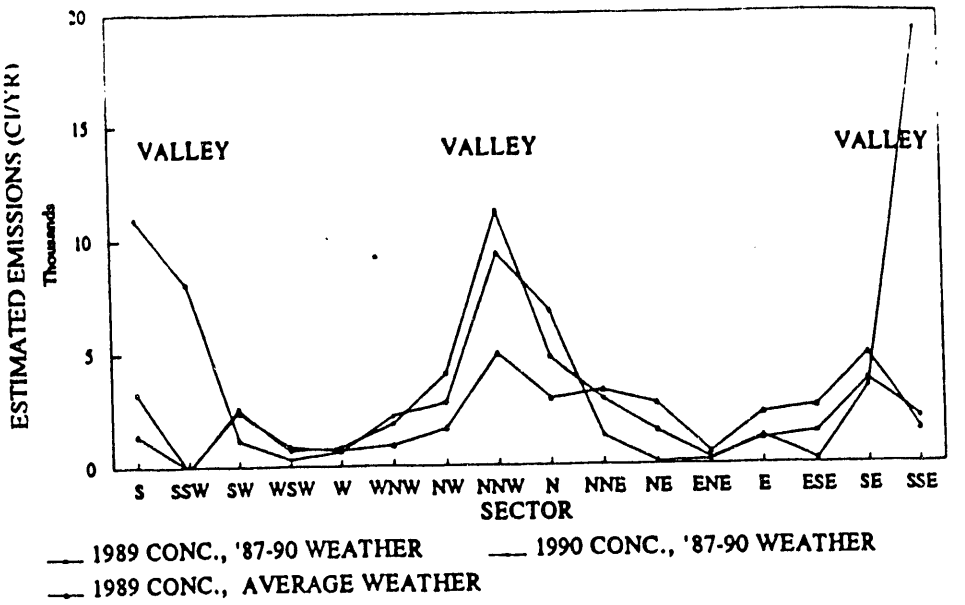
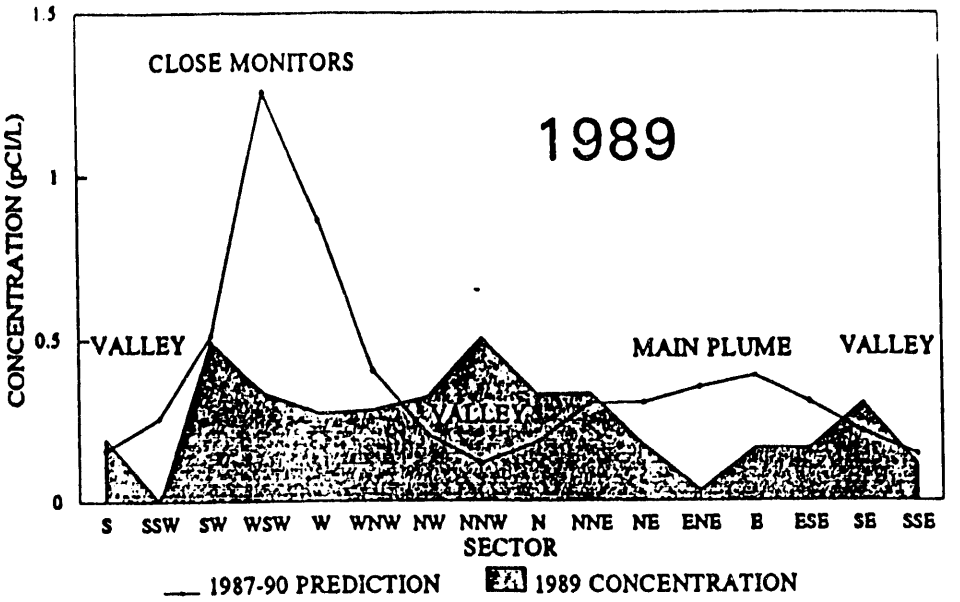
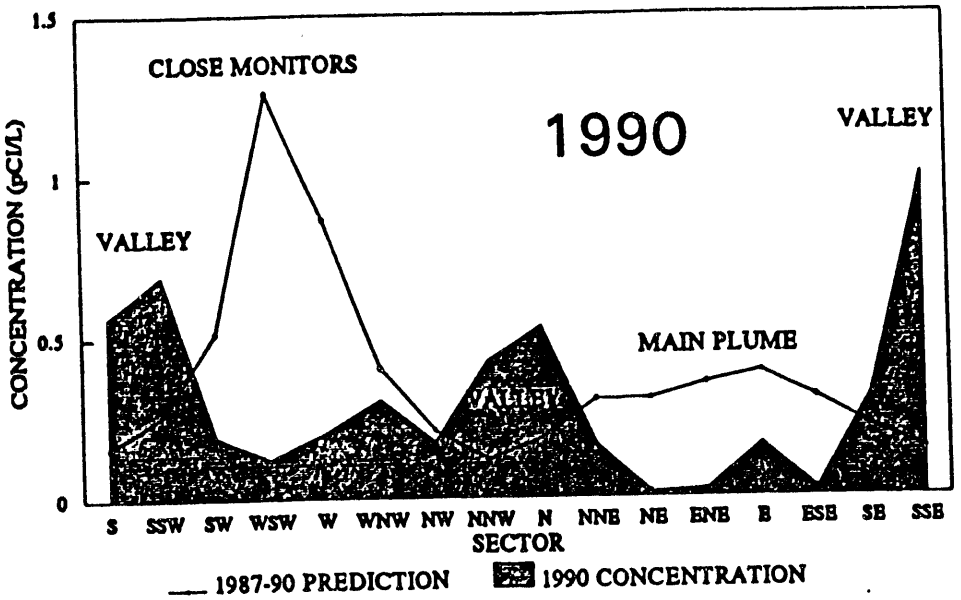


Figure 5-12. Measured and predicted radon concentrations.





5.5.2.3 Radon Production Rate

The rate of production of radon within the silos is not known. It must be estimated from measurements of radon concentrations in the head spaces of the silos and the estimated loss rates.

At equilibrium, the rate of production of radon must equal the sum of the losses from decay and from escape to the outside. Each silo contains about 33 Ci (1.2×10^3 GBq) of radon gas (BNI, 1990). At a half-life of 3.82 days, the decay rate of this radon must be 6.0 Ci/day (220 GBq/day). From our estimate of the source term, the average loss rate from each silo to the outside (1150 Ci/yr from both) must be 1.6 Ci/day (59 GBq/day). The total production rate at equilibrium was then estimated to be 7.6 Ci/day (281 GBq/day) per silo. The production rate for the minimum loss rate predicted from the radon monitors (575 Ci/year) is 6.8 Ci/day, and the production rate for the maximum loss rate (4024 Ci/year) is 11.5 Ci/day.

The estimated physical upper limit on the source term that corresponds with a total production rate of 7.6 Ci/day (281 GBq/day) is 5550 Ci/yr (2.1×10^5 GBq/yr). Note that this estimate is slightly circular, in that it includes our estimate of the source term obtained from a different calculation.

5.5.2.4 Routine Concentrations On and Offsite

Fenceline Residents - Monitoring Data

Because the gaussian plume model does not predict radon concentrations accurately along the FEMP boundary (see Section 5.5.2.2), monitoring data collected at three fenceline residences were used to describe the concentration of radon and radon progeny at houses along the FEMP western boundary. Figure 5-13 shows the location of these monitoring stations, and Table 5-6 gives measurements made at these stations in 1988 and 1989. The net increase in radon associated with the silos was estimated by subtracting the average outdoor background concentration in the area (0.5 pCi/L (18.5 Bq/m^3), section 5.5.1) from each measured value. The average of the net radon concentration at the three residential monitors is 0.45 pCi/L (16.7 Bq/m^3), and the standard deviation is 0.17.

Table 5-7 gives the calculated concentration of radon and radon progeny at these locations in units of Working Levels (WL). The equilibrium factor used to calculate concentrations in units of working levels for these stations was 0.5 indoors and 0.22, 0.26 and 0.33 outdoors for RES-1, RES-2 and RES-3, respectively (from Figure 5-6). These six pairs of concentrations (indoor and outdoor) were used to represent the distribution of concentrations to which fenceline residents are exposed.

Figure 5-13. Locations of FEMP residential monitors.

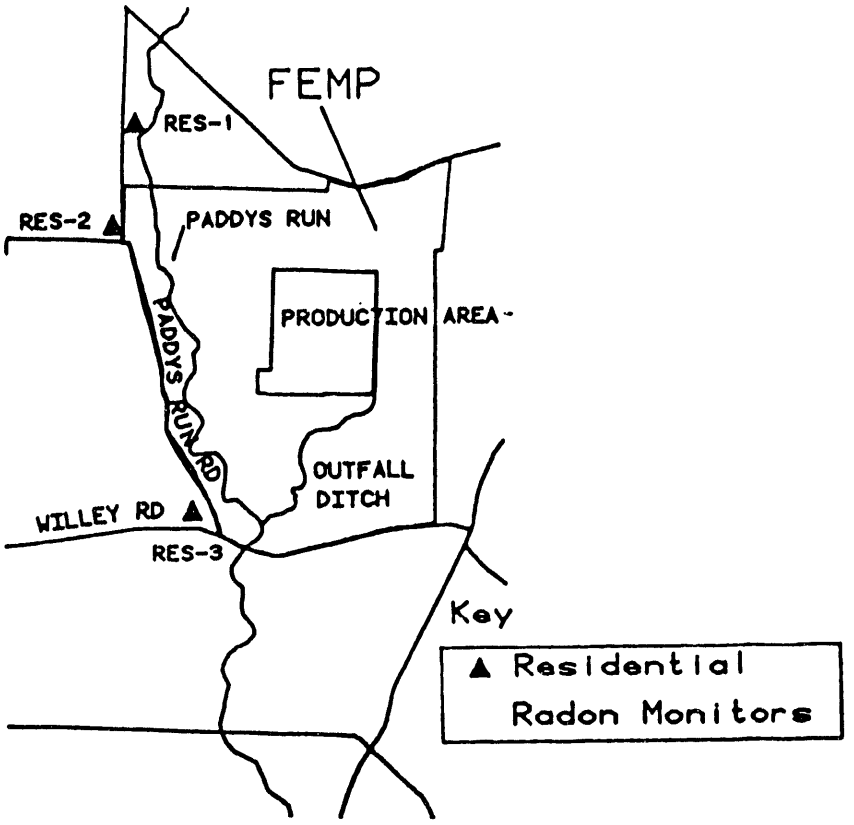


Table 5-6. Measured radon concentrations at three residential monitors (pCi/L, (Bq/m^3)).*

	1988 measured	net**	1989 measured	net**
RES-1	1.3 (48.1)	0.8 (29.6)	0.8 (29.6)	0.3 (11.1)
RES-2	0.9 (33.3)	0.4 (14.8)	0.9 (33.3)	0.4 (14.8)
RES-3	1.0 (37.0)	0.5 (18.5)	0.8 (29.6)	0.3 (11.1)

*data from USDOE (1990c)

**net radon increase associated with the silos; subtract background (0.5 pCi/L, 18.56 Bq/m^3) from the measured value.

Table 5-7. Calculated radon concentrations at three residential monitors (WL).

	Equilibrium Factor		Concentrations in Working Levels			
			1988		1989	
	OUT	IN	OUT	IN	OUT	IN
RES-1	0.22	0.5	1.7E-3	4.0E-3	6.5E-4	1.5E-3
RES-2	0.26	0.5	1.0E-3	2.0E-3	1.0E-3	2.0E-3
RES-3	0.33	0.5	1.6E-3	2.5E-3	9.8E-3	1.5E-3

Other Receptors - Gaussian Plume Model

The sector-average Gaussian plume model was applied to an assumed radon emission rate of 1150 Ci/yr (36.5×10^6 pCi/s, 4.3×10^4 GBq/yr) from the silos, using weather conditions measured at the 10 meter level of the site meteorological tower for the period 1987 through 1990, (M. Phillips, personal communication, August 1991).

It was assumed that the indoor radon concentration associated with the silos was the same as the predicted outdoor concentration, with an equilibrium factor of 0.5 applied to calculate concentration in units of working level (WL).

Figure 5-14 shows predicted concentrations of radon and radon progeny near the source. The material released from the silos is primarily the pure radon gas. This being the case, it requires some time for the decay products to appear. Peak concentration expressed as working levels (WL) is a function of the interaction between the rate of buildup and the rate of dispersion. Under average weather conditions, concentration of radon progeny is highest at about 500 m from the source. The maximum concentration predicted at 500 m was used to represent the concentration of radon to which indoor and outdoor onsite workers were exposed.

Table 5-8 summarizes the models and parameters used in predicting concentrations of radon and radon progeny resulting from the routine release of radon from the K-65 silos. Predicted concentration distributions at individual receptors are given in Table 5-9. Figure 5-15 shows the average predicted concentration of radon and radon progeny indoors and outdoors within 5 miles of the FEMP in units of working levels.

Figure 5-14. Predicted radon concentrations near the source.

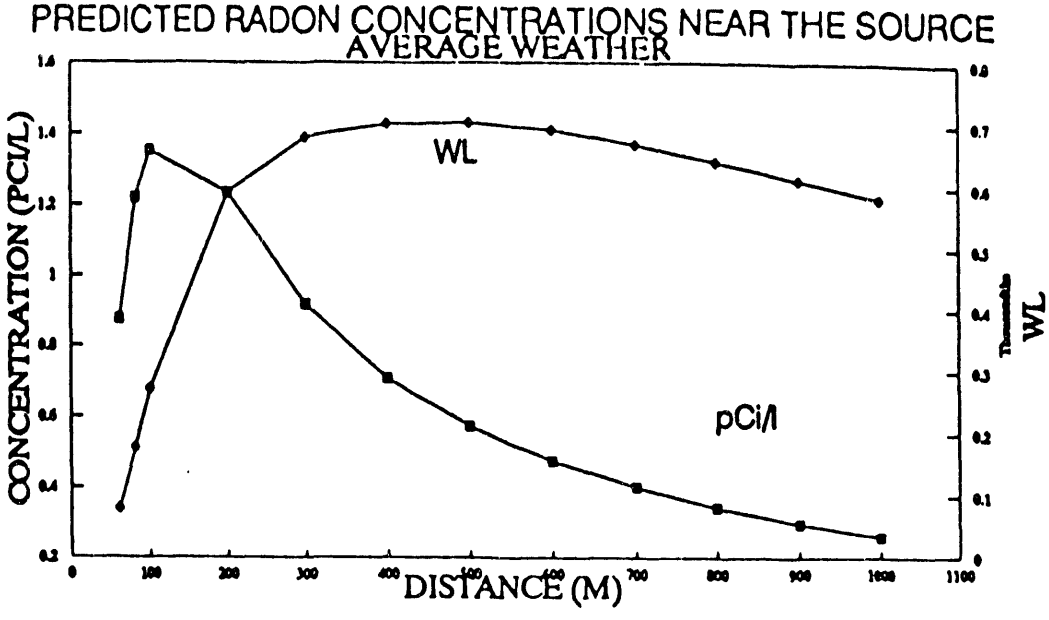


Table 5-8. Routine release scenario - models and parameters for predicting concentrations.

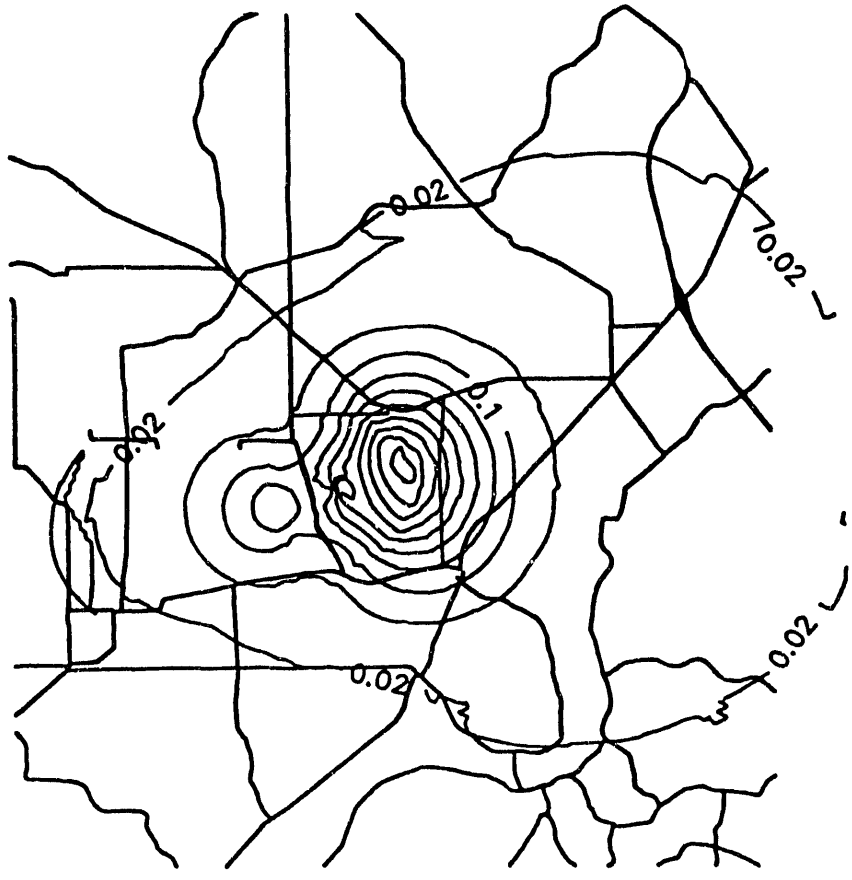
	OUTDOOR	INDOOR
radon concentration	predicted using gaussian plume model	predicted using gaussian plume model
source term	1150 Ci/year (4.3×10^4 GBq/yr)	1150 Ci/year (4.3×10^4 GBq/yr)
equilibrium factor	varies with time since release	0.5

Table 5-9. Routine release scenario -- predicted concentrations at individual receptors.

Receptor	Predicted Concentration (WL) OUTDOOR	INDOOR
Worker (500 m)*	8.4E-04	3.3E-03
Fenceline residents**	Table 5-7	Table 5-7
Within 1 mile	Figure 5-15	Figure 5-15
Within 5 mile	Figure 5-15	Figure 5-15
Hamilton	4.1E-05	3.0E-05
Cincinnati***	1.8E-05	1.32E-05

*maximum concentration at 500 m.
**measurements at three monitored houses.
***average predicted concentration for three polar grid nodes.

Figure 5-15. Estimated radon concentrations from routine emissions of 1150 Ci/yr (4.3×10^4 GBq/yr), outdoor concentrations in units of pCi/L (range is 0.003 to 0.4 pCi/L, contours are from 0.02 to 0.38 pCi/L, interval shown is 0.04 pCi/L).



5.5.2.5 Accidental Exposures

The accident analyzed is a collapse of the top of one silo with instantaneous release of the contents of the head space plus the entire radon production of the silo contents for seven days before they are again contained (BNI, 1990). This is a 33 Ci (1.2×10^3 GBq) puff plus a 53.2 Ci (2×10^3 GBq) leak over seven days (i.e. 7.6 Ci/day).

We assume that the probability of an accident is independent of meteorology, which is reasonable if we do not believe that a powerful storm is likely to cause the accident. This assumption is conservative, in that dispersion rates during a storm are much higher than those under more normal conditions, so the resulting exposures to the nearby population would be much lower.

Under this assumption, the probability distribution of possible outcomes of an instantaneous release is equal to the distribution of exposures from emission of 33 Ci (1.2×10^3 GBq) under the joint distribution of meteorological conditions over a year.

Exposures from an accidental release of seven days duration are more difficult to assess, because this is an inconvenient duration for quantifying the effects of weather. Lacking a statistical description of real weather patterns for one-week intervals, the best we can do is set bounds on the possible outcomes.

At one extreme, the weather could remain constant for the entire week in the conditions at the time of the accident. This produces a distribution of estimated exposures with much higher variability than would normally occur over a week. At the other extreme, the weather could go through the entire annual cycle of stability classes and wind speeds during the week -- annual average weather. This produces a distribution of estimates with much lower variability that would normally occur. Reality must be somewhere in between.

We make no attempt to include the uncertainty of the accident source term in this analysis. We have no credible way of estimating it. For purposes of this assessment, it was assumed that neither the workers nor the surrounding population were evacuated.

Concentrations in working levels (WL) are not predicted for the accidental release because the concentrations will vary over time during the event, and because the values of interest are individual and population exposure WLM for the event.

Table 5-10 summarizes the models and parameters used to predict exposures associated with the accidental release of radon. The distributions of the resulting accidental exposures are shown in Table 5-11. The expected population exposure within 5 miles is about 0.096 Person-WLM for both weather conditions.

Table 5-10. Accidental release scenario - models and parameters for predicting concentrations and calculating exposure.

	OUTDOOR	INDOOR
exposure in (WLM)	predicted using gaussian plume model	predicted using gaussian plume model
source term	initial release 33 Ci (1.2×10^3 GBq) 7 day continuous release of 7.6 Ci/day (GBq)	initial release 33 Ci (1.2×10^3 GBq) 7 day continuous release of 7.6 Ci/day (GBq)
equilibrium factor	varies with time since release	0.5

Table 5-11. Estimated range of accidental exposures to 1 mile.[a]

Weather	Exposure (P-WLM) Mean	Median	95% Range
Constant[b]	0.096	0.0014	0.0-0.62
Annual Average[c]	0.096	0.057	0.050-0.34

a. Initial emission of 33 Ci plus 7.6 Ci/day for one week. No evacuation.
b. Weather in the following week unchanged from the time of the accident.
c. Weather during the following week goes through the full annual distribution of stability classes and wind speeds over all sectors.

5.6 Exposures to Radon and Radon Progeny

Exposure estimates were developed for the receptors described in section 5.4, using predicted concentrations of radon and radon progeny originating in the K-65 silos, and measured concentrations at fenceline monitors.

For background levels and for levels associated with the routine release scenario, exposures were calculated in units of WLM/Lifetime for individual exposures and person-WLM/Lifetime for population exposures. For the accidental release scenario, exposures were calculated for the event in units of WLM.

Most estimates of individual exposure are presented as probability distributions rather than single point estimates. These exposure distributions were calculated using Latin Hypercube sampling of the parameter distributions included in the formulae described below.

Some of the estimates described below were calculated using single values, producing a deterministic estimate of exposure. The individual routine exposure to workers, and the individual exposures to workers and to offsite populations associated with the accidental event were calculated deterministically, using the

average values of the parameters in the formulae given below. Population exposures were calculated as single value estimates, using mean values of individual exposure.

5.6.1 Calculation of Exposure

5.6.1.1 Routine Release Scenario and Natural Background Exposures

The following formulae were used to calculate individual and population lifetime exposures for the routine release scenario and for natural background levels of radon.

Individual Lifetime Outdoor Exposure

OEXP (WLM/Lifetime) = OC (WL) x 51.5 (WM/year) x YE x OF

where:
OEXP = outdoor exposure (WLM/Lifetime)
OC = outdoor concentration (WL)
YE = years exposed (years)
OF = Fraction of year exposed outdoors

Individual Lifetime Indoor Exposure

IEXP (WLM/Lifetime) = IC (WL) x 51.5 (WM/year) x YE x IF

where:
IEXP = indoor exposure (WLM/Lifetime)
IC = indoor concentration (WL)
YE = years exposed (years)
IF = Fraction of year exposed indoors

Total Individual Lifetime Exposure

Total individual lifetime exposures (TEXP) were calculated for each scenario and receptor by summing the estimated lifetime indoor exposure (IEXP) and the estimated lifetime outdoor exposure (OEXP).

TEXP (WLM/Lifetime) = OEXP + IEXP

Population (Collective) Lifetime Exposure

Population lifetime exposures in person-WLM (indoor, outdoor, total) were calculated by multiplying the mean lifetime exposure estimate for each grid cell by the estimated population in the cell. These values were summed to estimate the total population exposure for a receptor.

5.6.1.2 Accidental Release Scenario

For the accidental release scenario, individual and population exposures were calculated by the model described in Section 5.5.2.5.

5.6.2 Natural Background -- Input Parameters and Exposure Estimates

Exposure Parameters

The number of years exposed was 70 to provide an estimate of lifetime exposure to natural background levels. The fraction of time spent indoors was described by a triangular distribution, with a minimum at 0.4 (9.6 hours), a maximum at 1 (24 hours) and an expected value of 0.924 (22.2 hours).

Exposure Estimates

Lifetime individual and population exposures were calculated as described in section 5.6.1.1. Table 5-12 presents the estimates of individual lifetime exposure from natural background concentrations. All identified receptors, including the onsite workers, were assumed to be exposed to the same distribution of indoor and outdoor natural background radon levels. The mean exposure to background concentrations of radon and radon progeny was 55.0 WLM, the median exposure was 36.1 WLM, and 95% of individuals are expected to be exposed to less than 169 WLM.

The natural background population exposures for all identified population receptors (Hamilton, Cincinnati, the population within 1 mile and the population within 5 miles) were calculated by multiplying the mean individual exposure (36.1 WLM, Table 5-12) by the total population of each of the receptors (Table 5-13). Table 5-13 gives the estimated population exposures associated with background concentrations of radon and radon progeny.

Table 5-12. Lifetime individual exposure estimates - background concentrations (WLM).

MEAN	MEDIAN	95% CL
55.0	36.1	169.0

Table 5-13. Lifetime population exposure estimates - background concentrations.

Receptor	Population	Population Exposure (person-WLM/Lifetime)
Within 1-mile	175	9.6E+3
Within 5-mile	14,729	8.1E+5
Hamilton	61,368	3.4E+6
Cincinnati	441,732	2.4E+7

5.6.3 Routine Release Scenario -- Input Parameters and Exposure Estimates

Individual and population exposures for the identified receptors were calculated as described in section 5.6.1. The exposure parameters used in the analysis are summarized in Table 5-14. Table 5-15 presents the estimates of lifetime exposure for individual receptors. Individual lifetime exposures are highest for indoor workers (mean: 2.2 WLM) and for fenceline residents (median: 8.9×10^{-1} WLM). The exposure for the individual receptor within 1 mile and within 5 mile is the exposure distribution for an individual living anywhere within 1 or 5 miles of the silos.

Exposure distributions can be calculated for individuals living at all 80 nodes of the polar grid used in the analysis, but this volume of information would be difficult to present and interpret. Average exposures (rather than exposure distributions) were calculated for each node of the polar grid to demonstrate the changes in exposure that occur over space. These average exposures were based on the concentration of radon predicted at each node, a residence time of 7.8 years and a fraction of time indoors of 0.924. These estimates of average individual exposure are shown in Figure 5-16.

Population exposures for the Hamilton and Cincinnati receptor populations were calculated by multiplying the mean individual exposure for the location by the population of the receptor. The population exposures for the within 1-mile and within 5-mile receptor populations were calculated in a different way, because the individual exposure distribution calculated for individuals in this population does not apply to the population of each polar grid node. Total population exposures for these two receptor populations were calculated by multiplying the average individual exposure at each grid node (Figure 5-16) by the population in that node (Figure 5-7). The resulting population exposures (Person-WLM/Lifetime) at each polar grid node are presented in Figure 5-17. The total population exposures for the within 1-mile and within 5-mile receptors were calculated by summing the population exposures for all grids within the appropriate distance. Table 5-16 summarizes the population exposure estimates resulting from exposure to radon released from the K-65 silos.

Table 5-14. Routine release scenario -- parameters for calculating exposure.

	OUTDOOR	INDOOR
YEARS EXPOSED		
worker	50	50
fenceline residents	Avg. 7.8 years, Figure 5-8	Avg. 7.8 years, Figure 5-8
population 1-5 miles	Avg. 7.8 years, Figure 5-8	Avg. 7.8 years, Figure 5-8
Hamilton, Cincinnati	70 years	70 years
FRACTION OF YEAR EXPOSED		
indoor worker	0	0.257
outdoor worker	0.228	0.0285
all other receptors	(1 - indoor fraction)	expected: 0.92 (triangular 0.4 - 1)

Table 5-15. Routine release scenario - lifetime individual exposure estimates.

Receptor	Exposure (WLM/Lifetime) mean	median	95% CL
Indoor Worker	2.2	--	--
Outdoor Worker	7.3E-1	--	--
Fenceline residents	1.4	8.9E-1	4.6
Within 1-mile	5.7E-1	3.6E-1	1.8
Within 5-mile	1.7E-1	1.1E-1	5.5E-1
Hamilton	2.3E-2	1.4E-2	7.4E-2
Cincinnati	1.0E-2	6.3E-3	3.3E-2

Figure 5-16. Average individual exposures within 5 miles of the K-65 silos (WLM) (data range is 0.007 - 0.76, contour range is 0.05 - 0.7, contour interval is 0.05).

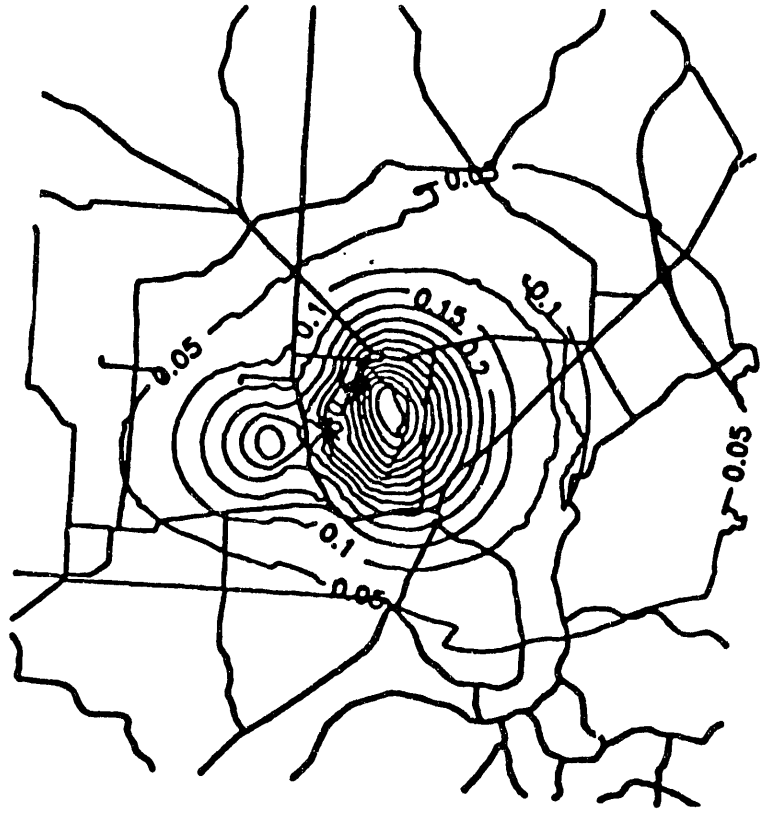
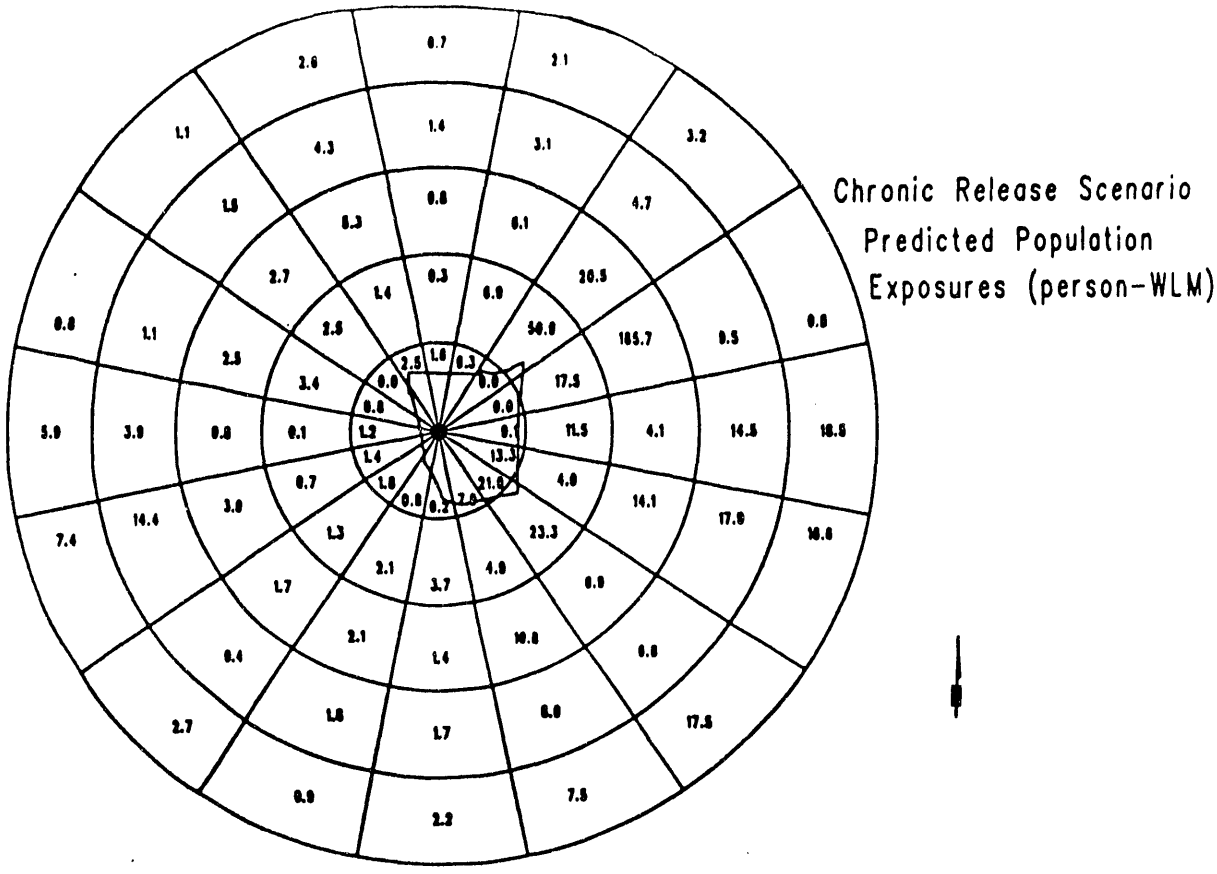


Table 5-16 Routine release scenario -- lifetime population exposure estimates.

Receptor	population	population exposure (person-WLM/lifetime)
Within 1-mile	175	67.0
Within 5-mile	14,729	657.5
Hamilton	61,368	1393.1
Cincinnati	441,732	4461.5

Figure 5-17. Average population exposures within 5 miles of the K-65 silos (outline at center of the polar grid is the FEMP site boundary).



5.6.4 Accidental Release Scenario -- Input Parameters and Exposure Estimates

Individual and population exposures were calculated as described in section 5.6.1. An analysis was performed for constant weather and annual average weather conditions (Section 5.5.2.5). The mean predicted exposure is the same for both weather conditions. The exposure estimates presented below are based on average annual weather conditions.

Individual exposures were estimated for the most highly exposed workers (indoor and outdoor workers), the most highly exposed fenceline resident, and individuals in Hamilton and Cincinnati. Population exposures were calculated for the population living within 1 mile and 5 miles of the silos, and for the populations of Hamilton and Cincinnati. The exposure period used in the estimation of exposure for the general population was 7 days, with 92.4% of time spent indoors. Indoor workers were assumed to be exposed to radon and radon progeny released in an accident for 9 hours per day (all indoors) for 7 days. Outdoor workers were assumed exposed for 7 days, 1 hour per day indoors and 8 hours per day outdoors. Table 5-17 presents the estimates of exposure for individual receptors. Table 5-18 summarizes the population exposure estimates for the accidental release scenario.

Table 5-17. Accidental release -- individual exposure estimates.

Receptor	Individual Exposure (WLM)
Indoor Worker*	4.7E-3
Outdoor Worker*	1.6E-3
Fenceline Resident*	3.1E-3
Hamilton	1.2E-4
Cincinnati	5.3E-5

* worker and resident with maximum predicted exposure

Table 5-18. Population exposure estimates -- accidental release.

Receptor	Population	Population Exposure Estimates (Person-WLM)
Within 1-mile	175	0.6
Within 5-mile	14,729	6.3
Hamilton	61,368	7.2
Cincinnati	441,732	23.3

5.7 Risk Characterization

5.7.1 Calculation of Risk

The individual lifetime risk associated with exposure to radon and radon progeny was calculated as:

$$ILR = TEXP \times RF$$

where:
ILR = individual lifetime risk
TEXP = total lifetime exposure (WLM/lifetime)
RF = risk factor (2.0×10^{-4} deaths /WLM)

When total lifetime exposures were calculated as distributions, the individual lifetime risk calculated using the above equation was also described as a distribution.

Some of the individual lifetime risk estimates described below were calculated using single values, resulting in a deterministic estimate of individual

lifetime risk. The individual routine risk to a worker, and the individual lifetime risks associated with the accidental event were calculated deterministically, using the average values of the parameters in the formulae given below. Population risks were calculated as single value estimates, using mean values of individual risk.

The total population risk for a given population receptor (predicted number of excess cancers) was calculated as:

$$PR = PE \times RF$$

where:
PR = total population risk (cancers)
PE = population exposure (person-WLM/lifetime)
RF = risk factor (2.0×10^{-4} deaths /person-WLM)

5.7.2 Natural Background -- Risk Estimates

Lifetime individual and population risks associated with exposure to natural background concentrations of radon and radon progeny were calculated as described in section 5.7.1. Table 5-19 and Figure 5-18 summarize the individual lifetime risks for individual receptors. Table 5-20 and Figure 5-19 summarize the population risks associated with exposure to natural background levels of radon and radon progeny.

Individual lifetime risks associated with background concentrations of radon and radon progeny have a mean value of 1.2×10^{-2}, and a median value of 7.3×10^{-3}. 95% of individuals are expected to have an individual lifetime risk less than 3.7×10^{-2}.

The number of cancers over a lifetime associated with natural background concentrations of radon and radon progeny is highest for Cincinnati (5168) because it has the largest population.

Table 5-19. Lifetime individual risks - background concentrations.

MEAN	MEDIAN	95% CL
1.2E-2	7.3E-3	3.7E-2

Figure 5-18. Distribution of individual lifetime risks associated with exposure to background concentrations of radon and radon progeny.

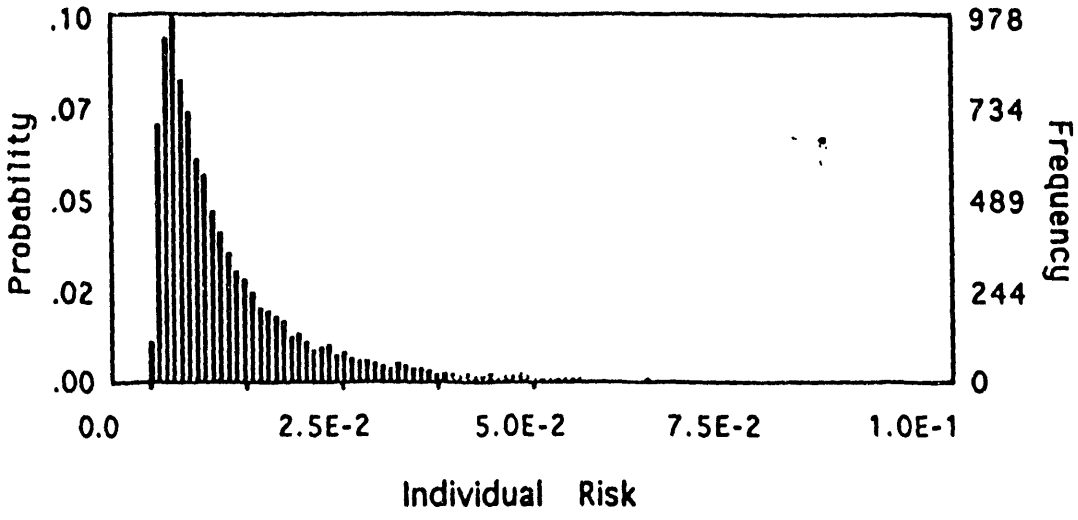
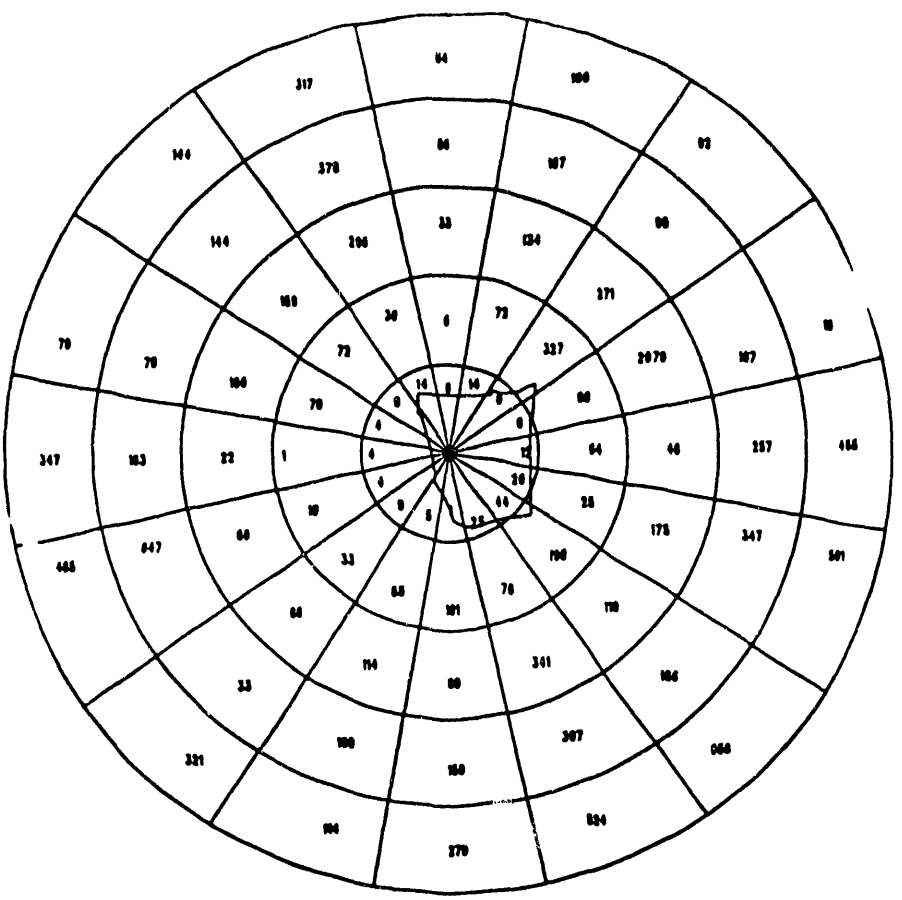


Table 5-20. Population (collective) risks due to natural background radon.

Receptor	Lifetime Population Risks (# of cancers)
Within 1-mile	2.0
Within 5-mile	172.3
Hamilton	718.0
Cincinnati	5168.3

Figure 5-19. Population (collective) risks due to natural background radon (predicted # of cancers x 10^{-2}) within 5 miles of the silos (outline at center of polar grid is FEMP site boundary.



5.7.3 Routine Release -- Risk Estimates

Lifetime individual and population risks associated with exposure to predicted concentrations of radon and radon progeny resulting from the routine release of radon from the K-65 silos were calculated as described in section 5.7.1. Table 5-21 summarizes the individual lifetime risks estimated for the routine release scenario. Figure 5-20 presents the distribution of lifetime individual risks for the most highly exposed fenceline resident.

Average individual lifetime risks were calculated for each node of the 5 mile polar grid based on the average individual exposures shown in Figure 5-16. These average individual risks are shown in Figure 5-21. Table 5-22 and Figure 5-22 summarize the population risks associated with lifetime exposure to radon and radon progeny originating in the K-65 silos.

Individual lifetime risks are highest for indoor workers (mean: 4.3×10^{-4}) and for fenceline residents (median: 1.8×10^{-4}). Median risks greater than 1×10^{-4} were predicted only for fenceline residents. The 95% CL is greater than 1×10^{-4} for fenceline residents and for the individual living within 1 mile and 5 miles of the

silos. Very few cancers (less than one in a lifetime) were predicted for the populations within 1-mile, within 5-miles, in Hamilton and Cincinnati (Table 5-22).

Table 5-21. Routine release scenario --lifetime individual risk estimates.

Receptor	Individual Lifetime Risk		
	mean	median	95% CL
Indoor Worker	4.3E-4	--	--
Outdoor Worker	1.5E-4	--	--
Fenceline resident	3.0E-4	1.8E-4	1.0E-3
Within 1-mile	1.2E-4	7.0E-5	4.2E-4
Within 5-mile	3.6E-5	2.2E-5	1.2E-4
Hamilton	4.9E-6	2.8E-6	1.6E-5
Cincinnati	2.1E-6	1.3E-6	7.3E-6

Figure 5-20. Routine release scenario -- distribution of individual lifetime risks for most highly exposed fenceline resident. (Note: the risks estimated for the routine release scenario are much smaller than those associated with background exposures; when comparing Figure 5-20 to Figure 5-18 please note that the units are different).

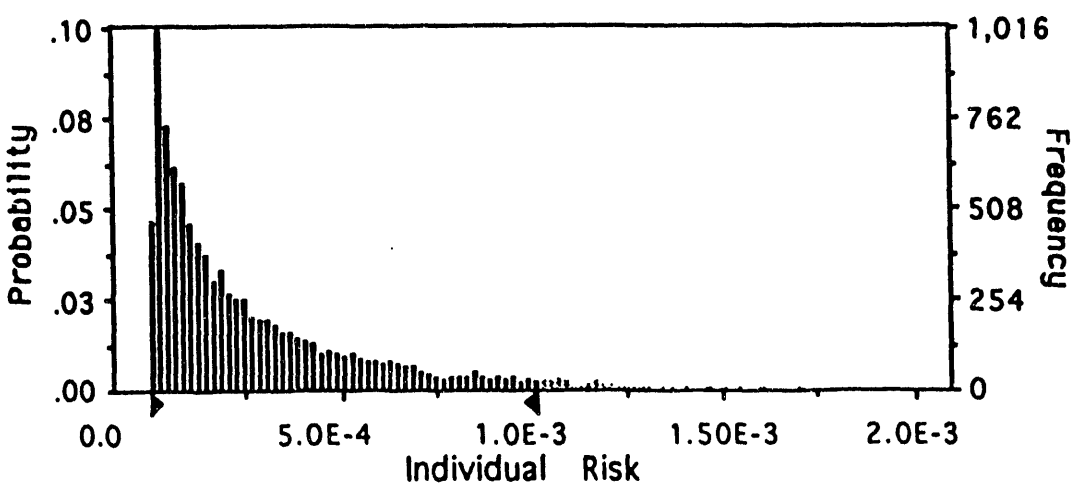


Figure 5-21. Routine release scenario -- average individual lifetime risks (x 10^{-3}) within 5 miles of the silos.

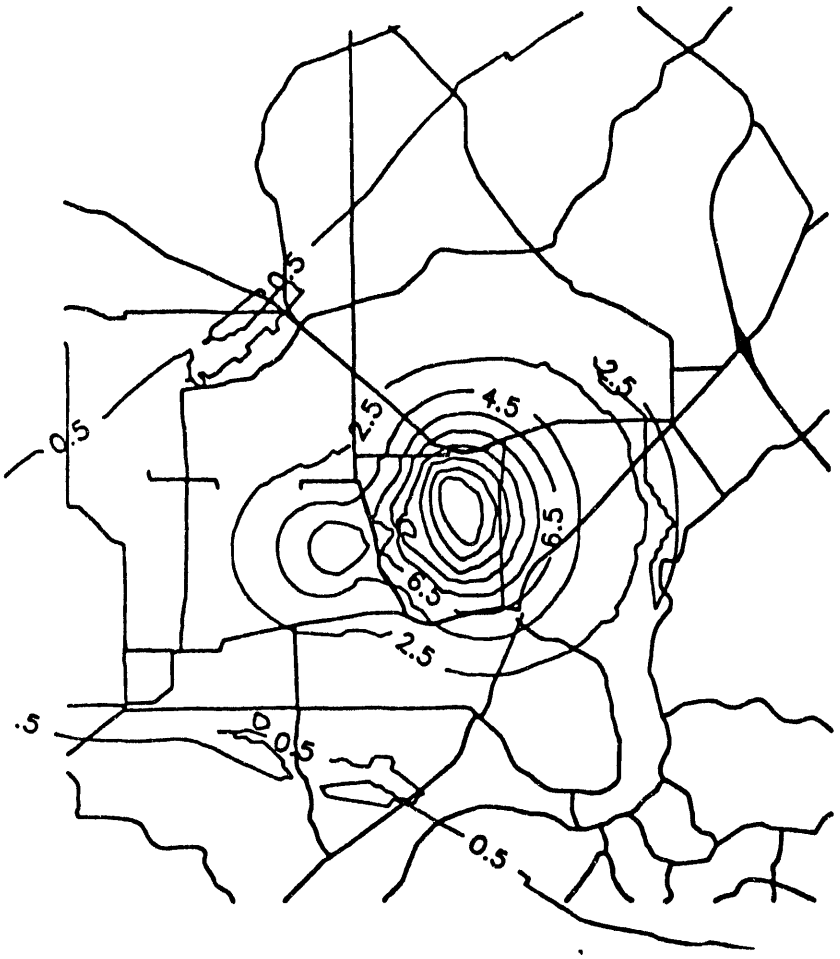
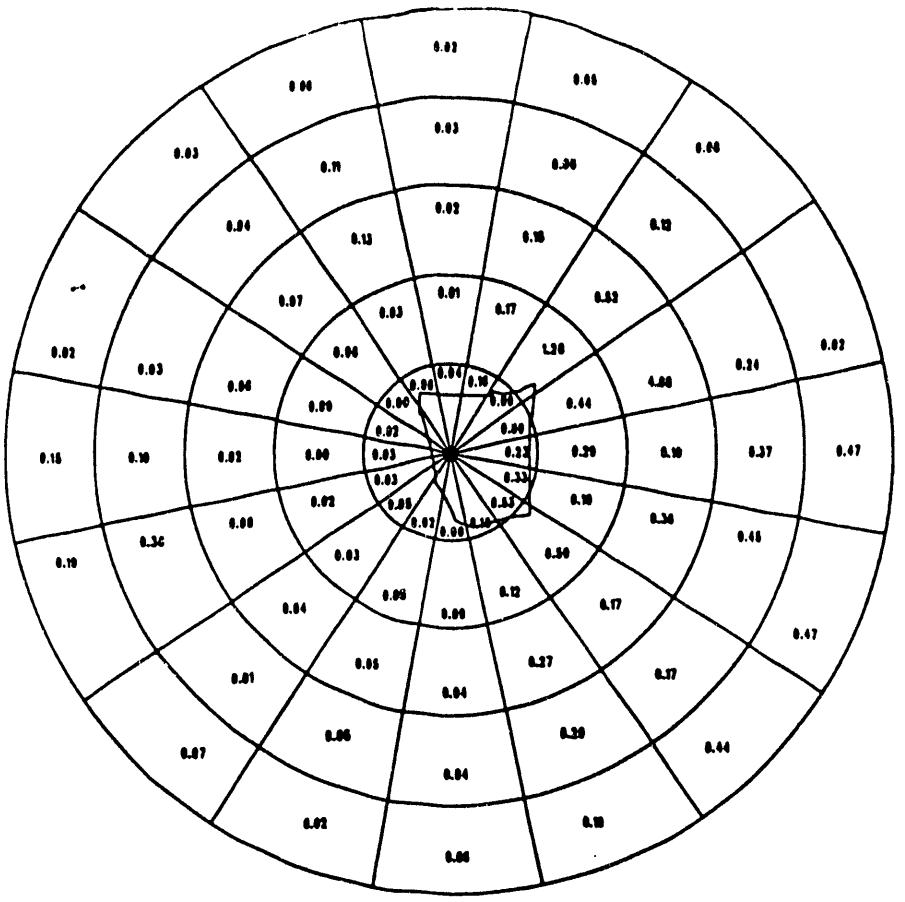


Table 5-22. Routine release scenario -- lifetime population risks.

Receptor	Lifetime Population Risks (# cancers)
Within 1-mile	0.01
Within 5-mile	0.13
Hamilton	0.30
Cincinnati	0.95

Figure 5-22. Population (collective) risks based on the routine release scenario (predicted # of cancers x 10^{-2}) within 5 miles of the silos (outline at center of polar grid is FEMP site boundary.



5.7.4 Accidental Release - Risk Estimates

Individual and population risks associated with exposure to predicted concentrations of radon and radon progeny resulting from an accidental release of radon from one of the K-65 silos were calculated as described in section 5.7.1. Table 5-23 summarizes the individual risks for specific receptors and Table 5-24 summarizes the population risks associated with exposure to radon and radon progeny due to an accidental release.

Individual risks associated with the accidental release of radon from the silos are highest for indoor workers (1.2×10^{-6}) and fenceline residents (7.9×10^{-7}). No individual risks greater than 1×10^{-6} were predicted. Population risks are highest in Cincinnati (0.005), but no population receptor had population risks greater than 1.0.

Table 5-23. Accidental release -- individual risk estimates.

Receptor	Individual Risk
Indoor Worker*	9.4E-7
Outdoor Worker*	3.2E-7
Fenceline Resident*	6.3E-7
Hamilton	2.4E-8
Cincinnati	1.1E-8

* most highly exposed worker and fenceline resident

Table 5-24. Accidental release - population risks.

Receptor	Population Risks - # cancers
Within 1-mile	0.0001
Within 5-mile	0.0013
Hamilton	0.0014
Cincinnati	0.0047

5.8 Assumptions and Uncertainties

This assessment included a number of assumptions and associated uncertainties which should be considered in interpreting the results of the analysis. The major assumptions incorporated into the assessment, and the associated uncertainties are described below. As noted, some of the uncertainties described have been incorporated into the assessment through the use of Monte Carlo techniques. Other uncertainties were treated in a separate uncertainty analysis (i.e. uncertainty in source term), and still others are discussed qualitatively.

5.8.1 Indoor and Outdoor Background Concentrations

The background concentration outdoors near the FEMP was described by the average concentration measured at 4 stations more than 25 km from the facility. This assessment assumed that outdoor background concentrations of radon from 1-10 miles from the site could all be represented by this value (0.5 +/- 0.18 pCi/L). In fact, natural background concentrations vary over space.

The indoor background concentrations near the FEMP site were described by a distribution of values derived from a study based on monitors sent to homes responding to a mail order survey (Cohen and Shah, 1991). Bias reduction techniques were employed in this survey, but the authors believe that the reported

values may be biased upward by as much as 20%. Sources of bias in this study include: 1) low income families are underrepresented; 2) high rise apartments were not included; 3) people concerned with environmental issues are overrepresented and a higher percentage of houses have been weatherized than the national average; 4) urban areas are underrepresented; 4) cigarette smokers are underrepresented; 5) rented houses are underrepresented and 6) a higher fraction of bedrooms in the sample were in basements than is usual (Cohen and Shah, 1991). Many of these variables are associated with reduced radon levels (Cohen and Shah, 1991). Many variables affect the indoor concentration of radon in homes, and the background concentration in homes near the FEMP is uncertain.

5.8.2 Prediction of Radon and Radon Progeny Concentrations

A sector-averaged gaussian plume model and site meteorological data were used to: 1) estimate the routine and accidental release source terms; and 2) predict the concentration of radon and radon progeny up to 10 miles from the silos for the routine and accidental release scenarios. Uncertainties associated with this analysis and its associated assumptions are discussed below.

The routine release source term estimate

The sector averaged Gaussian plume model and site meteorological data are not good representations of the micrometeorology of the FEMP site. The model does not accurately predict the spatial pattern of radon concentrations measured at the site boundary monitors. The uncertainty of the estimated source term is large because it was derived from source coefficients back calculated from measurements made at site boundary monitors. It was assumed that a least-squares regression analysis that minimizes the mean-square difference between the measured and the predicted concentrations over the 16 monitors would produce the best source term estimate. This yields an estimated source term of 1150 Ci/yr (4.3×10^4 GBq/yr). This approach should produce a reasonable estimate of the source term, even though we have little confidence in the individual estimates close to the silos. This calculated source term is similar to the value calculated by IT (1989) (for emissions before the addition of the foam cover) based on estimates of diffusion through the silo domes and free air exchange between the pore volume inside the domes and the outside atmosphere.

The source term back-calculated from the radon monitors is 1150 Ci/year (section 5.5.2.2). The minimum source term calculated from a single radon monitor was 575 Ci/year, and the maximum was 4025 Ci/year. An analysis was performed using the distribution of source terms resulting from the radon monitors to describe the possible range of source terms, and to predict radon and radon daughter concentrations within 1 mile of the silos. Table 5-25 compares the distribution of individual lifetime risks associated with this distribution of source terms to the risks predicted using the average source term of 1150 Ci/year. Including a range of possible source terms did not strongly influence the risk estimates -- the mean risk was increased by 30%, and the 95% Confidence Limit was increased by 33%.

Table 5-25. Uncertainty analysis for radon source term -- effect on individual lifetime risks for people living within 1-mile of the silos.

Source Term	Individual Lifetime Risk mean	median	95% CL
Average (1150 Ci/year)	1.2E-4	7.0E-5	4.2E-4
Distribution (575-4025 Ci/year)	1.6E-4	7.6E-5	5.7E-4

The accidental release source term estimate

The estimate of the amount of radon that would be lost during an accidental release from one of the silos is based on the amount of radon available in the head space, the production rate of radon in the silos, and the estimated loss rate. The amount of radon in the head space was previously estimated from gas samples (33 Ci, 1221 GBq, BNI, 1990). The production and loss rates depend on the calculation of the routine release source term described above and have the same uncertainties.

The source term for the accident based on the average routine source term estimate is 33 Ci instantaneously, with an additional 53.2 Ci spread over 7 days. The source term calculated for the minimum routine source term is 33 Ci + 47.6 Ci over seven days, and the maximum source term gives an estimate of 33 Ci + 80.6 Ci over 7 days.

The maximum potential source term is less than twice the average estimated source term. Estimated risks associated with the accidental release of radon from a single silo are small (6.3×10^{-7} at the fenceline) and even a doubling of the predicted concentration and risk would result in small estimates of risk (1.3×10^{-6}).

Treatment of meteorology during the accidental scenario

We assume that the probability of an accident is independent of meteorology, which is reasonable if we do not believe that a powerful storm is likely to cause the accident. This assumption is conservative, in that dispersion rates during a storm are much higher than those under more normal conditions, so the resulting exposures to the nearby population would be much lower. Under this assumption, the probability distribution of possible outcomes of an instantaneous release is equal to the distribution of exposures from emission of 33 Ci (1221 GBq) under the joint distribution of meteorological conditions over a year. Lacking a statistical description of real weather patterns for the remainder of the week of the accident, the best we can do is set bounds on the possible outcomes.

At one extreme, the weather could remain constant for the entire week in the conditions at the time of the accident. This produces a distribution of estimated exposures with much higher variability than would normally occur over a week. At the other extreme, the weather could go through the entire annual cycle of stability

classes and wind speeds during the week -- annual average weather. This produces a distribution of estimates with much lower variability that would normally occur. Reality must be somewhere in between.

5.8.3 Exposure Periods

Time at Home and Time Spent Indoors

This assessment assumes that people receive all of their radon exposure where they live. This is a conservative assumption (resulting in an overestimate of exposure and risk for radon associated with the silos) for people living within 5 miles of the silos, because most people will spend part of their day away from home where the radon concentration associated with the silos is smaller. This assumption was made because the analysis is for exposure to a spatially varying concentration of radon in a plume, and because it is not possible to account for the movements of all the people subject to exposure.

For calculation of routine exposure to radon this assessment assumed a distribution of average residence time for rural populations estimated from current residence time data (Israeli and Nelson, 1992). This distribution was derived from nationwide rural data, and may not be completely representative of the moving rates of the population near the FEMP. The application of this distribution assumes that once a household moves, it moves away from the impacted area -- this may not always be the case and could produce a small underestimate of the individual risks associated with the routine loss scenario within 5 miles of the FEMP.

A residence time of 70 years was assumed for the populations of Hamilton and Cincinnati because, for urban populations, moving could not be assumed to remove the population from the impacted area. This assumption will produce an overestimate of the individual risks associated with the routine loss scenario. However, the risks predicted for Hamilton and Cincinnati were small, (1.3×10^{-6} and 3.0×10^{-6} for Cincinnati and Hamilton, respectively) and this conservative assumption did not have an important effect on the assessment.

Time Spent Indoors

Because of the ingrowth and behavior of radon progeny, plus other factors, the exposure to radon and radon progeny in units of Working Levels is different indoors and outdoors. This assessment assumed a most probable time indoors of 92.4% The distribution used to describe time spent indoors for the probabilistic analysis was a triangular distribution, most probable value at 92.4%, minimum at 40% and maximum at 100%. The shape and bounds of the distribution are uncertain.

5.8.4 Risk Factor for Radon

This assessment assumed a risk factor distribution for radon and radon progeny described by a lognormal distribution, with a geometric mean of 2.0×10^{-4} per person-WLM and a geometric standard deviation of (Section 5.3.5). There is considerable controversy over the quantitative relationship between magnitude of exposure to alpha radiation from radon progeny and resulting risk of lung cancer (NRC, 1988). Lacking the necessary data, we must assume the simplest relationship which, at the current state of knowledge is a linear, nonthreshold function extrapolated from the high doses of miners to the much lower doses of the general

public. This extrapolation has exceedingly high uncertainty, and may overestimate the risk associated with exposure to radon and radon progeny. Some of the uncertainty in the risk factor was incorporated into the analysis by using a distribution of risk factors reported by a number of studies (section 5.3.5. ICRP, 1991a).

5.9 Summary - Risk Assessment for Radon Released from the K-65 Silos

Table 5-26 summarizes the lifetime individual risks predicted for background radon concentrations, the routine release scenario and the accidental release scenario. Median lifetime risks are given for the background and routine scenarios (except workers, value given is mean). Mean risk estimates are given for the accidental scenario.

Table 5-27 summarizes the population risks (predicted number of cancers) associated with the three scenarios. The number of cancers associated with background is much higher than those associated with either the routine or accidental release of radon from the FEMP silos.

Radon and radon progeny associated with the K-65 silos are estimated to result in individual lifetime risks greater than 1×10^{-4} only for indoor workers (mean: 4.3×10^{-4}) and fenceline residents (median: 1.8×10^{-4}) under the routine release scenario. Population risks associated with the routine and accidental release scenarios are less than 1.0 for all identified receptor populations.

Total individual and population risks associated with radon exposure (background plus radon from the silos) is dominated by risks associated with background radon for both the routine release and the accident scenarios.

Median individual risks associated with natural background concentrations of radon exceed risks associated with the routine release from the silos by 1 to 3 orders of magnitude (Table 5-26). Mean population risks associated with background radon are 2 to 3 orders of magnitude larger than those associated with routine releases (Table 5-27).

Risks associated with the accidental release of radon are also much smaller than median risks associated with natural background concentrations. Median individual risks associated with natural background concentrations of radon exceed risks associated with the accidental release from a silo by 2 to 5 orders of magnitude (Table 5-26). Mean population risks associated with background radon are 4 to 6 orders of magnitude larger than those associated with routine releases (Table 5-27).

Table 5-26. Individual lifetime risks.

Receptor	Background*	Routine*	Accidental**
Indoor Worker	7.3E-3	4.3E-4**	9.4E-6
Outdoor Worker	7.3E-3	1.5E-4**	3.2E-7
Fenceline Resident	7.3E-3	1.8E-4	6.3E-7
Within 1 mile	7.3E-3	7.0E-5	--
Within 5 mile	7.3E-3	2.2E-5	--
Hamilton	7.3E-3	2.8E-6	2.4-8
Cincinnati	7.3E-3	1.3E-6	1.1E-8

*median
**average

Table 5-27. Population risks.

Receptor	Background*	Routine*	Accidental**
Within 1-mile	2.0	0.01	0.0001
Within 5-mile	172.3	0.13	0.0013
Hamilton	718.0	0.30	0.0014
Cincinnati	5,268.3	0.95	0.0047

*median
**average

6 RISK ASSESSMENT FOR URANIUM IN GROUND WATER ("SOUTH PLUME")

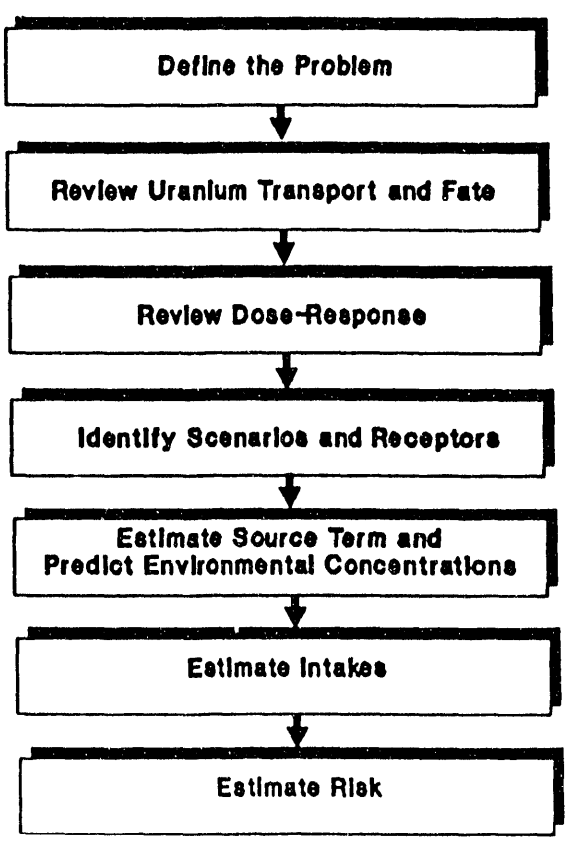
6.1 Problem Summary and Outline of Steps in the Assessment

6.1.1 Steps in the Assessment

Major steps in the risk assessment for uranium in ground water at the FEMP are listed below, and are summarized in Figure 6-1.

1. Define the problem (section 6.1).
2. Review information on uranium transport and fate (section 6.2)
3. Review dose-response information and identify appropriate cancer risk factors and toxicity threshold distributions (section 6.3)
4. Identify release scenarios, exposure scenarios and receptors (section 6.4)
5. Develop estimates of source term and predict concentrations of uranium in ground water, food and milk (section 6.5, 6.6).
6. Estimate intakes (section 6.7) by people in the potentially exposed groups.
7. Estimate individual lifetime fatal cancer risks and the annual risk of toxic effects (section 6.8)

Figure 6-1. Steps in the risk assessment for uranium in ground water at the FEMP.



6.1.2 Problem Summary

Elevated uranium concentrations have been detected in ground water south (and downgradient) of the FEMP. Concentrations of uranium greater than the proposed drinking water standard of 20 μg/L (Federal Register, 1991) are present at some locations south of the facility. The highest uranium concentrations detected are in the upper level of the aquifer.

There are two distinct areas of uranium ground water contamination at the FEMP. The "south plume" is centered outside the site boundary along Paddy's Road (Figure 6-2), and its projected path is south to the Miami River. The second plume is located onsite near the fly ash piles (Figure 6-2).

The principal sources of uranium in ground water south of the facility are historical releases of uranium in water from Paddy's Run and the storm sewer outfall ditch, which came from storm water runoff in the waste storage area. This contaminated water entered the aquifer through infiltration along Paddy's Run. This release of uranium to the aquifer through infiltration is continuing, although at greatly reduced levels.

Uranium in the storm sewer outfall ditch came from contaminated storm water diverted from the production area. This contaminated water also entered the ground water through infiltration. These releases were discontinued in 1989 with the construction of the storm water retention basin.

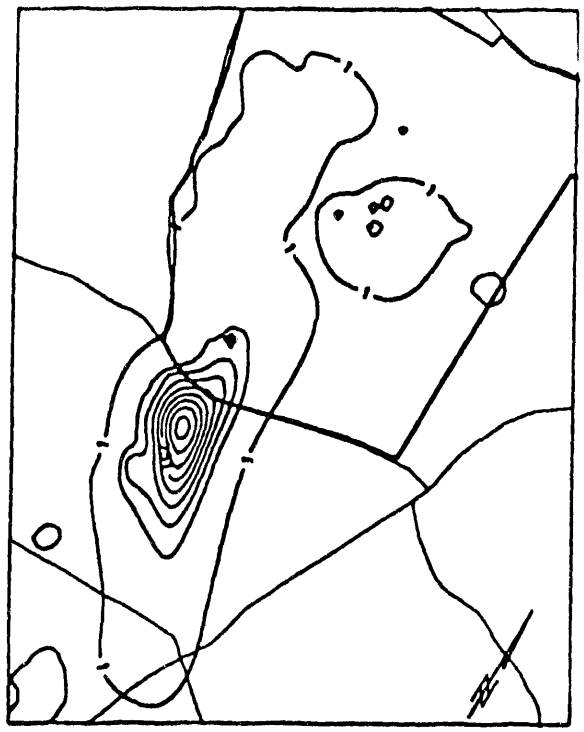
Uranium is generally considered to be a chemical toxicant first and a radiological toxicant second. The main concern is the possibility of toxic effects to people using ground water south of the facility. A number of potential ground water users are located downgradient of the FEMP. Potential future receptors include residential or agricultural wells in the path of the plume, and possible future wells in the potentially impacted area.

6.1.3 Current Status

A Remedial Investigation/Feasibility Study (RI/FS) is currently underway for Operable Unit 5, which includes the Great Miami Aquifer near the FEMP. The RI/FS investigation includes additional monitoring of the ground water, an ongoing modeling effort to better characterize uranium contamination of ground water, and a determination of the need for remedial action. Because the south plume has contaminated offsite groundwater in an area with potential human receptors, USDOE determined that a short term removal action was needed, prior to the completion of the RI/FS and implementation of the final remedial action for the regional aquifer.

An Engineering Evaluation/Cost Analysis (EE/CA) was completed (USDOE, 1990b), and the preferred alternative for the south plume removal action includes 1) providing an alternate water supply to two industrial users along Paddy's Run; 2) interception and collection of contaminated ground water near the southern limit of the south plume and discharge of contaminated water to the Great Miami River; 3) installation of an interim advanced wastewater treatment system within FEMP boundaries to treat an existing discharge to the Great Miami River, and 4) monitoring and industrial controls.

Figure 6-2. Current extent of uranium contamination in ground water (predicted by a ground water model, calibrated against monitoring data, Section 6.5.2). Concentration range is 1 - 450 μg/L, contour interval shown is 50 μg/L.



6.2 Uranium - Physical Properties and Environmental Fate

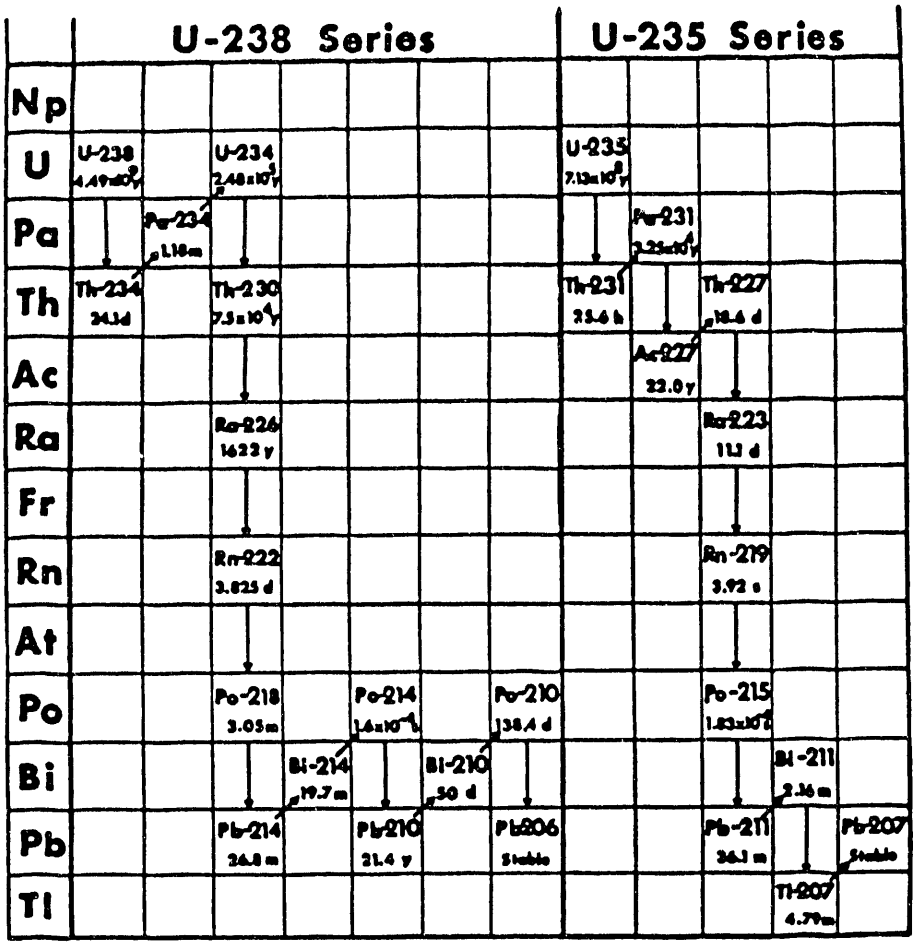
This section summarizes the physical and chemical properties of uranium, its geochemistry in soil and ground water, and the factors affecting uranium transport and fate in the environment.

6.2.1 Physical and Chemical Properties

Physical Properties

Uranium is a dense, silvery metal with a mean atomic weight of 238.03. Natural uranium contains three isotopes - 99.27% ^{238}U, 0.72% ^{235}U, and 0.0055% ^{234}U. The isotopes of uranium ultimately decay to stable isotopes of lead through a series of steps via beta and or alpha decay. The decay series of ^{238}U and ^{235}U are presented in Figure 6-3. Because the half life of ^{235}U is shorter than the half-life of ^{238}U, the ratio of $^{238}U/^{235}U$ has not been constant throughout earth's history. 4.55 billion years ago the ratio $^{238}U/^{235}U$ was only 3.33, compared to the current ratio of 136.5.

Figure 6-3. Uranium-238 and uranium-235 decay series.



Radioactive Materials and Nuclear Reactions

The two natural uranium decay series shown in Figure 6-3 lead to the formation of two stable isotopes of lead - ^{238}U decays via the production of 8 alpha particles to ^{206}Pb, and ^{235}U decays with the release of 7 alpha particles to ^{207}Pb.

Each radionuclide in the ^{238}U decay chain has a much longer half-life than the corresponding member of the ^{235}U chain with the exception of ^{231}Pa. Because of their relative longevity the presence of some ^{238}U daughters in the environment is of particular concern - namely ^{230}Th (ionium), ^{226}Ra, ^{222}Rn, ^{210}Pb, and ^{210}Po.

Uranium can undergo spontaneous fission in nature, or fission can be induced by bombarding ^{235}U or ^{233}U with neutrons. The enrichment of ^{235}U in natural uranium leads to the production of fission bomb and fuel uranium. The by-product of that enrichment process, from which ^{235}U has been removed, is known as depleted uranium.

Activity to Mass Ratio

The concentration of uranium in air or in aqueous solution can be expressed either as mass/volume or activity /volume. The principle isotopes of natural uranium, ^{238}U, ^{234}U and ^{235}U have different half-lives; therefore the same mass of each isotope will have different activities. Since the activity of a uranium solution depends on the relative abundance of uranium isotopes, conversion between the two ways of expressing uranium concentration requires some assumption about the relative abundance of each uranium isotope. The activity ratio is defined as the decays of ^{234}U relative to the decays of ^{238}U per unit time in a sample ($^{234}U/^{238}U$). The activity ratio of natural uranium at equilibrium is 1.

Because the concentration of ^{235}U in non-enriched uranium is so small, (Weston, 1987 indicates that the average plant product was depleted in ^{235}U) the contribution of ^{235}U to contaminant uranium at FEMP can be ignored. For the assessment at the FEMP, it was assumed that the uranium in ground water consists of ^{234}U and ^{238}U.

The activity/mass (pCi/mg) of naturally-occurring uranium can then be calculated from the measured activity ratio of $^{234}U/^{238}U$, AR, and the specific activity, a_S, of ^{238}U:

$$pCi/mg = (1+AR)*a_S$$

where the specific activity a_S of ^{238}U is calculated from the decay constant, λ, 4.889×10^{-18} sec^{-1}; Avogadro number, AV, 6.02×10^{23} atoms/mol; the atomic/molecular weight of ^{238}U, mol_U, 238 g/mol; and the value for a pico-Curie, pCi, (3.7×10^{-2} dis/sec·pCi):

$$a_S = \frac{\lambda * AV}{pCi * mol_U}$$

$$a_S = \frac{4.889 \times 10^{-18}\ sec^{-1} \cdot 6.02 \times 10^{23}\ atoms/mol \cdot 10^{-6}\ \mu g/g}{3.7 \times 10^{-2}\ dis/sec{\cdot}pCi \cdot 238.03\ g/mol}$$

$$a_S = 0.334\ pCi/\mu g$$

The estimate of the relative abundance of each uranium isotope for a given field situation should be based on an assessment of the geochemical environmental processes which alter the relative abundance of each isotope. For the purpose of this discussion it can be safely generalized that the relative abundance of uranium isotopes in solid rock and most soil materials will be very close to those expected at decay equilibrium, i.e., crustal abundance. Where there is no weathering, for example on the moon, there is no fractionation of uranium isotopes (Rosholt & Totsumoto, 1971). Heavily weathered surficial rocks will have activity ratios ($^{234}U/^{238}U$) lower than 1.00. Activity ratios as low as 0.5 have been measured in some weathered rocks (Richardson, 1964). The decay of ^{238}U produces two intermediate radionuclides, ^{234}Pa and ^{234}Th which along with ^{234}U may be more soluble than ^{238}U. ^{234}U is therefore preferentially removed by solution from weathered rocks, and dissolved uranium can be expected to have an excess of ^{234}U activity over ^{238}U, over the expected equilibrium concentration while ^{235}U will remain relatively unchanged. Therefore the activity of dissolved uranium will generally be greater than the activity of uranium at decay equilibrium since the half-life of ^{234}U (2.47×10^5 y) is much less than the half-life of ^{238}U (4.51×10^9 y).

Most investigators use an activity to mass ratio of 0.68 pCi/μg (AR = 1.00) to estimate the activity in a sample that has been measured as total uranium by mass. This conversion factor assumes that the three natural uranium isotopes are present at levels corresponding to the crustal abundance percentages. USEPA uses data from the National Inorganics and Radionuclides Survey (NIRS) (USEPA, 1990a; USEPA, 1991a) to support a higher ratio of 1.3 pCi/μg in natural water. This higher value can occur because natural geochemical and radiological processes may alter the proportions of isotopes. Because the contaminant uranium present at FEMP is not "natural", i.e., it is not leaching from soil and rock, and because concentrations of uranium in the contaminant plume are relatively high (> 1 μg/L), the conversion factor of 0.68 is used here. For background values where disequilibrium may have been established over long periods of time, and where the concentrations of dissolved uranium are relatively low (< 1 μg/L), the higher activity/mass ratio may be appropriate.

6.2.2 Geochemistry of Uranium

6.2.2.1 Occurrence and Distribution of Natural Uranium

The elements of which the earth is composed are distributed in concentrations that are roughly inversely proportional to the element's mass - the heavier the element, the rarer it is. Yet uranium, element 92, is the 46th most abundant element in the crust of the earth - more abundant than tin, arsenic, tungsten, cadmium or mercury, for example (Goldschmidt, 1954). This enrichment of uranium near the earth's surface is a consequence of the unique geochemical properties of this element - its association with certain lithophile mineral assemblages and its behavior in solution.

6.2.2.2 Solid Earth

Rocks

Uranium is present in measurable amounts in almost all common minerals. Some minerals, zircon for example, may have notably high concentrations of uranium, but the major ore of uranium is the oxide uraninite or "pitchblende", ranging in composition from UO_2 to U_3O_8. Another common mineral association of uranium is carnotite, $K_2[UO_2]_2V_2O_8$. Uranium commonly occurs in association with apatite, organic shales and peat in quantities that may be economic to recover if the mined material is used for other purposes.

Uranium in rocks at FEMP is not a significant source of background levels of the isotopes of importance in the study of ground water contamination since the contaminant plume is restricted to the soil horizons and alluvial deposits immediately west and south of the site. The behavior of uranium in soil is, therefore, much more important to understanding the nature of contaminant transport than is the concentration of uranium in rock.

Soil

Uranium not associated with unweathered rock minerals will be contained in or sorbed on the surface of organics, clays, oxides and other detrital and authigenic components of soils.

Soils in the FEMP area (see Lerch et al., 1982) were formed from parent materials, primarily loess and till, deposited by Pleistocene glaciers of Wisconsinan and Illinoian age. Two soil types have developed from these materials - a Fincastle-Xenia silt loam and a Fox-Genesee loam. The upper meter of both soils is strongly acid while below 1 meter each soil is mildly to moderately alkaline. Organic and clay contents for each soil are moderate to low. The Fincastle soil is found at the FEMP itself while the Fox-Genesee soil is located along Paddy's Run.

Fincastle-Xenia silt loam (an alfisol [mesic Aeric Ochraqualf-Aquic Hapludalf]) developed on 18 to 40 inches of loess over a limey silty till. This soil has poor drainage retaining high water contents during the winter and spring. Fox-Genesee loam (an alfisol [mesic Typic Hapludalf]), in contrast, is a well-drained soil developed on 24 to 40 inches of silty material over sand and gravel. Fox-Genesee soils are restricted to the Paddy's Run floodplain.

Naturally occurring ^{238}U concentrations in the soil in Ohio range from about 0.6 to 2.2 pCi/g (Myrich et al., 1983). The 1989 Environmental Monitoring report (USDOE, 1990c) assumes secular equilibrium between ^{234}U and ^{238}U and states that the total uranium activity will be about twice this amount because the three isotopes occur together in soil. The value used as "background" appears to be 4.4 pCi/g total uranium (6.5 μg/g) (USDOE, 1990c).

6.2.2.3 Ground Water

Uranium which originates from rocks at the surface of the earth will first enter solution by two roughly independent processes - one is the chemical weathering and disintegration of silicate and other minerals; the second is by a radiochemical process which preferentially delivers ^{234}U into solution relative to ^{238}U.

The first process, weathering, will lead to increased concentration of uranium in solution in proportion to the extent and duration of the weathering process. Thus, streams which drain heavily-weathered terrain with thick, permeable regolith will tend to have high concentrations of uranium. Similarly, ground water in contact with aquifer rocks under aerobic conditions will tend to continuously increase in uranium concentration with time.

The second process, radiochemical, is unique in nature to uranium and uranium-series elements. ^{234}U, a product of the decay of ^{238}U is preferentially removed from minerals following the decay. A mineral grain which in its unweathered state is in secular equilibrium with respect to the uranium-series isotopes will, in contact with water during weathering, tend to acquire a $^{234}U/^{238}U$ activity ratio somewhat less than unity depending on the rate at which ^{234}U is removed from its surface. Consequently most natural waters have $^{234}U/^{238}U$ activity ratios greater than unity.

The source of ground water is recharge through soils and weathered rock. Since soils and weathered rock are areas where uranium-series isotopes will be fractionated it is to be expected that most ground water will exhibit some degree of uranium-series disequilibrium dependent on conditions within the recharge zone.

As a consequence of uranium's redox-dependant behavior, for the purpose of discussing uranium in ground waters, such waters can be divided into three types (Osmond & Cowart, 1976):

within a matrix while chemical or biochemical reactions progress or because the solution moves from a region with certain chemical characteristics to another region with different characteristics. Each or any combination of these factors may remove uranium from or add uranium to a solution.

The fundamental properties of a solution can be summarized by a limited set of operators - the pH (or hydrogen ion activity of the solution), the Eh (pE or redox potential of the solution) and other controlling factors such as CO_2 or H_2S activities. A steady-state, equilibrium model could be used to predict how changes in those fundamental solution properties will change the concentration of uranium in solution. Natural solutions are not necessarily in equilibrium, however. As noted, for example, by Honeyman & Santschi (1988) there are a range of problems involved in applying equilibrium geochemistry to solute behavior in surface and ground waters, namely:

1. heterogeneous adsorption surfaces;
2. particle concentration effects;
3. non-additivity of multiple-adsorbent systems; and,
4. slow kinetics of sorption.

Despite such limitations, the first step in characterizing the behavior of uranium in the environment is to apply equilibrium concepts. That equilibrium formulation can then form the basis on which to assess departures from steady-state, etc.

Solution/Precipitation - pH and Oxidation-Reduction

Under equilibrium conditions the solubility of uranium species in water is a function primarily of pH and redox potential (Eh). The traditional way to present this two-dimensional behavior is by use of pH-Eh diagrams (Garrels & Christ, 1965; Krauskopf, 1979). In such diagrams predominance field for species of concern are plotted with boundaries defined either by a line of equal concentration with other species or by limits of the stability of water. To produce a diagram such as that given in Figure 6-4, one evaluates each reaction in turn for its dependance on pH and/or Eh. For example, the reaction:

$$UO_2^{2+} + 2e^- \longrightarrow UO_2$$

is entirely redox dependent and independent of pH. In contrast, the reaction:

$$U^{4+} + 2H_2O \longrightarrow UO_2^{2+} + 4H^+ + 2e^-$$

is dependent on both pH and redox potential. The position of the lines delineating fields on an Eh/pH diagram are calculated from the Nernst equation and are based on free energies (ΔGs) for each reaction (see, e.g., Garrels & Christ, 1965).

The limits on the stability of water, namely an upper limit defined by the oxidation of water releasing free oxygen and a lower limit for reduction of water releasing hydrogen gas, fix upper and lower bounds on the figure above and below which one would normally not expect aqueous reactions to occur.

The implications for the behavior of uranium and one of its daughters, thorium, can be summarized by examining Figure 6-4. Note that uranium (as UO_2 or $U(OH)_4$) is insoluble under virtually all reducing conditions except at very low

within a matrix while chemical or biochemical reactions progress or because the solution moves from a region with certain chemical characteristics to another region with different characteristics. Each or any combination of these factors may remove uranium from or add uranium to a solution.

The fundamental properties of a solution can be summarized by a limited set of operators - the pH (or hydrogen ion activity of the solution), the Eh (pE or redox potential of the solution) and other controlling factors such as CO_2 or H_2S activities. A steady-state, equilibrium model could be used to predict how changes in those fundamental solution properties will change the concentration of uranium in solution. Natural solutions are not necessarily in equilibrium, however. As noted, for example, by Honeyman & Santschi (1988) there are a range of problems involved in applying equilibrium geochemistry to solute behavior in surface and ground waters, namely:

1. heterogeneous adsorption surfaces;
2. particle concentration effects;
3. non-additivity of multiple-adsorbent systems; and,
4. slow kinetics of sorption.

Despite such limitations, the first step in characterizing the behavior of uranium in the environment is to apply equilibrium concepts. That equilibrium formulation can then form the basis on which to assess departures from steady-state, etc.

Solution/Precipitation - pH and Oxidation-Reduction

Under equilibrium conditions the solubility of uranium species in water is a function primarily of pH and redox potential (Eh). The traditional way to present this two-dimensional behavior is by use of pH-Eh diagrams (Garrels & Christ, 1965; Krauskopf, 1979). In such diagrams predominance field for species of concern are plotted with boundaries defined either by a line of equal concentration with other species or by limits of the stability of water. To produce a diagram such as that given in Figure 6-4, one evaluates each reaction in turn for its dependance on pH and/or Eh. For example, the reaction:

$$UO_2^{2+} + 2e^- \longrightarrow UO_2$$

is entirely redox dependent and independent of pH. In contrast, the reaction:

$$U^{4+} + 2H_2O \longrightarrow UO_2^{2+} + 4H^+ + 2e^-$$

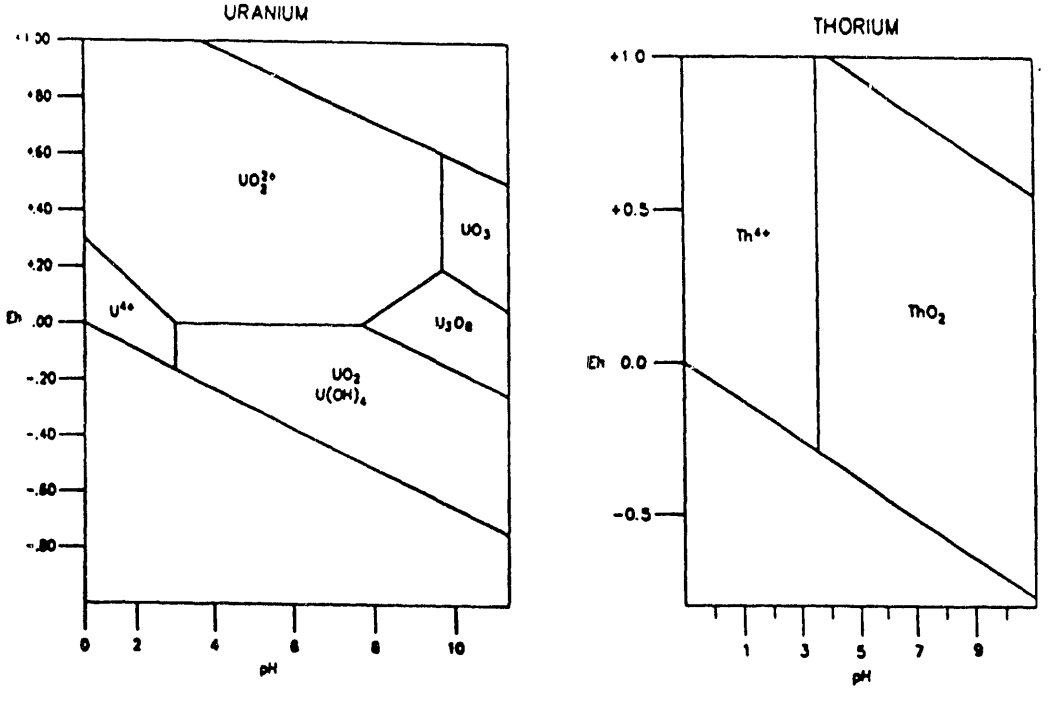
is dependent on both pH and redox potential. The position of the lines delineating fields on an Eh/pH diagram are calculated from the Nernst equation and are based on free energies (ΔGs) for each reaction (see, e.g., Garrels & Christ, 1965).

The limits on the stability of water, namely an upper limit defined by the oxidation of water releasing free oxygen and a lower limit for reduction of water releasing hydrogen gas, fix upper and lower bounds on the figure above and below which one would normally not expect aqueous reactions to occur.

The implications for the behavior of uranium and one of its daughters, thorium, can be summarized by examining Figure 6-4. Note that uranium (as UO_2 or $U(OH)_4$) is insoluble under virtually all reducing conditions except at very low

pH values (below about pH 3) Under oxidizing conditions uranium will be soluble even at very high (8 to 10) pHs. Thorium, on the other hand, will be essentially insoluble under all redox conditions at pHs greater than about 3.5. This sharp contrast between the behavior of U and Th is responsible for much of the disequilibrium in U/Th isotopes and explains why in many systems, U/Th daughters are found physically separated from their parent nuclides.

Figure 6-4. pH-Eh diagrams for uranium and thorium (after Brookins, 1984).



Sorption, K_d and retardation coefficient

The transport of solutes through ground water is often not a conservative process, i.e. solutes may interact with the solid matrix of the saturated zone so that solute concentrations may change with time and position within the moving body of ground water. That interaction will have the effect of slowing down the solute transport relative to the solvent, namely the water. These interactions should occur under equilibrium conditions and are ideally reversible, i.e., the quantity of solute associated with solid surfaces should be a linear function of the concentration of the solute in solution and should not involve any hysteresis when the process is reversed. This ideal process is known as sorption (Domenico & Schwartz, 1990) and should include such processes as cation and anion exchange, hydrophobic and hydrophilic reactions, etc. In practice all sorption phenomena are included within the concept of the distribution coefficient or K_d defined as the ratio of the concentration on solids, S, relative to the concentration in the solution, C:

$$K_d = \frac{\text{concentration on solids}}{\text{concentration in solution}} = \frac{\text{mass sorbed/mass of soil}}{\text{mass dissolved/volume of solution}} = S/C$$

If S is measured in terms of mg/g and C in mg/ml, then K_d will be expressed in terms of ml/g. The curve defining concentration in solution versus concentration on solids defines the linear sorption isotherm, a function which is, in reality, rarely linear. The Freundlich and Langmuir relationships are two different attempts to define the character of what, in reality, are non-linear sorption isotherms.

A parallel and related concept to that of K_d is the retardation coefficient, R, which is a measure of the degree to which contaminants have their flow velocity slowed down by interaction with solids relative to the solvent, water. R is defined by:

$$R = \frac{1 + d_b \cdot K_d}{n_e}$$

where d_b is the bulk density, and n_e the effective porosity of the soil or aquifer matrix.

Because geochemical conditions are often so difficult to quantify and because solute/matrix reactions are likely to be controlled by kinetic rather than equilibrium processes, detailed speciation modeling of ground water regimes is rarely performed. As a consequence the K_d and R concepts become a catchall for both sorption and solution/precipitation processes some of which may be irreversible and/or non-linear. Several authors have cautioned about the indiscriminate use of K_ds. For example, Domenico and Schwartz (1990) point out that "the processes are far too complex to be represented accurately by a simple one-parameter model." Honeyman and Santschi (1988) note, also, that K_ds tend to be site and material specific.

Despite these limitations, K_ds are used in every attempt to realistically model solute transport. Without the use of K_ds and R, velocities of solute transport under most field conditions would be unrealistically high since virtually all contaminants interact with the solid aquifer matrix. At FEMP, IT Corporation has calculated a range of K_ds and Rs based on calibration of a three-dimensional solute transport model (IT, 1991a). They report a range of K_ds from 1.0 to 1.34 and a corresponding range of 9 to 12 for R for the matrix of the south plume. While these K_d values are low relative to those general purpose values (K_d about 40) recommended by reviewers (see, e.g., Looney et al., 1987; Sheppard et al., 1984; Baes & Sharp, 1983; Ames & Rai, 1978; Callahan et al., 1979) they are consistent with the expected geochemical nature (moderate pH and oxidizing) of the FEMP surficial, unconfined aquifer.

6.2.3 Biological Uptake

Uranium is not readily taken up by plants because it is usually strongly fixed in surface soils. Sheppard et al. (1983) found uptake by swiss chard restricted to the root system. NCRP (1984) recommends default values of the soil to plant concentration ratios for uranium ranging from 3×10^{-4} to 3×10^{-3}.

Transfer to milk and meat is also relatively small. The parameter most often used in exposure assessment to describe the transfer of a nuclide to milk (f_{im}) is the

fraction of a nuclide ingested daily by a lactating animal that is secreted in 1 liter of milk under steady state conditions (NCRP, 1984). The default range of values for f_{im} recommended by NCRP (NCRP, 1984) is 1.2×10^{-4} to 6.0×10^{-4} day/L.

The parameter f_{if} is the fraction of a nuclide ingested daily by an animal that is found in 1 kg of muscle under steady state conditions (NCRP, 1984). NCRP (1984) recommends default values from 1.6×10^{-6} to 5.0×10^{-3} day/kg.

Bioaccumulation factors are used to describe the ratio of a nuclide concentration in an aquatic organism to that in the overlying water. Reported bioaccumulation factors for fresh water organisms range from 0.3 - 38 for fish and 2-40 for invertebrates (NCRP, 1984).

6.3 Human Health Effects

6.3.1 Introduction

Intake in food is the main source of uranium for the general population, although drinking water may be an important source in areas where concentrations in ground water are naturally high. Occupational exposures are primarily through inhalation.

Uranium tends to accumulate in bone and kidney. Potential health effects associated with the ingestion of uranium include chemical toxicity (kidney is the main target organ) and an increase in cancer risk (primarily bone carcinomas).

The following sections review the potential health effects associated with the ingestion of uranium in food and water, and document the parameters and dose-response relationships used in the risk assessment for uranium in ground water at the FEMP.

6.3.2 Toxicity

6.3.2.1 Toxic Effects in Humans

Toxicity effects have been demonstrated in animals for both long-term and short-term exposures to uranium, with the primary effect in the kidneys. Most animal data indicate that uranium accumulates on the brush border of the epithelial cells lining the lumen of the proximal convoluted tubules (USEPA, 1991b). Effects in animals include an increase in glucose, low molecular weight proteins and amino acids in the plasma and urine, and lesions in the proximal convoluted tubules of the kidney.

There is only limited evidence of toxicity effects from uranium ingestion in humans. Two case studies involving intravenous injection of uranyl nitrate in a total of 11 patients (Luessonhop et al. (1958), and Basset et al., (1948) as reported in Moore (1984) and USEPA (1991b)) suggested that kidney may be a target organ but did not clearly demonstrate effects on the kidney (USEPA, 1991b, NRC, 1988). Numerous animal studies identified the kidney as the principal target of ingested uranium.

One epidemiological study of ingested uranium is available. Persons from two towns in Russia with drinking water concentrations of 0.04 mg/L and 0.003 mg/L were studied (Novikov et al., 1968; Novikov 1967, 1972 as cited in USEPA 1991b). The only difference observed between the two populations was a relative decrease in serum albumin/globulin ratio for people living in the town with the higher uranium water concentration (USEPA, 1991b).

This analysis focuses on potential nephrotoxic effects of long-term exposure to uranium via ingestion such as occurs with drinking water contamination.

6.3.2.2 Realistic Risk Analysis for Uranium Toxicity

Kidney damage is generally accepted to be the principal toxic effect of ingested uranium. Much is known about this effect, but many questions remain and the nature of the pathology makes quantification difficult. In the past, uncertainties have been overcome by including conservative assumptions (i.e., assumptions leading to overestimates of risk) in the development of a dose-response function.

This approach satisfied radiation protection and regulatory needs, but is unsatisfactory for use in risk assessments for several reasons: 1) it provides no insight into the relative uncertainty contributed from different factors making up the dose-response function, and so does not provide useful feedback on priorities to biomedical researchers; 2) combined with conservative estimates on exposure, it provides no information on the size of the overestimate, leading to inappropriate judgments on the relative risk of uranium when compared to other pollutants; 3) it may result in an excessive degree of conservatism, leading to wasteful expenditures of resources that could be spent more effectively elsewhere; and 4) it puts the decision on how conservative one should be in the scientific arena rather than in the political arena, often obscuring the relative contribution of data and assumptions.

In this analysis, an attempt was made to describe uncertainties explicitly, a task appropriate for science. When combined with exposure estimates that also include explicit uncertainty estimates, the public will have sufficient information to make rational decisions balancing cost and benefit.

USEPA recently proposed drinking water regulations for uranium (Federal Register, 1991) following a traditional conservative approach to risk analysis. The sources of the differences between our results and those of the USEPA are examined in Appendix A. To provide a reference point to the USEPA proposed standard, we compare the parallel USEPA value to our distribution for each parameter. Since we take a more detailed approach to estimating toxicity levels, there is not always a parallel USEPA value.

6.3.2.3 Method

Risk of kidney damage was modeled using a modification of the method used by ICRP (ICRP 1960; ICRP, 1979; Rich et al., 1988). ICRP (1979) quantified these relationships with specific numbers; here the parameters were generalized and characterized as probability distributions to explicitly include uncertainty. The model was then solved using a Monte Carlo technique.[1] The analysis determines the probability of exceeding a threshold for a given daily ingestion rate. The derivation of the probability distributions used is described below.

[1]Crystal Ball version 2 for Windows by Decisioneering, Inc., operating in Microsoft Excel.

The model includes several steps: fractional transfer from the gut to the blood and from the blood to the kidney, a biological half-life in the kidney, and an effect based on concentration of uranium in the kidney. The retention function for ingested uranium in the kidney at t days after ingestion is given by the following formula:

$$Rk(t) = f_1 * [\alpha_1 * e^{-t \ln 2/T1} + \alpha_2 * e^{-t \ln 2/T2}]$$

Where:
f_1 = fractional transfer gut to blood
α_i = fractional transfer blood to kidney term i
T_i = biological half life of U in kidney term i

Integrating over t to express build up associated with continuous exposure over time gives:

$$Rk(t) = f_1 * [(\alpha_1 T_1/\ln 2)(1-e^{-t \ln 2/T1}) + (\alpha_2 T_2/t \ln 2)(1-e^{-t \ln 2/T2})]$$

For equilibrium conditions (t = infinity), the amount in the kidney (A_∞) is:

$$A_\infty = I * (f_1/\ln 2) * [\alpha_1 T_1 + \alpha_2 T_2]$$

where I = daily intake (μg)

Wrenn et al. (1985) use this model, although they represent retention in the kidney in a single set of parameters, eliminating the second term in both equations. Kocher (1989) discussed the quantitative difference between kidney burdens obtained with both model forms. The effect in the kidney is usually expressed as a threshold level (Rich et al., 1988), but different classes of effect with different severity may occur at different threshold concentrations.

6.3.2.4 Characterization of the Probability Distributions

Fractional transfer, gut to blood (f_1)

Only a small fraction of ingested uranium reaches the bloodstream. Uptake through the gut is fairly quick, so it is not necessary to consider time dynamics, especially when the concern is for the risk associated with chronic exposure. It is generally assumed that there is little or no transfer of uranium from blood back to the GI tract. Absorption to blood is not directly measured, but is inferred from other measurements. Some of the differences seen in different experiments may be due to different approaches to estimating the fractional transfer to blood.

ICRP (1979) reviewed studies that estimated the percentage of ingested soluble uranium compounds absorbed through the gut to the bloodstream. The percentages reported in these studies ranged from 0.5% to 5%. Taking a conservative approach, ICRP (1979) selected the upper end of this range for use in its analyses. Wrenn et al. (1985) conducted an exhaustive review and analysis of the metabolism of ingested uranium and radium. As an annex to this review, they

conducted a meta analysis of the reviewed data. They concluded that all data for man and animals, except rats, fell on the same curve (linear on a log-log plot). This curve (Wrenn et al., 1985, Figure A-1) suggested a decline in absorption with increasing dose, but the slope was not statistically significant. Their curve for absorption in rats appeared similar in slope, but more than an order of magnitude lower (mean of 1.2% Ŷ 0.57 vs 0.055% Ŷ 0.016, with the "least reliable result" omitted in each case). Their "consensus" estimate of f_1 was 1.4%. In reaching this value, they took into account the (non-significant) slope and omitted what they termed the least reliable result. The latter was a mixture of analysis of city-wide "market basket" samples with uranium excretion rates from selected small populations.

LaTouche et al. (1987), reporting new experimental results, demonstrated that rats were not different; but that the displacement of the rat data represented feeding ad libitum while data on humans and other animals were based generally on overnight fasting. This was further supported by a recent correction in one data point for rabbits fed ad libitum (Tracy et al., 1992), which put this data point among those for the rats.

We repeated Wrenn et al.'s analysis (Wrenn et al.'s Figure A-1, LaTouche et al.'s Figure 7), separating "fasted" and "fed" groups. For the "fasted" group, we followed Wrenn et al. by omitting their point E1 (the least reliable point). We moved the rat studies used by Wrenn et al. (Their Table A-2, points 6a-6e) and Tracy et al's rabbit study (Wrenn et al.'s Table A-2, point 2) to the fed group. We added LaTouche et al.'s rat results and new results on fasting human volunteers (Wrenn et al., 1989, 1990). We also analyzed results of four studies of human populations exposed only to natural background levels (Singh et al., 1990; Spencer et al., 1990; Dang et al., 1992; Masuda, 1971). The analysis examined the variation among the individual study results. It did not consider the uncertainty within each individual study; in some cases insufficient information was available to calculate this uncertainty.

There were thus three data sets: the fasted group, the fed group, and the environmental background group. These are listed in Table 6-1. None of the data sets showed a statistically significant trend in absorption with intake. We also combined the fasted and environmental background groups, seeking a trend in all studies below 0.1 μg/kg/d and below 1.0 μg/kg/d; no statistically significant trend was found. With no dose rate effect, one would expect that the free human population would exhibit absorption between the fed and fasted groups, nearer to the "fed" group, but including some ingestion of uranium in water before breakfast. This was not borne out by the results. The mean of the environmental background studies was remarkably similar to that of the fasted group and incompatible with the fed group (Table 6-2).

We represent absorption of uranium through the gut to the bloodstream with the lognormal distribution described in Table 6-2 for the environmental background group. This distribution has a geometric mean of 1.03, and 95 percentile bounds of 0.33 and 3.17.

Table 6-1. Data used to estimate uranium uptake through the gut.

Fasted Group from Wrenn et al., 1985, with LaTouche and Wrenn 89 added					
Data Set		Species	ug/kg/d	%Absorb	
A2-1	Harrison and Stather, 81	hamster	630	0.77	
A2-3	Fish et al., 60	dog	700	1.55	
A2-4	Larson et al., 84	baboon	0.5	1.2	
A2-5a	Butterworth, 58	man	6700	0.73	
A2-5b	Hursh et al, 69	man	132	1.4	
A5-2	Yamamoto et al., 68; Masuda, 1971 a-d	man	0.15	1.6	
A5-3	Fisher et al., 83	man	0.34	0.76	
A5-4	Somayajula et al, 1980	man	0.76	2.2	
LaT-1	Latouche et al., 87	rat	30	0.78	
LaT-2	Latouche et al., 87	rat	300	1.08	
LaT-3	Latouche et al., 87	rat	3000	1.78	
LaT-4	Latouche et al., 87	rat	3.00E+04	0.64	
LaT-5	Latouche et al., 87	rat	4.50E+04	2.82	
Wr89	Wrenn et al., 89	man	3.57E+00	0.6	
Environmental Exposure Studies of Humans					
Data Set			ug/kg/d	%Absorb	
Dang	Dang et al., 92	man	0.011	1.6	
Masuda-T	Masuda, 71	man	0.131	1.61	
Masuda-N	Masuda, 71	man	0.080	1.32	
Masuda-A	Masuda, 71	man	0.056	0.69	
Masuda-U	Masuda, 71	man	0.025	0.34	
Spencer	Spencer et al, 90	man	0.063	1.5	
Singh	Singh et al, 90	man	0.067	1	
"Fed" Group from Wrenn Table A-2, with Tracy rabbits added					
Data Set			ug/kg/d	%Absorb	
Tracy	Tracy et al., 92	rabbit	3.90E+04	0.06	
6a	Hamilton, 48	rat	3.00E+02	0.35	
6b1	Sullivan, 80	rat	2.3	0.06	
6b2	Sullivan, 80	rat	4.00E+03	0.06	
6c1	Sullivan, 83	rat	5.10E+03	0.044	
6c2	Sullivan, 83	rat	1.30E+04	0.044	
6c3	Sullivan, 83	rat	2.50E+04	0.088	
6d	Tracy et al., 1983	rat	3.30E+04	0.035	
'6e1	Maynard et al, 53	rat	2.00E+04	0.052	
'6e2	Maynard et al, 53	rat	9.60E+04	0.059	
'6e3	Maynard et al, 53	rat	2.00E+05	0.06	
'6e4	Maynard et al, 53	rat	1.20E+05	0.038	
'6e5	Maynard et al, 53	rat	4.70E+05	0.078	
'6e6	Maynard et al, 53	rat	9.70E+05	0.04	

Table 6-2. Comparison of absorption of uranium through gut to the bloodstream per unit intake from three classes of study (data given in Table 6-1).

Study Design	Geometric Mean	Geometric Std. Dev.
Overnight Fasting	1.14	1.63
Environmental Background	1.03	1.78
Fed Ad Libitium	0.06	1.77

Much clearly remains to be explained about factors affecting absorption of uranium through the gut. There is some suggestion that absorption rates may be different for uranium ingested in water relative to food. Spencer et al. (1990), using total intake (food plus water), estimated 1.5% uptake, while if only water was considered, uptake was 5%. They concluded that water was the principal source of gastrointestinal uptake of uranium in humans; uptake from food was insignificant. This would appear to be contradictory to results of animal studies in which the dose was given in food, but perhaps relates to differences in solubility of uranium in food at background levels relative to higher experimental intake levels. Singh (1988) also suggested higher uptake of uranium in water in dogs, possibly related to the fasting effect or to the uranium in water being in a more soluble form. Sullivan et al. (1986) found introduction of an oxidizing agent into the GI tract increased absorption to the blood. They suggest that food residues and secretions such as bile have a reducing effect, tending to shift uranium to the tetravalent state, which is far less soluble (ICRP,1979 suggests an uptake value of 0.2% for insoluble uranium compounds such as UO_2 and U_3O_8). While some of these suggestions have merit, there is inadequate data at present to adequately quantify their implication.

Fractional transfer, blood to kidney (α) and retention half-time (T)

Uranium in the blood complexes with bicarbonate. Thus, although the chemical form and valance state of the uranium is important for uptake through the gut, once in the blood the original form of the uranium is assumed to make little difference. This bicarbonate complex enters the kidney tubules. "In the proximal convoluted tubule, the urine becomes more acid, and the complex with bicarbonate is dissociated, leaving the uranyl ion free to attach to protein of the tubular epithelial cells" (Stannard, 1988, p. 93).

In estimating blood to kidney transfer rate, time dynamics are important. Wrenn et al. (1985) relied on studies of comatose people described by Hursh and Spoor (1973), in combination with animal data, because they were the only data with the needed time dynamics. Although the kidney is the primary site of concern for toxic effects, uranium is transferred to other sites also, especially bone. In experiments that follow the effect of a single injected dose, as uranium is excreted in urine, the concentration in blood decreases, and uranium originally deposited in bone is released back into the bloodstream, which carries it to the kidney. The observed time curve of uranium concentration in the kidney extends further in time than would otherwise be expected because of the (continually declining) deposition of "new" uranium released from bone. Here, however, we treat equilibrium conditions associated with long term, chronic exposure.

The Wrenn results were supplemented with those of a recently published study (Tracy et al., 1992). These data, used to derive probability distributions for α

and T, are given in Table 6-3 (ICRP, 1979 numbers for reference only). Both Hursh and Spoor (1973), an important basis for ICRP (1979), and Wrenn et al. (1985), who also relied in part on Hursh and Spoor, used short-term data (up to 74 days) as the basis for the α value, even though Hursh and Spoor include data out as far as 556 days. While these results were used to predict build-up in the kidney associated with a continuing, chronic exposure, there are no studies that directly observed effects of a chronic exposure in man. The use of a single α value by Wrenn et al. (1985) and Tracy et al.(1992) missed the long-term flattening out of the curve, clearly shown by Hursh and Spoor (their Figure 4.2). This was compensated for, however, by a longer biological half-time. These studies followed the dynamics of urinary excretion and of uranium retention in the kidney following a single injected dose. Tracy et al. (1992) reported a 91 day chronic exposure study of rats and rabbits in which the dose was given via drinking water. They developed a biokinetic model for uranium, using their own and other data. Although uptake from the gut to blood appears to be much smaller in their rats and rabbits (fed ad libitum), they concluded that "...once uranium enters the bloodstream, uptake and retention by the kidney appear to be similar for all three species" [rats, rabbits, and man]. One might expect to find an inverse correlation between α and T since they are estimated together; it can be seen from Table 6-3, however, that this is not the case. Independent lognormal distributions of both parameters based on the tabulated values from Wrenn et al. and Tracy et al. were used in the analysis. Wrenn et al. (1985) indicated that, below an (unspecified) critical uranium concentration, uranium remains in solution, indicating the possibility of a threshold below which there is no retention. This possibility was not included in the analysis. Although the distribution of α extends down to zero, the lower 95% confidence bound on the distribution is 1.84.

Table 6-3. Blood to kidney fractional transfer rates with half-times.

	α_1 (%)	T_1 (days)	α_2 (%)	T_2 (days)
ICRP 1979*	12.	6.	0.052	1500.
Wrenn et al.	11.	15.	0	0
Tracy et al. - Rats	>7.2**	5-9	0	0
Tracy et al. - Rabbits	3.2	13.6	0	0
Geometric mean	6.3	11.1	--	--
Geometric Std. Dev.	1.87	1.55	--	--

Rat and Rabbit estimates from Tracy et al. (1992).
*ICRP 1979 numbers for comparison only, not included in the calculation of means and standard deviations.
**Tracy et al. (1992) begin with an upper bound of 45%, but then show through their analysis that it is unrealistically high.

Kidney Toxicity

Wrenn et al. (1985) considered irreversible kidney injury from uranium to be a non-stochastic, threshold effect. The generally accepted threshold value used for kidney damage from uranium in the past was 3 μg/g kidney. Confidence in this number seemed primarily based on the absence of clinical symptoms in overexposed

workers (Alexander, 1988, p. 153). For example, Moore (1988) reported that histopathology of kidneys from long-term, retired workers, occupationally exposed to uranium at this level could not be distinguished from unexposed individuals. Although experience with occupational exposure appears widely accepted to validate the 3 μg/g threshold, uranium concentrations in the urine of workers was closely monitored and, when it appeared a worker had reached the threshold levels, he would be removed from exposure until the level in urine returned to normal (Moore, 1988, p. 155). This suggests that either the people were removed from the exposure before kidney damage was done or that repair takes place once the exposure is removed. The occupational exposure experience thus is inadequate to demonstrate the applicability of this threshold level to chronic, unmonitored exposure to the public. Furthermore, recent reviews (Liggett, 1989; Diamond, 1989; Diamond et al., 1989; Kathren and Weber, 1988) contain arguments that a lower threshold be used.

We assume nephrotoxicity to be a threshold effect. There are several reasons for the complexity, however, in characterizing an appropriate threshold level:

1. The first difficulty is the variety of qualitatively different effects. Wrenn et al. (1987) describe four classes or stages of effect: biochemical change, histopathological change, chronic poisoning, and acute poisoning. The Nuclear Regulatory Commission classifies effects less than lethality as permanent renal damage, transient renal injury, and no effect thresholds (McGuire, 1991). The severity of the effects produced increases with dose. A kidney concentration of 3 μg U /g has been used as a threshold of chemical toxicity (Rich et al., 1988). This level was believed to be sufficient to prevent serious damage in people who were occupationally exposed and who generally were monitored with routine bioassay of urine. Even early experiments indicated mild renal injury at lower kidney concentrations (Liggett, 1989). In a review of different kinds of effects, Diamond (1989) noted nephrotoxicity in animal studies at kidney concentrations below 1 μg U /g. There is some controversy over the conclusions from animal studies (Kathren and Weber, 1988). While some interpret the earlier studies as supporting the 3 μg/g threshold, others conclude that kidney damage was shown to occur in all species studied at levels of 0.5-1 μg U/g. At what point should an "effect" be considered toxicity?

2. Most studies focus on acute or short-term exposures. Few studies directly address long-term, chronic exposure of the kind likely to be associated with exposure to low levels of uranium in drinking water. Acute exposure studies indicate that the damage of low-level exposure is completely repaired. Replacement cells, however, may be of different character than the original cells, especially as the level of damage increases. One need not be overly concerned with minor, repairable damage from an acute exposure, but if the exposure is chronic, reliance on repair becomes problematic. We assume no repair.

3. The kidney has a large reserve capacity. Minor damage resulting from low-level exposure may not lead to degraded function but simply to a reduction in reserve. The effect of this reduction may appear only as a result of an independent challenge from another source. This has been suggested as a reason why no effect has been observed among thousands of people occupationally exposed under the limit of 3 μg U/g (Kathren and Weber,

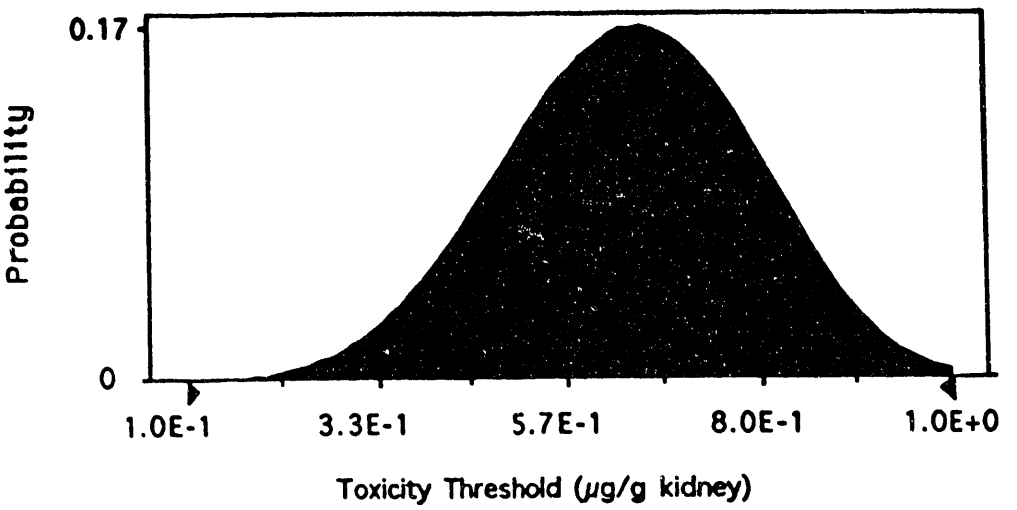
1988). Reducing reserve is clearly not as important as a direct and immediate reduction in function, but it is clearly an adverse effect.

4. Although it was assumed above that once in the bloodstream the original chemical form of uranium no longer mattered, it has been suggested that the uranium concentration required to cause an effect may depend on the initial chemical form of uranium (Liggett, 1989). Many of the experiments (but not all) showing low-level effects used fluoridated uranium compounds (e.g., UO_2F_2), which are of special concern for occupational inhalation of UF_6, but of less concern in drinking water. Without greater evidence, however, it seems better to retain the assumption that the chemical form affects only absorption into the bloodstream.

5. Most studies of kidney toxicity relate toxic effects directly to dose, described either as intake by mouth or breathing or injection. This ignores the pharmacokinetics discussed above and makes direct interspecies extrapolation difficult. In many cases kidney concentrations were calculated using the ICRP (1979) formula. Inconsistencies can arise when comparing reports of measured kidney concentration with reports in which kidney concentration has been estimated from intake.

Wrenn et al. (1985) suggest a threshold of 1 μg U/g. Kocher (1989) applied a safety factor of 10 to this to protect maximally exposed individuals in the public, using 0.1 μg U/g. Several reports of animal experiments demonstrate effects in the range 0.5-1 μg U/g. These effects are perhaps not as severe as those on which the original occupational standard of 3 μg U/g was based, but may be more appropriate end-points for chronic exposure to the public, whose members do not have the additional protection of routine medical surveillance and bioassay. A range of 0.1-1 μg U/g appears appropriate. The threshold value was characterized as a Weibull distribution with location parameter 0.1 to reflect a lower limit of 0.1 μg U/g, a scale parameter of 0.6 and shape parameter of 4. These produce a maximum value of about 1 μg U/g with a mean of 0.6 (Figure 6-5).

Figure 6-5. Distribution of threshold of effect of uranium in kidney.



6.3.3 Carcinogenicity

6.3.3.1 Evidence for Uranium Cancer Risk in Humans

There is only limited evidence for an increase in cancer risk associated with the ingestion of natural uranium. The available animal and epidemiological data have been reviewed in detail elsewhere (NRC, 1988; USEPA 1991b).

Most epidemiological studies of the effects of uranium are confounded by smoking or other etiologic agents, primarily radon. A few studies have been performed on uranium millers or workers in a uranium enrichment plant. Only one of these studies (Cookfair et al., 1983) had a positive finding, with the relative risk of lung cancer increasing with exposure.

The available epidemiological data do not allow the direct determination of a risk factor for chronic, low levels of uranium exposure. Some researchers have estimated the risk associated with exposure to uranium isotopes by analogy with radium. Mays et al. (1985) estimated a risk of bone sarcoma over a lifetime associated with a daily ingestion of 1 pCi uranium to be 1.5 bone sarcomas per million persons. This calculation assumed a linear dose-response relationship.

In this analysis, the dose and risk associated with chronic, low level exposure to uranium ingested in food and water was estimated using models and methods described by ICRP (ICRP 1979; ICRP 1991a; ICRP, 1991b).

6.3.3.2 Retention Function

Retention is the amount of a substance remaining in a tissue or organ at some time after uptake. The retention function used here is the toxicokinetic model developed by the ICRP (1979). Distribution and retention of uranium is modeled for three compartments: bone, kidney, and other soft tissues:

$$R_{bone}(t) = 0.2\, e^{-0.693t/20d} + 0.023\, e^{-0.693t/5000d}$$

$$R_{kidney}(t) = 0.12\, e^{-0.693t/6d} + 0.00052\, e^{-0.693t/1500d}$$

$$R_{other}(t) = 0.12\, e^{-0.693t/6d} + 0.00052\, e^{-0.693t/1500d}$$

where R_{bone}, R_{kidney} and R_{other} are the retention functions for bone, kidney and other tissues, respectively.

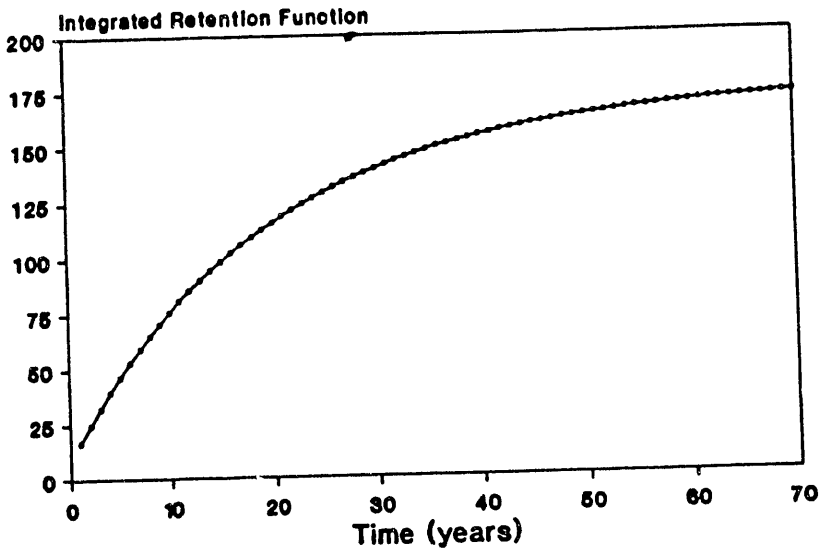
Of the uranium taken into the blood, fractions 0.2 and 0.023 go to mineral bone, with half-lives of 20 and 5,000 days, respectively (ICRP, 1979). Fractions 0.12 and 0.00052 go to kidney with half-lives of 6 and 1500 days, respectively (ICRP, 1979). Fractions 0.12 and 0.00052 go to all other tissues with half-lives of 6 and 1500 days, respectively (ICRP, 1979). The remainder of the uranium entering the transfer compartment (fraction: 0.536) is assumed to be directly excreted through the urinary pathway.

The whole-body retention function for uranium is thus:

$$R(t) = 0.24\, e^{-0.693t/6d} + 0.2\, e^{-0.693t/20d} + 0.00104\, e^{-0.693t/1500d} + 0.023\, e^{-0.693t/5000d} + 0.536\, e^{-0.693t/0.25d}$$

Figure 6-6 presents the integrated retention function for uranium for all tissue over time.

Figure 6-6. Integrated retention function for uranium in humans.



6.3.3.3 Effective Dose

Because uranium concentrates in bone and delivers a dose over time, we used the concept of the committed effective dose (ICRP, 1991a) to estimate the dose and risk associated with uranium ingested in food and water.

ICRP (1991b) gives ALI values (annual limits on intake) in units of Bq, for workers, assuming the dose is integrated over a working lifetime of 50 years. These ALI values are associated with a committed effective dose (E_{50}) of 0.02 Sv, and represent the limits for intake in one year. The values for uranium given in ICRP (1991b) are listed in Table 6-4.

The E_{50} values derived by the ICRP (ICRP, 1991b) were adjusted for a population with an average lifetime of 70 years. The retention ratio for a 70 year vs. a 50 year lifetime was calculated using the dominant retention function for uranium -- the retention function for bone. The retention function for bone (ICRP, 1979, see

section on retention function above) was integrated over 50 years to derive R(50) and 70 years to derive R(70). This calculation yields a R(50) of 1600, and an R(70) of 1700. The retention ratio is R(70)/R(50) = 1700/1600 = 1.1

The population 70 year committed effective dose (E_{70}) was then calculated by multiplying the 50 year worker committed effective dose (E_{70}) by 1.1 (Table 6-4). The committed effective dose (E_{50}, E_{70}) for total uranium (Table 6-4) was calculated assuming the natural percentages of U-234 (0.0055%), U-235 (0.72%) and U-238 (97.27%).

Table 6-4. ALIs and associated committed effective doses, E_{50}, E_{70} for uranium isotopes.

	ALI ($Bq \cdot yr^{-1}$)	E_{50} (Sv)	E_{70} (Sv)
U-234	7×10^{5}	2.86×10^{-8}	3.15×10^{-8}
U-235	7×10^{5}	2.86×10^{-8}	3.15×10^{-8}
U-238	8×10^{5}	2.50×10^{-8}	2.75×10^{-8}
Total U*	--	2.50×10^{-8}	2.75×10^{-8}

* assuming natural percentages of U-234 (0.0055%), U-235 (0.72%) and U-238 (97.27%)

6.3.3.4 Dose per Unit Intake

Table 6-5, column 2 gives the integration over time of the whole-body retention function for uranium. Based on the retention function, the fraction of the committed effective dose delivered each year from intake during year 1 was calculated (Table 6-5, column 3).

Table 6-5, columns 2 and 3 were used to calculate the fraction of the committed effective dose that will actually be received over a lifetime, for intake in each year of life (column 4). This calculation was based on the assumption of a 70-year lifetime, and allows the dose to be committed only up to 70 years, rather than an additional 70 years into the future. The fraction of the committed effective dose received for each year of life was calculated by subtracting the fraction of the committed effective dose that would be received in the years after age 70.

Finally, the effective dose delivered over a lifetime (70 years) for a unit intake (1 $Bq \cdot yr^{-1}$) in each year of life (Table 6-4, column 5) was calculated by multiplying the fraction of the committed effective dose for each intake year (column 4) by the committed effective dose for uranium estimated as described above (E_{70}, 2.75×10^{-8} Sv). We call this value the "lifetime modified" committed effective dose per unit intake. The values in Table 6-5 allow the calculation of the effective dose for any exposure level, exposure period and year of life exposed.

Table 6-5. Uranium retention function, fraction of committed effective dose delivered over time for intake at year 1 and effective dose (Sv) for a unit intake of uranium (1 Bq·yr^{-1}) in each year of a 70-year lifetime (based on f1 = 5%).

	URANIUM RETENTION FXN			E_{70} PER UNIT (1 Bq·yr^{-1}) INTAKE
year post intake	integ[1] ret fxn	fraction[2] E_{70} intake year 1 (Sv)	fraction[3] E_{70} intake each year (Sv)	lifetime[4] modified E_{70} (Sv)
1	16.58	0.0967	1.000	2.75E-08
2	24.06	0.0471	0.999	2.75E-08
3	32.30	0.0446	0.997	2.74E-08
4	39.55	0.0423	0.995	2.74E-08
5	46.41	0.0400	0.994	2.74E-08
6	52.92	0.0380	0.992	2.73E-08
7	59.09	0.0360	0.990	2.73E-08
8	64.94	0.0341	0.988	2.72E-08
9	70.49	0.0324	0.986	2.71E-08
10	75.76	0.0307	0.984	2.71E-08
11	80.76	0.0292	0.982	2.70E-08
12	85.51	0.0277	0.979	2.70E-08
13	90.02	0.0263	0.977	2.69E-08
14	94.30	0.0250	0.974	2.68E-08
15	98.36	0.0237	0.971	2.67E-08
16	102.22	0.0225	0.968	2.67E-08
17	105.89	0.0214	0.965	2.66E-08
18	109.37	0.0203	0.962	2.65E-08
19	112.68	0.0193	0.958	2.64E-08
20	115.83	0.0183	0.955	2.63E-08
21	118.82	0.0174	0.951	2.62E-08
22	121.66	0.0166	0.947	2.61E-08
23	124.36	0.0157	0.943	2.60E-08
24	126.92	0.0150	0.938	2.58E-08
25	129.36	0.0142	0.934	2.57E-08
26	131.67	0.0135	0.929	2.56E-08
27	133.87	0.0128	0.924	2.54E-08
28	135.97	0.0122	0.918	2.53E-08
29	137.96	0.0116	0.912	2.51E-08
30	139.85	0.0110	0.906	2.50E-08
31	141.64	0.0105	0.900	2.48E-08
32	143.35	0.0100	0.893	2.46E-08
33	144.97	0.0095	0.886	2.44E-08
34	146.52	0.0154	0.879	2.42E-08
35	147.98	0.0086	0.871	2.40E-08
36	149.38	0.0081	0.863	2.38E-08
37	150.70	0.0077	0.854	2.35E-08
38	151.96	0.0074	0.845	2.33E-08
39	153.16	0.0070	0.836	2.30E-08
40	154.30	0.0066	0.826	2.27E-08

Table 6-5 (cont).

	URANIUM RETENTION FXN			E_{70} PER UNIT (1 Bq·yr^{-1}) INTAKE
year post intake	integ[1] ret fxn	fraction[2] E_{70} intake year 1 (Sv)	fraction[3] E_{70} intake each year (Sv)	lifetime[4] modified E_{70} (Sv)
41	155.38	0.0063	0.815	2.24E-08
42	156.41	0.0060	0.804	2.21E-08
43	157.39	0.0057	0.793	2.18E-08
44	158.32	0.0054	0.780	2.15E-08
45	159.21	0.0052	0.768	2.11E-08
46	160.05	0.0049	0.754	2.08E-08
47	160.84	0.0047	0.740	2.04E-08
48	161.60	0.0044	0.725	2.00E-08
49	162.33	0.0042	0.709	1.95E-08
50	163.01	0.0040	0.693	1.91E-08
51	163.67	0.0038	0.675	1.86E-08
52	164.29	0.0036	0.657	1.81E-08
53	164.88	0.0034	0.638	1,76E-08
54	165.44	0.0033	0.617	1.70E-08
55	165.97	0.0031	0.596	1.64E-08
56	166.48	0.0030	0.573	1.58E-08
57	166.96	0.0028	0.550	1.51E-08
58	167.42	0.0027	0.525	1.44E-08
59	167.85	0.0025	0.498	1.37E-08
60	168.26	0.0024	0.471	1.30E-08
61	168.66	0.0023	0.441	1.22E-08
62	169.03	0.0022	0.411	1.13E-08
63	169.39	0.0021	0.378	1.04E-08
64	169.73	0.0020	0.344	9.48E-09
65	170.05	0.0019	0.308	8.49E-09
66	170.35	0.0018	0.270	7.44E-09
67	170.64	0.0017	0.230	6.34E-09
68	170.92	0.0016	0.188	5.18E-09
69	171.18	0.0015	0.143	3.95E-09
70	171.43	0.0015	0.097	2.65E-09

[1] integrated retention function for uranium
[2] fraction of committed effective dose delivered each year after intake in year 1
[3] fraction of committed effective dose received over a lifetime for intake in each year of life
[4] lifetime modified effective dose delivered over a lifetime for a unit intake (1 Bq·yr^{-1}) in each year of life

Gut Uptake Factor and Calculation of Effective Dose

Not all uranium taken into the body through ingestion is taken up through the gut (See section 6.3.2). The ICRP recommends a value of 5% for the gut uptake factor, f_1. The committed effective dose estimates presented in Table 6-5 are based on an f_1 of 5%.

An independent analysis of the available data suggested a distribution of f_1 values, described by a lognormal distribution, with a geometric mean of 1.03 and a

geometric standard deviation of 1.78 (section 6.3.2.4, see section 6.8.2). This distribution was used to describe f_1 for the calculation of dose and risk in this assessment. The lifetime modified committed effective dose given in Table 6-5 was multiplied by a correction factor equal to a sample from the distribution of the f_1 derived in section 6.3.2.4 divided by the ICRP f_1 (5%).

6.3.3.5 Risk Factor

ICRP (1991a) derived an "average" value of 10×10^{-2} deaths/Sv for the probability of fatal cancer associated with acute, high dose exposure. This value is based on estimates given by UNSCEAR (1988), BEIR V (NRC, 1990) and ICRP (1991a). ICRP (1991a) applied a dose and dose rate effectiveness factor of two, to obtain a nominal value of 5×10^{-2} deaths/Sv for the probability of fatal cancer in a population after low dose, low dose rate irradiation.

The BEIR V Committee (NRC, 1990) estimated the uncertainty associated with the risk factor for exposure to low dose radiation, contributed by three major factors. These factors are: statistical uncertainties caused by the small number of cases used in the risk models; uncertainty in the estimates of the dose rate effectiveness factor; and uncertainty contributed from external factors such as 1) extrapolating from a Japanese population, 2) dosimetry system and 3) model mis-specification. The geometric standard deviation for each of these factors was estimated as described below.

The BEIR V Committee (NRC, 1990) estimated that the 90% confidence limits for its estimates of the risk factor based only on sampling variation are 1,100 and 2,400 in 200,000 people. These values result in a geometric standard deviation of 1.3.

The second factor contributing to the total geometric standard deviation is the uncertainty in the estimate of the dose rate effectiveness factor (DREF). There are very little data available to support an estimate of the dose rate effectiveness factor, and most data on the DREF for tumorigenesis are from animal data (NRC 1990). BEIR V suggests a range of values (limited range, relevant studies) of 2 - 5 (NRC 1990, p. 23). If these values are assumed to represent the 90% confidence interval of a lognormal distribution, the calculated geometric standard deviation for this factor is 1.3.

The final factor contributing to overall uncertainty in the total geometric standard deviation (GSD) for the risk factor comes from what BEIR V (NRC, 1990) terms external factors. BEIR V estimated the GSD for females to be 1.27, and the GSD for males to be 1.31. We combined the GSD for males and females in quadrature to arrive at an overall estimate of the variance. This results in a combined GSD for external factors of 1.4. The method for combining these two GSDs is given in BEIR V (NRC, 1990):

$$GSD_{external} = \exp\sqrt{(\ln GSD_{male})^2 + (\ln GSD_{female})^2}$$

The total geometric standard deviation for the risk factor is calculated by combining in quadrature the GSD's estimated for each factor contributing to the uncertainty:

$$GSD_{risk} = \exp\sqrt{(\ln 1.3)^2 + (\ln 1.3)^2 + (\ln 1.4)^2} = 1.65$$

This assessment assumed that the risk factor for low dose radiation was lognormally distributed, with a geometric mean of 0.05 deaths/Sv and a geometric standard deviation of 1.65.

6.4 Identification of Scenarios and Receptors

6.4.1 Release Scenarios

Two release scenarios were considered in this assessment. The first scenario assumed there will be a continuous release of uranium to the ground water at current levels. This represents a baseline analysis and assumes no further control of runoff. The second scenario is based on the assumption that the runoff contributing to the uranium in ground water will be controlled in the next ten years and that no further uranium will be added to Paddy's Run or the ground water. This scenario is more realistic because the FEMP plans to eliminate additions of uranium to the plume. Neither scenario includes the remedial actions planned for the short-term corrective action discussed in section 6.1.3 and thus represent a baseline for the south plume Engineering Evaluation/Cost Analysis (EE/CA). The risks associated with background concentrations of uranium in ground water were also estimated.

6.4.2 Receptors

Important receptors are current and future users of ground water in the area potentially impacted by the south plume. Potential receptors include residential or agricultural wells in the path of the plume, and possible future wells in the potentially impacted area. Figure 6-7 gives the location of currently located potential users of ground water south of the FEMP.

Three sets of potential receptors were identified. They include five representative residential wells located south of the facility, four wells located along the centerline of the developing uranium plume, and potentially impacted future wells which could be located anywhere in the area just south of the FEMP. These receptors are discussed in more detail in section 6.5.2.4.

6.4.3 Exposure Scenarios

The exposure scenarios considered in this assessment are those associated with the use of ground water south of the FEMP. Potential uses of ground water include ingestion of drinking water, irrigation of crops, and watering of dairy cattle. Important exposure pathways include direct ingestion in water, ingestion in homegrown fruits and vegetables, and ingestion in homeproduced milk. Uranium is not easily taken up through the skin or inhaled in the shower, and these pathways were not included in this assessment.

The FEMP is located in a rural area, and it was assumed that residents produce their own fruits, vegetables, and milk. Irrigation of commercial crops is not widely practiced in Hamilton and Butler counties (IT, 1991b). It was assumed that family gardens are irrigated, but that feed and forage for dairy cows are not. Some portion of the intake of fruits and vegetables was assumed to be homegrown, and it was assumed that all milk was home produced, but that other dairy products were not.

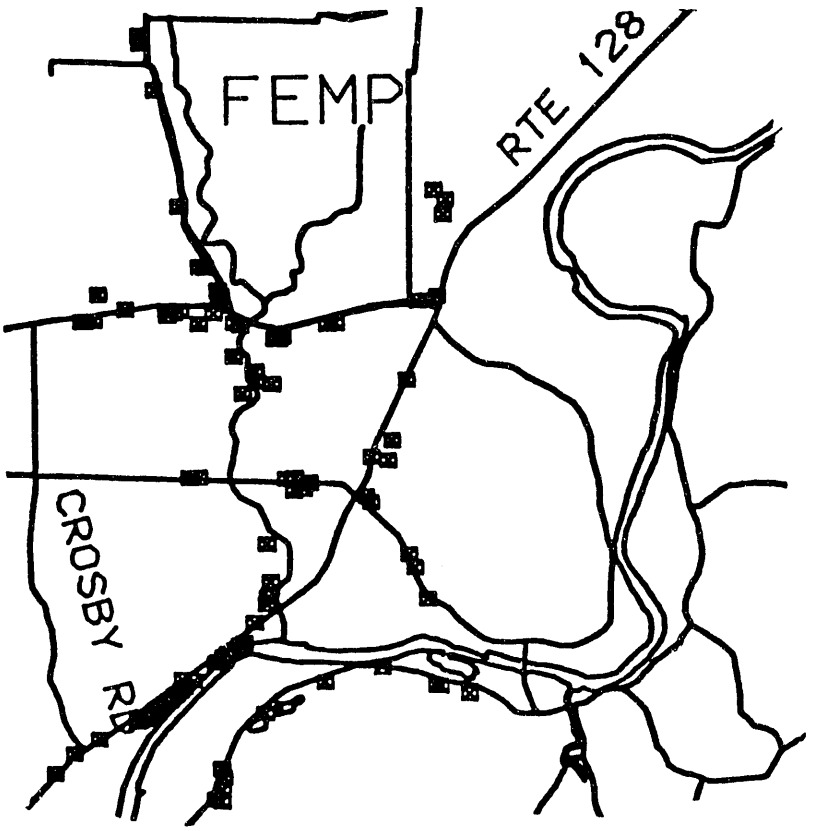
Exposures and risks were predicted 70 years into the future (year 0 = 1989). The impact of the predicted plume on the Great Miami River was not included in this assessment. Loadings to the river from the south plume are likely to be small because of the dilution achieved as the plume moves south. The river flow south of the facility is quite high, and any uranium entering the river will be quickly diluted. The exposure route associated with the ingestion of uranium in fish caught in the Great Miami River will be included in a second risk assessment study for uranium at the FEMP.

6.4.4 Incremental vs. Total Exposure and Risk

The uranium cancer risk assessment was based on an assessment of the incremental cancer risk associated with uranium in the ground water plume emanating from the FEMP. For the cancer risk assessment, uranium concentrations in drinking water, homegrown food and homeproduced milk were estimated based only on predicted uranium concentrations associated with the plume. Exposures and risks associated with natural background uranium concentrations in water, soil or vegetation, or uranium in soil and vegetation associated with historical airborne uranium releases were not included in the cancer risk assessment.

Uranium toxicity, however, was assumed to be a threshold effect, and the toxicity assessment required that other sources of uranium exposure through ingestion be considered in the analysis. Additional sources of uranium include natural background concentrations in water, milk, soil and vegetation; uranium in soil, milk and vegetation associated with the facility, and uranium in the diet from sources other than home gardens.

Figure 6-7. Currently located potential receptors for uranium in ground water



6.5 Uranium Concentrations in Ground Water

6.5.1 Natural Background Concentrations

Uranium occurs naturally in ground water near the FEMP. USDOE (1990c) refers to a study of data collected from private wells north of the site (Varchol, 1990). This study concluded that background concentrations of uranium in ground water range from about 0.068 to 2.0 pCi/L (0.0025 to 0.074 Bq/L; 0.1 to 2.94 μg/L).

Uranium measured in offsite wells near the facility (Figure 6-8) can be used to describe the largest value for background uranium concentrations -- natural background may be smaller than these values but is not likely to be larger because these samples may have been impacted by the facility. Average concentrations of total uranium in offsite wells in 1989 are given in Table 6-6. Wells 12, 15 and 17 are obviously impacted (1990 concentration of 27 - 190 pCi/L; 1.0 - 7.0 Bq/L). Average background concentrations of uranium in ground water appear to be in the range of about 0.093 to 1.2 pCi/L (0.0034 to 0.044 Bq/L, 0.14 to 1.76 μg/L). This assessment used the average uranium concentration measured in offsite wells (excluding wells 12, 15 and 17) as an estimate of background concentrations near the FEMP (0.63 pCi/L, 0.023 Bq/L, 0.926 μg/L).

Figure 6-8. Offsite wells (from USDOE, 1990c).

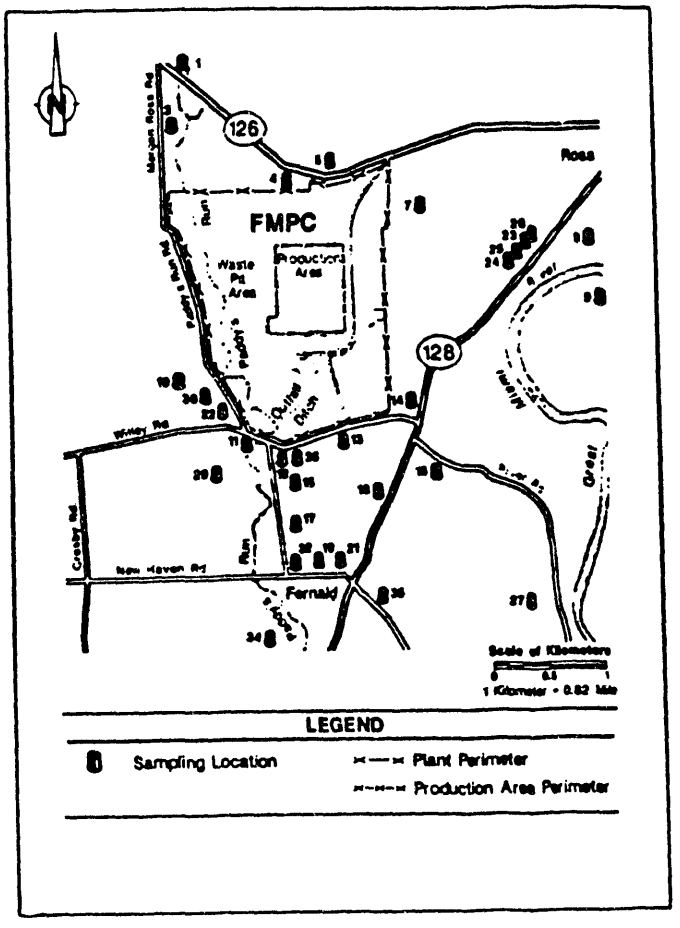


Table 6-6. Concentration of uranium in offsite wells, 1989.

Well	Average Concentration in Water		
	(pCi/L)	(Bq/L)	(μg/L)
1	0.17	0.01	0.25
3	0.17	0.01	0.25
4	1.4	0.05	2.06
5	1.5	0.06	2.21
7	1.1	0.04	1.62
8	0.6	0.02	0.88
9	1.0	0.04	1.47
10	0.52	0.02	0.76
11	1.1	0.04	1.62
12	170	6.29	250.0
13	0.37	0.01	0.54
14	0.88	0.03	1.29
15	190	7.03	279.4
16	0.52	0.02	0.76
17	27	1.00	39.7
18	0.34	0.01	0.50
19	0.12	0.00	0.18
21	0.25	0.01	0.37
22	0.79	0.03	1.16
23	0.58	0.02	0.85
24	0.4	0.01	0.59
25	0.27	0.01	0.40
26	0.14	0.01	0.21
27	0.5	0.02	0.74
28	0.57	0.02	0.84
29	1.1	0.04	1.62
30	0.38	0.01	0.56
32	0.093	0.00	0.14
34	0.83	0.03	1.22
35	1.2	0.04	1.76

6.5.2 Prediction of Uranium Concentrations in the South Plume

6.5.2.1 Introduction

A three-dimensional flow and transport model of the ground water at the FEMP is being developed in support of the ongoing Remedial Investigation /Feasibility Studies (RI/FS). The modeling work is being performed for the U.S. Department of Energy by IT Corporation.

The model is complex and is still being fine-tuned for use at the FEMP. A preliminary set of results was made available by IT Corporation and the USDOE, and these predictions were used in this analysis. The model and model predictions were reviewed by an independent consultant and found to adequately describe flow and transport at the facility.

There are several limitations to the model in terms of its application to a risk assessment study. The transport model domain is limited and does not extend to the Great Miami River South of the FEMP. This limitation prevents the estimation of uranium loading to the river and an analysis of exposure to uranium ingested in fish. The second limitation is the time over which model predictions are available. This analysis was based on model predictions of uranium concentrations in ground water 70 years into the future.

These limitations were not considered serious in the current assessment of risk associated with the ingestion of uranium in ground water. Predictions of uranium transport 70 years into the future describe the highest potential exposure to both currently located wells and to potential future wells. Loadings to the river from the south plume are likely to be small because of the dilution achieved as the plume moves South. The river flow south of the facility is quite high, and any uranium entering the river will be quickly diluted. The exposure route associated with the ingestion of uranium in fish caught in the Great Miami River will be included in a future assessment for uranium at the FEMP.

As the model is fine-tuned, the model domain expanded, and predictions made further into the future, the risk assessment will be updated.

6.5.2.2 Ground Water Flow and Transport Model

Modeling Approach

A comprehensive effort was undertaken to define the ground-water flow system and the transport of dissolved uranium at the FEMP. SWIFT III was selected as the code for use in the hydrogeologic investigation of the FEMP site-wide RI/FS.

The SWIFT (Sandia Waste Isolation, Flow and Transport) code was originally developed by Sandia National Laboratory for use in the high-level nuclear waste isolation program. Modifications to the original code (SWIFT II and SWIFT III) are documented in Reeves (1985) and Geotrans (1988).

SWIFT III is a fully transient, three-dimensional, block centered, finite difference code. The model can solve the ground water flow, solute transport, heat transport and density dependent flow equations. These equations, as programmed in the code, also account for water-table conditions, attenuation/retardation processes, decay chains, adsorption, and temperature or concentration effects on fluid viscosity.

The simulation of ground-water transport of uranium at the FEMP is a product of both analytical and numerical (two- and three-dimensional) modeling efforts. Only the numerical three-dimensional regional flow model and the numerical three-dimensional local solute transport model are discussed here. The other models were constructed and run as part of the effort to determine hydraulic parameters and initial conditions to be used in the three-dimensional models and to facilitate sensitivity analysis.

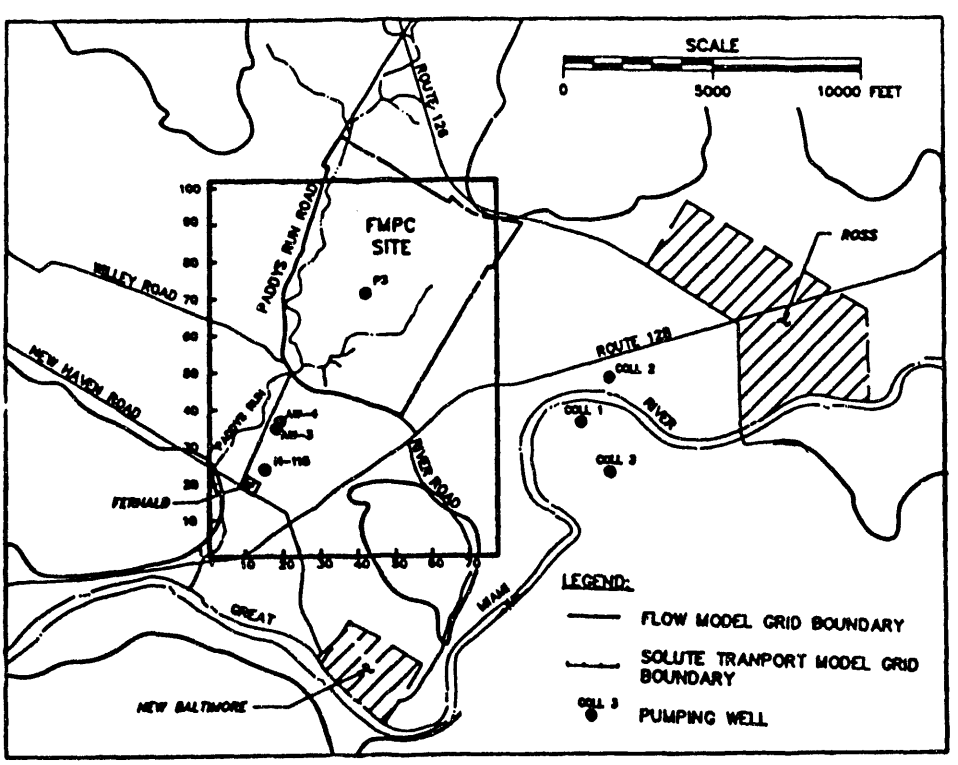
The description of the modeling effort given here was extracted from reports prepared by IT Corporation for the United States Department of Energy (USDOE, 1990b, USDOE 1990d).

Flow Model -- Model Construction and Input Parameters

The flow model grid and boundaries include the main collector wells, the Great Miami River and important features of the bedrock valleys. The boundaries are considered to be outside the area of influence of any pumping wells that affect ground-water flow within the site and/or plume area. The model grid covers approximately 29 square miles and is shown in Figure 6-9.

The model was vertically discretized into five layers which represent the full thickness of the Great Miami Aquifer. The top layer, Layer 1, represents the water table. Layer 2 represents the lower part of the upper aquifer, and overlies the clay interbed. Model layer 3 represents the clay unit beneath the FEMP and, where the clay is absent, is considered to be a sand and gravel unit. Layers 4 and 5 represent the lower part of the aquifer and overlie the bedrock surface. The bedrock, which is considered to be impermeable, is the lower boundary of the model.

Figure 6-9. Flow and transport model boundaries (from USDOE, 1990b).



Horizontal hydraulic conductivity for the upper three layers was typically 450 ft/d, while 600 ft/d was more representative for the lower two model layers. Conductivity of the clay interbed beneath the site was approximately 0.0003 ft/d. Several local areas, along bedrock valley walls, in layers 1 through 4 had conductivities as low as 60 ft/d. The ratio of horizontal to vertical conductivity was 10:1 throughout the model.

Recharge rates varied from 6.0 in/yr north and west of the site, to 14.0 in/yr and 6.0 in/yr to the south and east. The recharge rate in the vicinity of the site was 2.0 in/yr. The flow model was calibrated to April 1986 ground-water conditions, assuming that the ground water system was in equilibrium at that time.

Transport Model -- Model Construction and Input Parameters

A local three-dimensional solute transport model was developed for the FEMP site and downgradient areas. The horizontal discretization scheme for the transport model was more detailed than that of the flow model (Figure 6-9). Each cell was 125 feet by 125 feet. The vertical representation of the ground-water system was the same as the flow model.

The distribution of recharge was similar to the flow model except in the area of Paddy's Run. A recharge rate of 32.0 in/yr was applied over reaches of Paddy's Run in the local transport model, based on the assumption that a potential for high infiltration exists in the stream channel.

Source Loading

Initial loading rates and time periods were taken from literature reviews of the site's operation and estimates of the rates of contaminant escape. Four distinct time periods were identified during which different source loading rates existed. The four periods (1952-1958,1959-1966,1967-1975,1976-1988) represent the various operational times of the waste pits over a 37 year period. Although loading rates and loading areas were varied during calibration, these four time periods did not change. The final calibration run included a fifth loading period representing the year 1989.

After extensive literature review, it was determined that six potential source areas existed. Those areas chosen for inclusion in the model were the Storm Sewer Outfall Ditch area (including the fly ash piles and Southfield), Paddy's Run downstream of the Storm Sewer Outfall Ditch, Paddy's Run between the Storm Sewer Outfall Ditch and the Waste Storage Area, the Waste Pit Area, the Production Area, and the Silos Area.

Model Calibration

The model was calibrated by dividing model time into source loading periods, introducing reasonable initial estimates of uranium source loading for each source cell, and establishing the best initial values for longitudinal and transverse dispersivity and the distribution coefficient for uranium. Adjustments were then made in source loading, source loading periods, dispersivities, and the distribution coefficient until concentrations calculated by the model were close to those measured in the field.

The preferred calibrated value for the distribution coefficients was 0.022 ft^3/lb which corresponds to a retardation factor of 12. The associated longitudinal and transverse dispersivities were 100 feet and 10 feet, respectively.

Calibrated total loading rates for all source areas combined were 0.552, 0.986, 0.149, 0.571, and 0.318 (lb/day) for the five loading periods respectively. The two areas delineated as the highest contributing source areas during the 1952 to 1989 time period were along Paddy's Run from the Waste Storage Area to the Storm Sewer Outfall Ditch and from the Storm Sewer Outfall Ditch to the Albright-Wilson Company wells.

6.5.2.3 Simulations and Model Predictions

Two predictive simulations were run for a period of 70 years each, assuming that the ground-water flow field would not be stressed or changed from existing conditions (1989). All hydraulic parameters and coefficients were those used in the final calibrated transport simulation. The initial condition for the uranium distribution was the same as the model generated distribution from the final transport calibration run. Only predicted concentrations from Model Layer 1 are presented here. Although part of the plume migrates within lower model layers, the highest observed concentrations have occurred in the upper part of the aquifer. In the analysis presented here, it was assumed that private wells serving houses are or will be located in the upper part of the aquifer, represented by Model Layer 1.

Source Removed

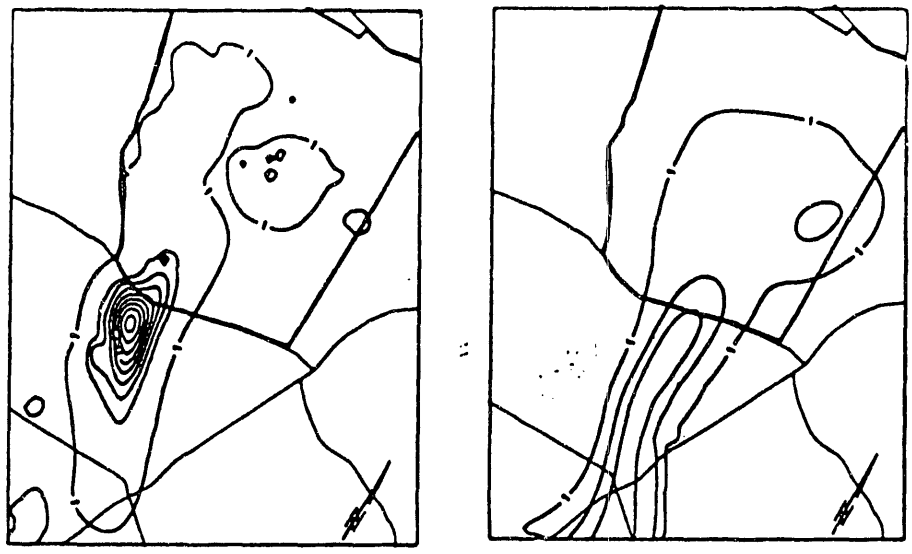
The first predictive simulation was run assuming that after ten years, all sources of uranium were removed. Contours of predicted uranium concentrations in ground water at time zero and 70 years are given in Figure 6-10. Because the concentrations in the plume rise so rapidly, it is difficult to show the values associated with each contour interval in Figure 6-10, and the reader is referred to the figure captions which give the contour ranges and intervals. Over the simulated seventy year period, the predicted plume concentrations continually diminish as the plume dissipates and flows beyond the boundary of the local model area. At the end of the simulation period, the majority of the plume area exhibits concentrations of under 10 $\mu g/L$.

Continuing Source

The second simulation assumed that a continuing source of uranium existed and that loading to the aquifer occurred at the same source locations and rates as defined during the last period (1989) of the final calibration run.

Contours of predicted uranium concentrations in ground water at time zero and 70 years are given in Figure 6-11. Because the concentrations in the plume rise so rapidly, it is difficult to show the values associated with each contour interval in Figure 6-11, and the reader is referred to the figure captions which give the contour ranges and intervals over the seventy year period simulated. Predicted uranium concentrations in the original plume (1989 conditions) continually diminish as the plume dissipates and moves beyond the boundary of the local model (Figure 6-11). At the same time, a second plume develops from an area designated as a 1989 source loading area (primarily the storm sewer outfall ditch). This indicates that the highest concentrations found within the observed plume did not originate at one of

Figure 6-10. Predicted uranium concentrations (μg/l), source removed, initial conditions and 70 years.



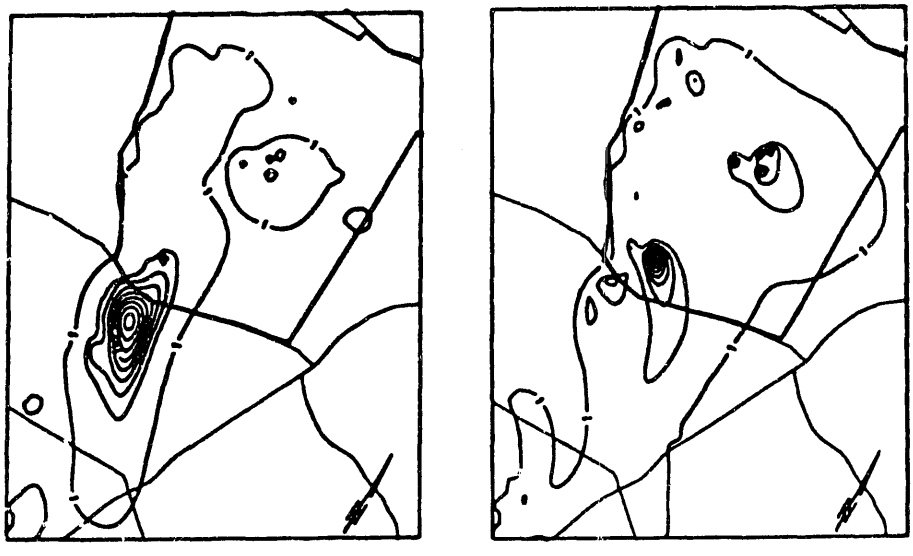
Initial Conditions

(0-420 ug/l; contour interval=50)

70 Years

(0-15 ug/l; contour interval=5)

Figure 6-11. Predicted uranium concentrations (μg/l), source continued, initial conditions and 70 years.



Initial Conditions

(0-420 ug/l; contour interval=50)

70 Years

(0-463 ug/l; contour interval=20)

the locations designated as a source area (storm sewer outfall ditch) for the seventy year predictive simulation. It also reiterates a conclusion drawn during previous efforts, that the highest present concentrations are not found at any of the designated source loading areas for the 70 year simulation. This is due to higher loading rates which occurred in the past, which are now manifest as the plume core downgradient from the source loading areas. The highest concentrations at the continuing source area (storm sewer outfall ditch) are approximately 460 μg/L.

6.5.2.4 Predicted Concentrations at Receptor Locations

The model output available for this assessment included predictions of uranium concentrations at identified receptors for model years 0 (1989), 10, 20, 30, 40, 50, 60 and 70. The assessment of uranium toxicity was performed for the concentrations in water, food and milk predicted for these model years.

Cancer risk was estimated over a lifetime. Because uranium is deposited in bone it delivers its dose over a period of years after the initial intake (see section 6.3.3). Exposures had to be estimated on a yearly basis to allow a reasonable estimate of the effective dose received over a lifetime.

For predictions at individual well locations, a simple interpolation approach was used to estimate the concentrations of uranium in ground water for the years between those simulated by the ground water model. The predicted concentration at year 5 was estimated by averaging the concentrations predicted for year 0 and year 10. Years 1-4 were estimated by averaging the predicted concentrations for year 0 and year 5. Years 6-9 were estimated by averaging the predicted concentrations for year 5 and year 10. This approach was extended to all 70 years of model predictions, and applied to all identified receptors.

Representative Residential Wells

Five wells were located to represent single residential wells or clusters of residential wells currently located downgradient of the FEMP (Figure 6-12). These representative wells were located to allow an estimate of uranium concentrations (and exposure) over time in currently located residential wells.

Predicted uranium concentrations in Model Layer 1 for the stop source and continue source scenarios at the 5 "representative" well locations for time 0, 10, 20, 30, 40, 50, 60 and 70 years are shown in Figure 6-13 and are given in Appendix B (Table B-1).

Centerline wells

Four wells were located along the approximate centerline of the developing plume to represent potential receptors with worst-case exposures. These "centerline" locations are approximately evenly spaced along the centerline of the developing plume. The location of these centerline wells are shown in Figure 6-12.

The concentration of uranium was predicted for the four centerline receptor locations for 70 years in increments of 10 years. Concentrations over time are given in Figure 6-14 (and Table B-2 in Appendix B) for both the continuing source simulation and the removal of source simulation. Inspection of the predicted concentrations shows that both the existing plume and the developing plume impact certain receptor locations. In the continuing source case, at location #3 (Figure 6-

14), concentration levels, which rise and fall over the first half of the simulation, reflect the migration of the existing plume even though the receptor location was chosen to monitor the developing plume.

Figure 6-12. Representative and centerline receptor wells.

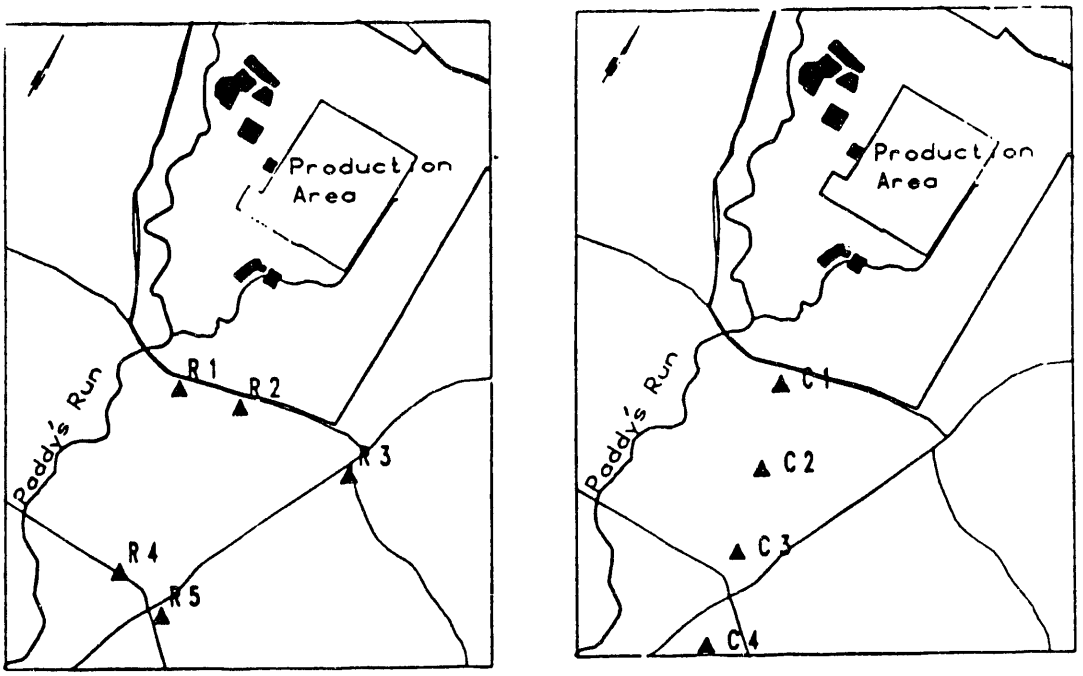


Figure 6-13. Predicted uranium concentrations (μg/l) over time at five representative well locations.

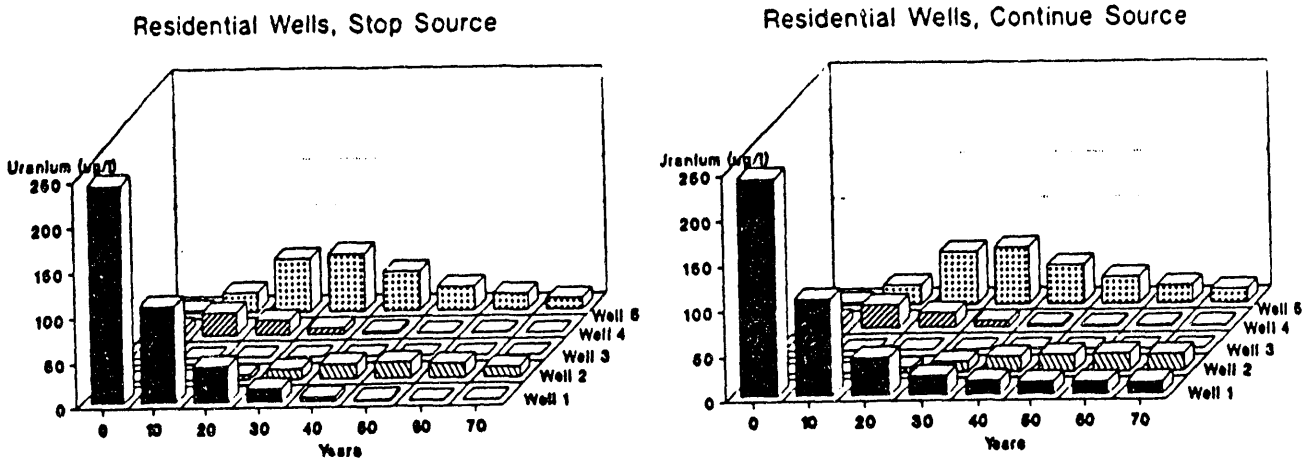
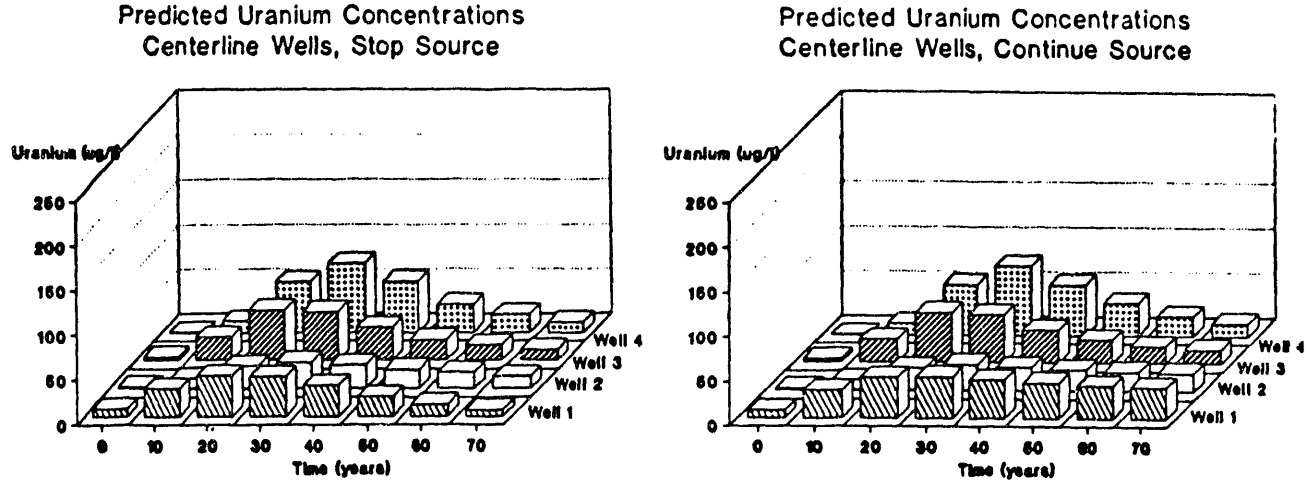


Figure 6-14. Predicted uranium concentrations (μg/l) over time at four centerline wells.



Potential Wells

The area covered by the transport model domain (Figure 6-15) was examined to identify grid cells (125 x 125 feet) that have the potential of being impacted by the south plume. These grid cells are those downgradient of the facility, excluding grid cells located above bedrock or west of Paddy's Run (Figure 6-15) and total 2179 cells. Each grid cell was assumed to represent a potential future well location.

The average uranium concentration predicted for each ten-year interval (0-10, 10-20, 20-30, 30-40, 40-50, 50-60 and 60-70 years) was calculated for each grid cell. For each ten-year period, the number of cells (representing potential future wells) in the concentration ranges 0-1, 1-20, 20-100, 100-300 and 300-500 $\mu g/L$ were counted. These counts give an estimate of the probability of locating a well in areas associated with high uranium concentrations. These probabilities are presented in Figure 6-16 for both the stop source and continue source simulation scenarios.

Figure 6-15. Grid cells in model domain representing potentially impacted wells.

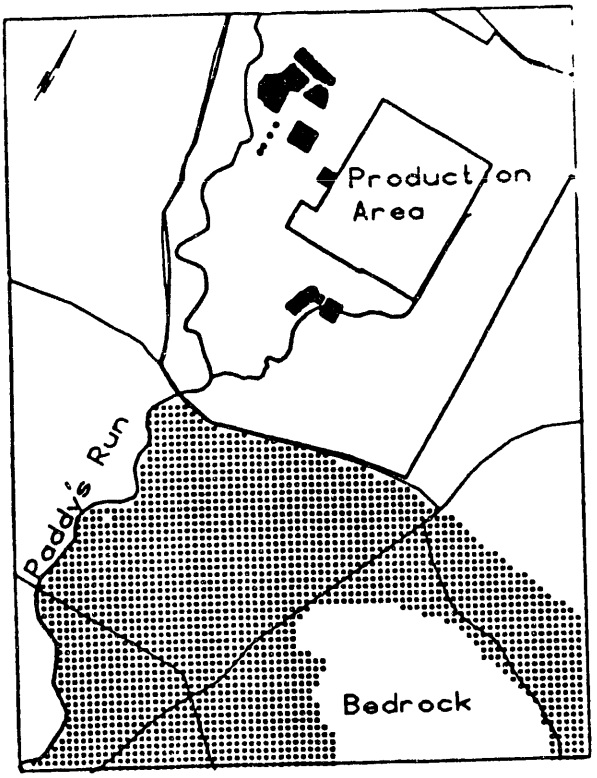
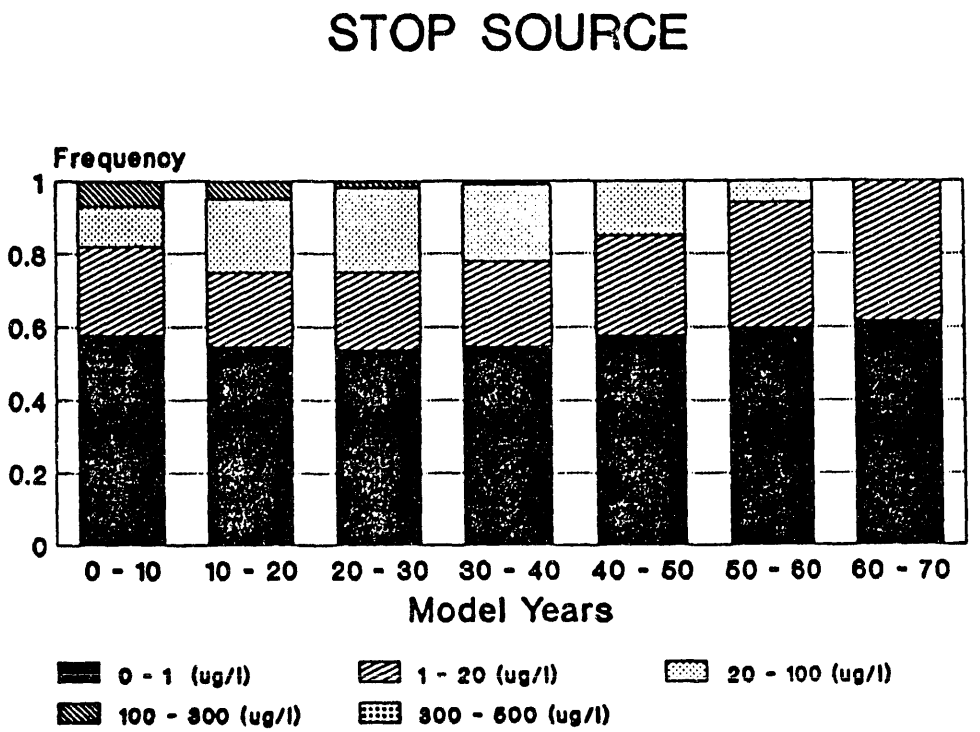
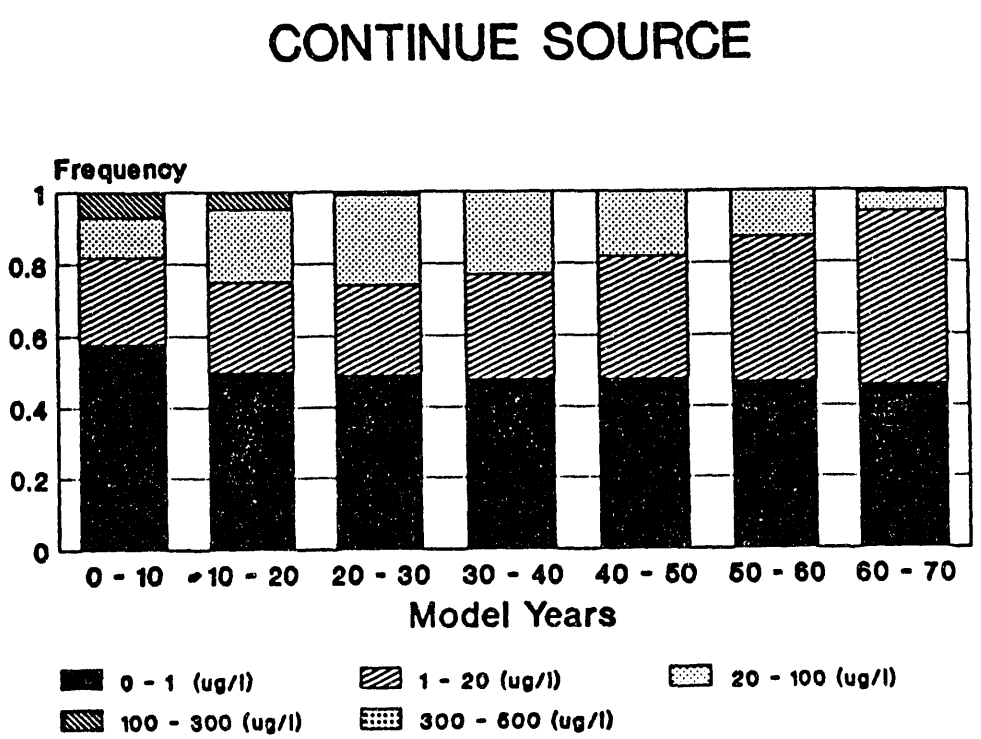


Figure 6-16. Model predictions: probability of locating a well in ground water with specified range of uranium concentrations (μg/L) for each year predicted by the model (note that the frequency of low concentration grid cells (0-1 μg/L) for the continue source scenario is smaller than for the stop source scenario because the frequency of higher concentration grid cells has increased).





6.5.3 Model Uncertainty

Because of the complexity of the ground water transport model, predictions of uranium concentrations in ground water were produced in a deterministic way. Single estimates of the uranium concentrations at each grid cell and model time period were produced in the simulations. The estimates of uranium concentrations in homegrown food and homeproduced milk described in the following section were based on these single deterministic estimates.

The importance of the uncertainty in the estimates of the ground water transport model was examined for toxicity effects in Section 6.8 and for cancer risk in Section 6.9. In this uncertainty analysis, the ground water model was assumed to have an uncertainty of less than 1.5. This was included in the uncertainty analysis as a lognormal distribution with a geometric mean of 1 and a 95% range of +/- 1,5 times, leading to a geometric standard deviation of 1.2 (arithmetic mean = 1.02 and standard deviation = 0.19).

6.6 Uranium Concentrations in Food and Milk

Concentrations of uranium in homegrown food and milk were estimated assuming irrigation with water containing uranium concentrations predicted as described in Section 6-5. For the assessment of toxicity risks, average background concentrations in water (0.926 μg/L, section 6.5.1) were added to the concentrations predicted by the model, and uranium concentrations in vegetation and in soil near the facility were assumed to be available to cows. The cancer risk analysis was based on an assessment of the incremental risk associated only with uranium in ground water associated with the FEMP.

The concentration of uranium in homegrown food and milk was calculated using equations given in USNRC (1977) (sections 6.6.1 and 6.6.2). Many of the parameters used in these calculations were drawn from a position paper prepared by IT (1991b) in support of the Remedial Investigation/Feasibility Study at the FEMP. The values of the parameters used to calculate the concentrations of uranium in homeproduced food and milk are documented in section 6.6.3. Resulting concentration estimates for all identified receptors are given in section 6.6.4.

6.6.1 Calculation of Uranium Concentrations in Homegrown Food

The concentration of uranium in homegrown food was calculated using the model described in USNRC Regulatory Guide 1.109 of the United States Nuclear Regulatory Commission (USNRC, 1977). Uranium in fruits and vegetables irrigated with contaminated ground water results from deposition onto plant foliage and uptake from the soil. The concentration of uranium in the edible portions of homegrown fruits and vegetables was calculated using the following equation (USNRC, 1977).

$$d_i = C_W \times IR$$

$$\lambda_{Ei} = \lambda_i + \lambda_w$$

$$C_{iv} = d_i \left[\frac{r[1-\exp(-\lambda_{Ei} t_e)]}{Y_v \lambda_{Ei}} + \frac{f_i B_{iv}[1-\exp(-\lambda_i t_b)]}{P\lambda_i} \right] \exp(-\lambda_i t_h)$$

where:

C_W = concentration of uranium in irrigation water (mg/L)
IR = average irrigation rate ($l/m^2/hr$)
d_i = deposition rate ($mg/m^2/hr$)
λ_{Ei} = effective removal rate constant for the crop (hr^{-1})
λ_i = uranium radioactive decay constant (hr^{-1})
λ_w = removal rate constant for loss by weathering (hr^{-1})
C_{iv} = concentration of uranium in the edible portion of the crop (mg/kg wet weight)
r = fraction of deposited activity retained on crops
t_e = period of crop exposure during growing season (hrs)
Y_v = agricultural productivity (kg/m^2, plant wet weight)
f_i = fraction of the year crops are irrigated
B_{iv} = concentration factor for uptake of uranium from soil by edible parts of crops (mg/kg wet weight per mg/kg dry soil)
t_b = period of time soil is exposed to contaminated water (hrs)
t_h = holdup time - interval between harvest and consumption (hrs).
P = effective surface density for soil ($kg(dry\ soil)/m^2$)

6.6.2 Calculation of Uranium Concentrations in Homeproduced Milk

Uranium in homeproduced milk can come from uranium ingested by dairy cattle in drinking water, forage and feed, and soil. Irrigation of farmland is not widely practiced near the FEMP (IT, 1991b) and it was assumed that dairy cattle did not ingest forage or feed grown with contaminated irrigation water. It was assumed that dairy cows were fed entirely with locally grown feed and forage. Cows may also ingest a significant amount of uranium in soil near the FEMP. Uranium in soil and vegetation was assumed to be available to cows only for the toxicity assessment.

The concentration of uranium in homeproduced milk was calculated using equations presented in USNRC (1977). The equation calculates the concentration of uranium in milk resulting from the ingestion of uranium in contaminated feed, water and soil:

$$C_{im} = F_{im}[(C_W)(Q_{mW}) + (C_F)(Q_{mF}) + (C_S)(Q_{mS})]$$

where:

C_{im} = uranium concentration in milk (mg/L)
F_{im} = stable element transfer coefficient to milk (day/L)
C_W = concentration of uranium in water (mg/L)
Q_{mW} = consumption rate of contaminated water by an animal (l/day).
C_F = concentration of uranium in feed (mg/kg dry weight)
Q_{mF} = consumption rate of contaminated feed by an animal (kg/day dry weight).
C_S = concentration of uranium in soil (mg/kg)
Q_{mS} = consumption rate of soil by livestock (kg/day)

6.6.3 Values Used in Calculation of Uranium Concentrations in Homeproduced Food and Milk

Table 6-7 summarizes the values used in the calculation of uranium concentrations in homeproduced food and milk. Each value and its source is discussed in the following sections. Values of C_W (concentration of uranium in irrigation water) were estimated in Section 6.4 for each identified receptor and simulated time period (0-70 years).

Table 6-7. Parameters used to calculate the concentration of uranium in homeproduced food and milk.

Parameter	Value	Reference
Uranium Physical and Transfer Parameters		
λ_i (U-234)	3.23E-10 (hr^{-1})	BRH, 1970
λ_i (U-235)	1.12E-13 (hr^{-1})	BRH, 1970
λ_i (U-238)	1.77E-14 (hr^{-1})	BRH, 1970
B_{iv}	0.0025	NCRP (1984)
F_{im}	0.0006 (day/L)	NCRP (1984)
Agricultural Parameters		
IR	0.08 ($l/m^2/hr$)	IT (1991b)
r	0.25	USNRC (1977)
λ_w	0.0021 (hr^{-1})	USNRC (1977)
f_i	0.38	IT (1991b)
t_b	131000 (hrs)	USNRC (1977)
P	225 (kg/m^2)	IT (1991b), USDA(1979)
t_e	1440 (hrs)	USNRC (1977)
Y_v	1.5 (kg/m^2)	IT (1991b), USDA (1979)
t_h	24 (hrs)	USNRC (1977)
Animal Consumption Parameters		
Q_{mW}	60 (l/day)	USNRC (1977)
Q_{mF}	16 (kg/day dry wt)	Shor and Fields (1979)
Q_{mS}	0.5 (kg/day)	Zack and Mayoh (1984)
Uranium Concentration in Soil and Grass		
C_S (toxicity)	6.91 (mg/kg dry weight)	Section 6.6.3
C_F (toxicity)	0.054 (mg/kg dry weight)	Section 6.6.3
C_S (cancer)	0 (mg/kg dry weight)	Section 6.6.3
C_F (cancer)	0 (mg/kg dry weight)	Section 6.6.3

Uranium Physical and Transfer Parameters

The decay constants (λ_i) for the principal isotopes of uranium (U-234, U-235 and U-238) are 3.23E-10 (hr^{-1}), 1.12E-13 (hr^{-1}) and 1.77E-14 (hr^{-1}), respectively. These decay constants correspond to half-lives of 2.47E+5 years for U-234, 7.1E+8 years for U-235 and 4.51E+9 years for U-238 (BRH, 1970).

The concentration factor for uptake of uranium from soil by the edible parts of crops (B_{iv}) ranges from 2.9×10^{-4} to 2.5×10^{-3} (mg/kg wet weight per mg/kg dry soil) (NCRP, 1984). This assessment used the more conservative default value of 2.5×10^{-3} recommended in NCRP (1984). Suggested values for the transfer factor coefficient of uranium to milk range from 1.2×10^{-4} to 6.0×10^{-4} (NCRP, 1984). This assessment used the more conservative value of 6.0×10^{-4}.

Agricultural Parameters

The average irrigation rate (IR) for the area near the FEMP was calculated in IT (1991b) based on the water needs for a corn crop. This value is 0.08 ($l/m^2/hr$). The fraction of the year crops are irrigated (f_i) was estimated to be 0.38 (IT, 1991b) based on a growing season in Hamilton County of 138 days (USDA, 1970).

The fraction of deposited activity retained on crops (r) was assumed to be 0.25 (USNRC, 1977), and the removal rate constant for loss by weathering (λ_w (hr^{-1})) was 0.0021 (hr^{-1}) (USNRC, 1977).

The period of time the soil is exposed to contaminated water was assumed to 131000 hours (15 years) (USNRC, 1977). The effective surface density for soil near the FEMP was assumed to be 225 kg/m^2 (IT, 1991b from USDA, 1979), and the period of crop exposure during growing season t_e (hrs) was 1440 hours (USNRC, 1977). Agricultural productivity Y_v for crops ingested by man was assumed to be 1.5 kg/m^2 wet weight (IT, 1991, USDA, 1979) and the holdup time or interval between harvest and consumption of homegrown crops was 24 hours (USNRC, 1977).

Animal Consumption Parameters

The consumption rate of contaminated water by a dairy cow (Q_{mW}) was assumed to be 60 l/day (USNRC, 1977). The consumption of feed and forage by dairy cattle was assumed to be 16 kg/day dry weight (Shor and Fields, 1979), and the consumption rate of soil by livestock (Q_S) was 0.5 (kg/day) (Zack and Mayoh, 1984).

Uranium Concentration in Soil and Vegetation

Cancer Risk - For the assessment of uranium cancer risk, it was assumed that no uranium was available to cows in feed or soil. All uranium intake (and cancer risk) was associated with uranium in ground water originating at the FEMP and was calculated as described in section 6.6.1 and 6.6.2.

Toxicity - For the assessment of uranium toxicity, the concentration of uranium in soil (C_S) and vegetation (C_F) near the FEMP was estimated from data published in the Annual Environmental Monitoring Report for Calendar Year 1989 (USDOE, 1990c). Some of the uranium measured in soil and vegetation samples

taken near the FEMP is associated with natural background levels of uranium, and some is associated with airborne releases from the facility.

Figure 6-17 shows the location of stations sampled for uranium in soil and vegetation in 1989. Uranium concentrations measured in soil and grass at the sampling stations close to the area impacted by the south plume (Stations 10, 11,15,17,20 and 32) are given in Table 6-8. The average concentration in soil for these stations was 4.7 pCi/g dry weight (0.17 Bq/g, 6.91 mg/kg), and the average concentration in vegetation was 0.037 pCi/g dry weight (0.001 Bq/g, 0.054 mg/kg).

Figure 6-17. Locations sampled for uranium in soil and vegetation in 1989 (from USDOE, 1990c).

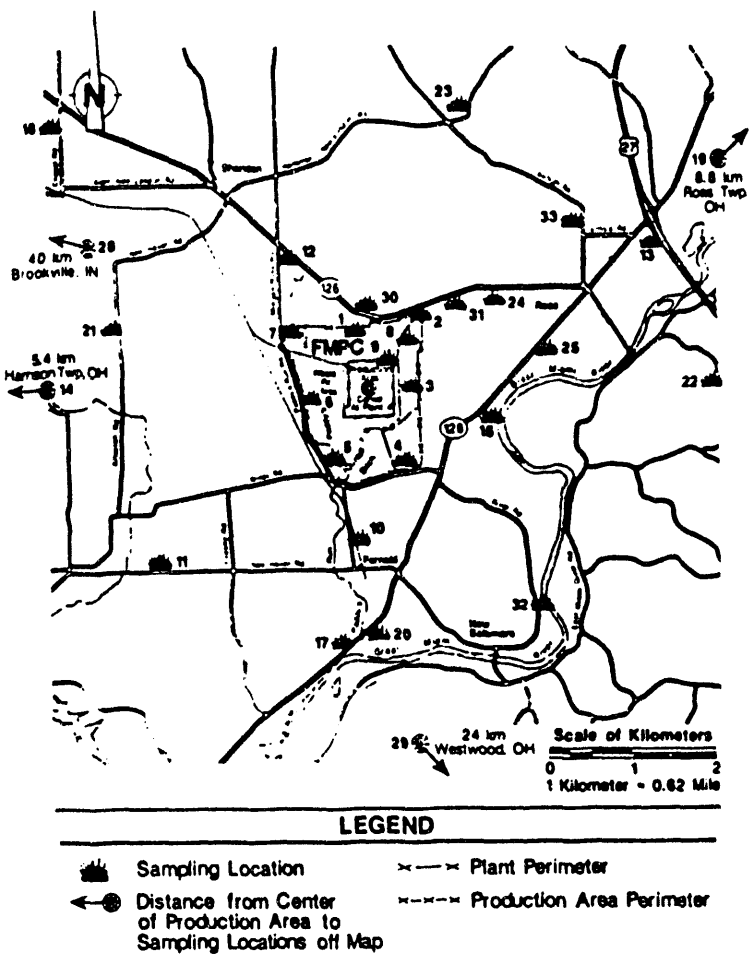


Table 6-8. Concentration of total uranium in soil and vegetation samples near the FEMP (USDOE 1990c)*.

Station	Conc. in Soil (pCi/g)	(Bq/g)	(mg/kg)	Conc. in Vegetation (pCi/g)	(Bq/g)	(mg/kg)
Impacted Area						
10	3.2	0.12	4.71	0.08	0.003	0.118
11	5.9	0.22	8.68	0.06	0.002	0.088
15	2.2	0.08	3.24	0.03	0.001	0.044
17	6.8	0.25	10.0	0.03	0.001	0.044
20	2.4	0.09	3.53	0.01	0.0004	0.015
32	7.4	0.27	10.9	0.01	0.0004	0.015
average	4.7	0.17	6.91	0.04	0.001	0.053
Background**						
14	2.7	0.10	3.97	0.02	0.0007	0.029
19	3.8	0.14	5.59	0.03	0.001	0.044
28	2.8	0.10	4.12	0.01	0.0004	0.015
29	8.0	0.30	11.76	0.01	0.0004	0.015
average***	3.1	0.11	4.56	0.02	0.0007	0.029

*all concentrations are dry weight
**stations greater that 5.4 km from the site
***soil average excludes station 29

Natural Background

Naturally occurring ^{238}U concentrations in Ohio range from about 0.6 to 2.2 pCi/g (Myrich et al., 1983). The 1989 Environmental Monitoring report (USDOE, 1990) assumes secular equilibrium between ^{234}U and ^{238}U and states that the total uranium activity will be about twice the activity of ^{238}U because the three isotopes occur together in soil. The value used as "background" appears to be 4.4 pCi/g total uranium (6.5 μg/g) (USDOE, 1990c). Uranium concentrations in offsite samples close to the facility are usually above this value.

Background concentrations of uranium in soil and vegetation near the FEMP were estimated from monitoring data. Samples at large distances from the FEMP (greater than 5.4 km) were used to describe the average background uranium concentrations in soil and vegetation near the facility. The average uranium concentration measured in soil at station 29 (8.0 pCi/g) was excluded from the calculation of average background concentrations because it was almost twice the background concentration expected for Ohio (4.4 pCi/g). The average total uranium concentration in soil for background stations was 3.1 pCi/g dry weight (0.11 Bq/g, 4.6 mg/kg), and the average concentration in vegetation was 0.02 pCi/g dry weight (0.0007 Bq/g, 0.03 mg/kg).

Because it is difficult to estimate natural background uranium concentrations in soil and vegetation near a facility that has contributed uranium to the nearby environment, the estimates of cancer risk associated with background uranium levels were based on estimated background concentrations in water based on monitoring data (0.924 μg/L, section 6.5.1) and on vegetation and milk concentrations estimated using the food chain model described in section 6.6. Food

and milk uranium concentrations based on background water, vegetation and soil measurements are higher than these values (see section 6.6.4). This results in an under-estimate of the risks associated with background uranium near the FEMP.

6.6.4 Estimated Uranium Concentrations in Homeproduced Food and Milk at Identified Receptors

Natural Background Concentrations

Based on a natural background concentration of 0.924 μg/L uranium in ground water, the estimated background concentrations were 0.006 mg/kg (0.15 Bq/kg) in food and 3.6×10^{-5} mg/kg (9.1×10^{-4} Bq/L) in milk. These values were used in the assessment of risk associated with natural background uranium concentration at the FEMP facility. Background concentrations in food and milk estimated from monitoring data near the facility are higher (food: 0.028 mg/kg, 0.7 Bq/kg, milk 0.0017 mg/L, 0.043 Bq/L).

Representative Residential Wells

Tables B-3 and B-4 in Appendix B give estimated uranium concentrations in ground water, homegrown food and homeproduced milk over time at the five representative well locations for the two discharge scenarios (stop source, continue source). These tables include estimates for both the cancer risk assessment (no contribution from background ground water concentrations or contribution of uranium in soil and vegetation to milk) and the toxicity assessment (contribution from background ground water concentrations and contribution of uranium in soil and vegetation to milk included). Estimated uranium concentrations in homegrown food range from 0.009 to 2.87 mg/kg. Concentrations in homeproduced milk range from 0.003 to 0.02 mg/L.

Centerline Wells

Tables B-5 and B-6 in Appendix B give estimated uranium concentrations in ground water, homegrown food and homegrown milk over time at the four centerline well locations for the two discharge scenarios (stop source, continue source). These tables include estimates for both the cancer risk assessment (no contribution from background ground water concentrations or contribution of uranium in soil and vegetation to milk) and the toxicity assessment (contribution from background ground water concentrations and contribution of uranium in soil and vegetation to milk included). Estimated uranium concentrations in homegrown food range from 0.006 to 1.47 mg/kg. Concentrations in homeproduced milk range from 0.003 to 0.011 mg/L.

Potential Wells

The concentration ranges chosen to describe the probability of locating a well in ground water with high uranium concentrations (0-1, 1-20, 20-100, 100-300 μg/L) were used to estimate the associated ranges of uranium concentrations in homeproduced food and milk. Table 6-9 gives the predicted concentration of uranium in food and milk associated with uranium ground water concentrations for each concentration in these identified ranges. This table contains estimates for the toxicity assessment which includes contributions of uranium from soil and vegetation to milk concentrations, and for the cancer assessment which excludes these sources of uranium.

Table 6-9. Potential wells-- concentrations of uranium in homegrown food and homeproduced milk for ground water concentration ranges.

Conc. in Water (μg/L)	TOXICITY Food (mg/kg)	Milk (mg/L)	CANCER RISK Food (mg/kg)	Milk (mg/L)
0	0	0.0026	0	0
1	0.006	0.0026	0.006	0.000036
20	0.122	0.0033	0.122	0.00072
100	0.608	0.006	0.608	0.0036
300	1.825	0.0134	1.825	0.0108
500	3.04	0.0206	3.04	0.018

6.6.5 Model Uncertainty

Because of the complexity of the food chain model, predictions of uranium concentrations in food and milk were produced in a deterministic way. Single estimates of the uranium concentrations at each grid cell and model time period were produced in the calculations (based on single estimates of ground water concentrations produced by the ground water transport model).

Uranium in food and milk were found to contribute only a small amount of the uranium exposure and risk (always less than 10%, most exposure is through water ingestion), and the uncertainty in the food chain model was ignored in this assessment.

6.7 Uranium Intake

Intake estimates were developed for the receptors described in section 6.4, using the predicted concentrations of uranium in ground water and food. Uranium intakes associated with uranium in food from other sources were also estimated.

Estimates of individual intake rates for the centerline and residential wells are presented as probability distributions. These distributions were calculated using Latin Hypercube sampling of the parameter distributions included in the formulae given below (section 6.7.1). These parameter distributions are documented in section 6.7.2.

6.7.1 Calculation of Intake

Ingestion in Water

Uranium intake through water ingestion (WI_c for cancer risk, WI_t for toxic effects) was calculated using the expressions:

$$C_{wc}\ (Bq/L) = C_{wt}\ (\mu g/L) \times 0.68\ (pCi/\mu g) \times 0.037\ (Bq/pCi)$$

$$WI_c\ (Bq/day) = C_{wc}\ (Bq/L) \times W\ (l/day)$$
$$WI_t\ (\mu g/day) = C_{wt}\ (\mu g/L) \times W\ (l/day)$$

where:

C_{wc} = uranium concentration (Bq/L)
C_{wt} = uranium concentration ($\mu g/L$)
WI_c = uranium intake (Bq/day)
WI_t = uranium intake ($\mu g/day$)
W = water intake (l/day)

Ingestion in Homeproduced Milk

Uranium intake in homeproduced milk (MI_c for cancer risk, MI_t for toxic effects) was calculated using the expressions:

$$C_{imc}\ (Bq/L) = C_{imt}\ (mg/L) \times 0.68\ (pCi/\mu g) \times 1000\ (\mu g/mg) \times 0.037\ (Bq/pCi)$$

$$MI_c\ (Bq/day) = C_{imc}\ (Bq/L) \times M\ (l/day)$$
$$MI_t\ (\mu g/day) = C_{imt}\ (mg/L) \times 1000\ (\mu g/mg) \times M\ (l/day)$$

where:

C_{imc} = uranium concentration in milk (Bq/L)
C_{imt} = uranium concentration in milk (mg/L)
MI_c = uranium intake in milk (Bq/day)
MI_t = uranium intake in milk ($\mu g/day$)
M = homeproduced milk intake (l/day)

Ingestion in Homegrown Fruits and Vegetables

Uranium intake through ingestion of homegrown fruits and vegetables (VI_c, VI_t) was calculated using the concentration of uranium in food (C_{ivt} mg/g) estimated for each receptor identified in section 6.6.

$$C_{ivc}\ (Bq/kg) = C_{ivt}\ (mg/kg) \times 0.68\ (pCi/\mu g) \times 1000\ (\mu g/mg) \times 0.037\ (Bq/pCi)$$

$$VI_c\ (Bq/day) = F\ (kg/day) \times FH \times C_{ivc}\ (Bq/kg)$$
$$VI_t\ (\mu g/day) = F\ (kg/day) \times FH \times C_{ivt}\ (mg/kg) \times 1000\ (\mu g/mg)$$

where:

F = amount of fruits and vegetables consumed (kg/day wet weight)
FH = fraction of fruits and vegetables that are homegrown
C_{ivt} = concentration of uranium in food from homegrown sources (mg/kg)
C_{ivc} = concentration of uranium in food from homegrown sources (Bq/kg)
VI_c = uranium intake in homegrown food (Bq/day)
VI_t = uranium intake in homegrown food (μg/day)

Ingestion in Food From Other Sources

The toxicity assessment for uranium in the south plume considered other sources of uranium in the diet which contribute to total uranium exposure. The amount of uranium ingested from other sources (OI_t) was calculated as:

OI_t (μg/day) = OF (μg/day) x FN

OI_t = amount of uranium consumed in fruits and vegetables from other sources for identified receptors (μg/day)
OF = amount of uranium consumed in fruits and vegetables from other sources for general population (μg/day)
FN = fraction of fruits and vegetables not grown at home (1-FH)

Total Intake

Total uranium intake (TI_c, Bq/day; TI_t μg/day) was calculated by summing all of the intake rate estimates described above.

TI_c (Bq/day) = WI_c + MI_c + VI_c
TI_t (μg/day) = WI_t + MI_t + VI_t + OI_t

6.7.2 Parameter Distributions

Parameters distributions used in the exposure calculations described above are given in Table 6-10. The following sections describe these distributions and document their sources.

Table 6-10. Parameter distributions used in exposure calculations*.

Parameter	Value	Reference
W (water intake)	lognormal distribution mean: 1.203 l/day sd: 0.689	Ershow and Cantor (1989)
M (milk intake)	logormal distribution mean: 0.254 l/day sd: 0.87	derived from USEPA (1984)
F (food intake)	lognormal distribution mean: 0.36 kg/day wet wt. sd: 0.13	derived from USEPA (1990b)
FH (fraction grown at home)	lognormal mean: 0.23 sd: 0.09	derived from USEPA (1990b)
FN (fraction not grown at home)	1 - FH	Section 6.7.2
OF (conc. in other food)	normal distribution mean: 0.934 μg/day sd: 0.069	USEPA (1991b)

*all means and standard deviations are arithmetic

Water Intake

Ershow and Cantor (1989) estimated total water and total tapwater intake for the population of the United States. This analysis was based on data collected in the 1977-1978 Nationwide Food Consumption Survey of the United States Department of Agriculture (USDA, 1984).

The water intake distributions reported by Ershow and Cantor (1989) are generally lognormally distributed (Roseberry and Burmaster, 1992). The average tapwater intake for the Midwest (all ages, both sexes) reported by Ershow and Cantor (1989) is 1.203 l/day, with a standard deviation of 0.689. This assessment assumed a lognormal distribution for water intake (W), with the parameters for the Midwest population reported by Ershow and Cantor (1989).

Intake of Fruits and Vegetables

USDA (1984, 1989) conducted a Nationwide Food Consumption Survey in 1977-1978 and estimated that the average amount of total fruits consumed on any one day was 140 g/day, and the average amount of vegetables consumed on any one day was 200 g/day (total of 340 g/day wet weight). These data were based on the results of a three day diet recall survey. Distributions of intake rates are available for individual fruit or vegetable types (e.g. oranges or green beans), but not for the larger categories of all fruits or all vegetables.

The USDA also gathered data on the percentage of consumed fruits and vegetables that were homegrown (USDA, 1989). These data were also summarized as distributions only for single fruit and vegetable types. Assuming a mix of all listed vegetables, the USEPA (1990b) estimated the overall average homegrown fraction of vegetables to be 0.25, and the overall average homegrown fraction of fruits to be 0.2. These calculations assumed that homegrown fruits are used to make juice.

These data are very uncertain -- the percentage of consumed fruits and vegetables that are homegrown varies with season, region, growing season and cost of fruits and vegetables in the market (USEPA, 1990b).

A central estimate of the fraction of consumed fruits and vegetables that are homegrown can be derived from the USEPA suggested (USEPA, 1990b) values of the average intake of fruits and vegetables (F=340 g/day) and the average fraction of food that is grown at home (0.25 vegetables, 0.2 fruits). These assumptions result in an assumed fraction of total fruit and vegetable consumption that is grown at home (FH) equal to 0.22 ([50 g/day vegetables + 28 g/day fruit]/ 340 g/day).

A distribution about these estimates can be derived by assuming that the central estimates derived above represent the geometric means of lognormal distributions. If it is assumed that the 95% confidence interval around these geometric means is + or - a factor of 2, the geometric standard deviation of these distributions can be calculated. These assumptions result in the following distributions:

	FOOD INTAKE	FRACTION HOMEGROWN
geometric mean:	0.34 kg/day	0.22
geometric sd:	1.42	1.42
arithmetic mean:	0.36 kg/day	0.23
arithmetic sd:	0.13	0.09

The risk assessment presented here was based on the these distributions.

Intake of Homeproduced Milk

USEPA (1984) estimated the consumption rates for fresh milk in the United States based on an analysis of data collected in the USDA Nationwide Food Consumption Survey conducted in 1977-1978 (USDA 1984). The reported mean intake for fresh milk (all ages) was 0.254 l/day, with a standard error of 0.005. This analysis was based on a survey of approximately 30,700 individuals. A standard deviation of 0.87 was derived for this distribution. The estimated consumption rates for all dairy products was 308.6 g/day.

For the purpose of this assessment it was assumed that all milk consumption was homeproduced, and all other dairy products were obtained from other sources. The distribution of milk intake was described by a lognormal distribution with an arithmetic mean and standard deviation of 0.245 l/day, and 0.87 respectively.

Amount of Uranium Ingested in Fruits and Vegetables Not Grown at Home

A few "market basket" studies have been conducted to estimate the amount of uranium in the diet. Welford and Baird (1967) studied the uranium content of adult diets in New York City, Chicago and San Francisco. The average daily intake for adults in the three cities was similar: 1.3 μg/day in New York City, 1.4 μg/day in Chicago and 1.3 μg/day in San Francisco.

USEPA sponsored a re-evaluation of the data reported by Welford and Baird (1967). Meglen (1985, as cited in USEPA, 1991b) used the average uranium concentrations measured in various food groups in the Welford and Baird (1967) study as input to a model based on more recent data on food consumption rates (USDA, 1984). The daily average intake rates estimated by this model for the various adult age and sex categories ranged from 0.7 to 1.1 μg/day.

USEPA (1991b) assumed a normal distribution of uranium intakes, with an average of 0.934 μg/day and a standard deviation of 0.069. This distribution was based on modeled results (Meglen, 1985 as cited in USEPA 1991b) for six adult male age groups for each of the three cities in the Welford and Baird (1967) study. This distribution was used in this assessment to describe the amount of uranium ingested in food from sources other than home gardens (OF).

The fraction of food ingested from sources other than home gardens (FN) was assumed to be 1 minus the fraction grown at home.

6.7.3 Intake Calculations: Background, Toxicity Assessment, and Cancer Risk

6.7.3.1 Natural Background

Intakes associated with background concentrations of uranium in ground water (0.924 μg/L, Section 6.5.1) were calculated using the formulae described above, and the predicted concentrations in food and milk given in Section 6.6.4 (food: 0.006 mg/kg, 0.15 Bq/kg ; milk: 3.6×10^{-3} mg/L; 9.1×10^{-4} Bq/L).

6.7.3.2 Toxicity Assessment

Intake Estimates for Individual Wells

For the prediction of toxic effects, individual exposures were calculated in units of μg/day, using water, food and milk concentrations for the years simulated by the ground water transport model described in section 6.6.5.2 (time 0, 10, 20, 30, 40, 50, 60 and 70 years). The contribution from background uranium levels in ground water, and the contribution of uranium in feed and soil to milk was estimated.

Intake Estimates for Potential Future Wells

Grid cells (125 x 125 feet) representing potential wells were grouped according to the predicted uranium water concentration in the grid (0-1, 1-20, 20-100, 100-300, 300-500, section 6.5.2.3). Food and milk concentrations associated with water from each group were calculated (Section 6.6). Concentrations in water, food and milk associated with each group was expressed as a uniform distribution covering the stated range of concentrations. These concentrations were then multiplied by the distributions of amount eaten or drunk daily (section 6.7.2 , Table 6-10).

6.7.3.3 Cancer Risk

Intake Estimates for Individual Wells

For the calculation of cancer risk, intake distributions were estimated in units of Bq/day for the estimated concentrations of uranium in water, food and milk predicted for each year of a 70-year lifetime. These distributions were used to estimate the committed effective dose (Sv) and risk associated with chronic uranium intake over a lifetime. These dose calculations and results are described in the section on risk characterization (Section 6.8.2).

Intake Estimates for Potential Future Wells

Average uranium intakes were calculated for each potential future well location identified in section 6.5.2.3. Average intakes were estimated for each ten year increment of a lifetime, based on the average uranium concentration predicted for ground water during that period. These intakes were used to estimate the committed effective dose and risk associated with intake of uranium over a lifetime at each location. These calculations and results are described in the section on risk characterization (Section 6.8.2).

6.8 Risk Characterization

6.8.1 Risk of Toxic Effects

Estimation of the risk of toxic effects associated with ingested uranium includes two steps: 1) calculation of the amount of uranium expected in the kidney and 2) comparison of the amount of uranium in the kidney to the distribution of the effect threshold derived in Section 6.3.2.

The amount of uranium expected in the kidney for a given chronic intake (or distribution of intakes) was calculated as described in Section 6.3.2, using the retention function for the kidney, modified by f_1, the fractional transfer of uranium from gut to blood.

$$A = TI_t \times (f_1/Ln2) \times (\alpha T)$$

where:

A = concentration of uranium in kidney (μg/g)
TI_t = uranium intake rate (μg/day)
f_1 = fractional transfer gut to blood
α = fractional transfer blood to kidney compartment 1
T = biological half-life of uranium in kidney compartment 1

Values for the parameters used in this calculation were derived in section 6.3.2. Table 6-11 summarizes the parameters and distributions of parameters used in the calculation of the amount of uranium in the kidney. The intake distributions (TI_t) used in the analysis were those calculated in Section 6.7.

Table 6-11. Estimated values and distributions for parameters used in calculating the amount of uranium in the kidney .

	Geometric Mean	Geometric Std. Dev.
f_1	1.03	1.78
α	6.3%	1.87
T	11.1 days	1.55

all distributions are lognormal

The distribution of the threshold concentration in the kidney was derived in section 6.3.2. This distribution is described by a Weibull distribution, with a lower limit of 0.1 μg/g, scale parameter of 0.6 and a shape parameter of 4. These parameters produce a maximum threshold value of about 1 μg/g with a mean of 0.6.

The probability of a toxic effect was defined as the probability of the kidney concentration exceeding the threshold. In the calculation, this is the fraction of the distribution [kidney concentration/threshold] that exceeds one.

Probability of Toxic Effects at Background Water Concentrations

Risks associated with the intake of uranium associated with background ground water concentrations (0.924 μg/L) were calculated as described above. The predicted kidney concentrations were always smaller than the threshold concentrations, and no toxic effects were predicted.

Probability of Toxic Effects at Individual Well Locations

The model described above was used to calculate the range of kidney concentrations expected for the four centerline and five residential well locations. In all cases the predicted kidney concentrations were always smaller than the threshold concentration, and no toxic effects were predicted.

Probability of Toxic Effects for Potential Future Wells

Grid cells representing potential future wells were binned into four groups based on the predicted uranium water concentration for the cell (each model year). The five groupings were: 0-1, 1-20, 20-100, 100-300 and 300-500 μg/L. Uranium intakes for these ranges were calculated as described in section 6.7.4.1. Monte Carlo simulations were run to estimate distribution of kidney concentration and probability of exceeding the threshold for each group. None of the concentration groups had any probability of exceeding the threshold. The full range of equilibrium kidney concentrations were:

WATER	KIDNEY
0-1 μg/L	0-0.001 μg/g
1-20 μg/L	0-0.015 μg/g
20-100 μg/L	0-0.074 μg/g
100-300 μg/L	0-0.22 μg/g
300-500 μg/L	0-0.38 μg/g

Although this analysis predicted no toxic effects for any potential future wells, the largest predicted kidney concentration was 72% of the threshold level. An additional assessment was performed incorporating additional potential sources of error.

The ground water, food and milk concentration estimates (section 6.5) did not include uncertainty. We judged the ground water model, especially in the higher concentrations of most concern, to have an uncertainty of less than 1.5. We included this in the analysis as a lognormal distribution with a geometric mean of 1 and a 95% range of +/-1.5 times, leading to a geometric standard deviation of 1.2 (arithmetic mean = 1.02 and standard deviation = 0.19). The food and milk pathways contribute less than 10% of the total uranium exposure, and the uncertainty in the food chain model was ignored.

A second source of uncertainty was in the pharmacokinetic model formulation itself. While it is a straight-foward process to develop probability distributions representing uncertainty of parameters in a model, it is more difficult to express the uncertainty in the model formulation itself. As a practical approach to this problem, we assumed that uncertainty from this source was +/- a factor of two. We implemented this by specifying a lognormal distribution with geometric mean of one and geometric standard deviation of 1.4 (arithmetic mean = 1.06, standard deviation = 0.37).

This revised formulation resulted in a probability of 0.02% of exceeding the threshold for water concentrations in the range of 300-500 μg/L. In all other groups the probability of exceeding the threshold remained zero.

Water concentrations above 300 μg/L were predicted only for the initial conditions (Model Year 0, 1989). In Model Year 0, 2% of the grid cells in the potentially impacted area had predicted uranium concentrations in the range 300-500 μg/L. The overall probability of exceeding the toxic threshold in a randomly drilled future well in Model Year 0 (1989) and in the next 70 years was 0.02 x 0.0002 = 4×10^{-6}.

6.8.2 Cancer Risk

Because uranium delivers an internal dose over a period of time (Section 6.3.3), the calculation of the integrated lifetime dose (and risk) for an individual receptor was based on the concept of an effective dose (ICRP 1991a). This required assumptions about the length of the exposure, and the year of life in which each annual exposure occurs.

This assessment assumed a 70-year lifetime and exposure period (the length of the model simulation), and an initial intake in the first year of life. This is a conservative assumption because most people will not live in one place for their entire lifetime (see Section 5.4.3). The method described here can be applied to any exposure period, beginning at any year of life.

Committed effective doses (E_{70}, Sv) per unit annual intake ($Bq \cdot yr^{-1}$) were calculated for intake during each year of life in Section 6.3.3. The committed effective dose for intake during each year was calculated for each receptor by multiplying the estimated annual intake ($Bq \cdot yr^{-1}$) times the "lifetime modified" committed effective dose for a unit intake in that year of life, correcting for the f_1 distribution derived in section 6.3.2. These effective doses for intake each year were then summed to calculate the effective dose associated with the lifetime intake of uranium.

$$E_{70}(Y) = CF \times (E_{70} / 1\ [Bq \cdot yr^{-1}]) \times TI_c(Y) \times 365\ [d \cdot yr^{-1}]$$

$$E_{70}L = \text{SUM } y=0,70\ E_{70}A(Y)$$

where:

$E_{70}(Y)$ = effective dose (Sv) integrated over a 70-year lifetime, for intake in year y
$CF = f_1$ distribution (section 6.3.2)/5
$TI_c(Y)$ = daily intake in year y ($Bq \cdot d^{-1}$)
$E_{70}L$ = sum of effective doses (Sv) over a 70 year lifetime, for lifetime intake (years 0-70)

Individual lifetime risks for cancer mortality risks (ILR) were calculated for each receptor by multiplying the committed effective dose ($E_{70}L$) for lifetime exposure to uranium as described above, by the distribution for the risk factor (RF) derived in section 6.3.3.4 (lognormal distribution, geometric mean 0.05 deaths/Sv, geometric standard deviation 1.4).

$$ILR = E_{70}L\ [Sv] \times RF\ [deaths/Sv]$$

Cancer Risk for Natural Background Water Concentrations

The individual lifetime fatal cancer risk predicted for background uranium concentrations in ground water had an arithmetic mean of 6.9×10^{-7}, a median of 4.9×10^{-7} and an upper 95% confidence level of 1.8×10^{-6}.

Cancer Risk at Individual Well Locations

Table 6-12 presents the individual lifetime fatal cancer risk distributions estimated for the five representative residential and four centerline well locations. Average predicted individual lifetime risks ranged from 7.2×10^{-12} to 1.2×10^{-5} for the stop source scenario and from 5.6×10^{-11} to 1.3×10^{-5} for the continue source scenario. The largest average individual lifetime risk was predicted for Residential Well 1 (continue source scenario, 1.3×10^{-5}).

Cancer Risk at Potential Future Well Locations

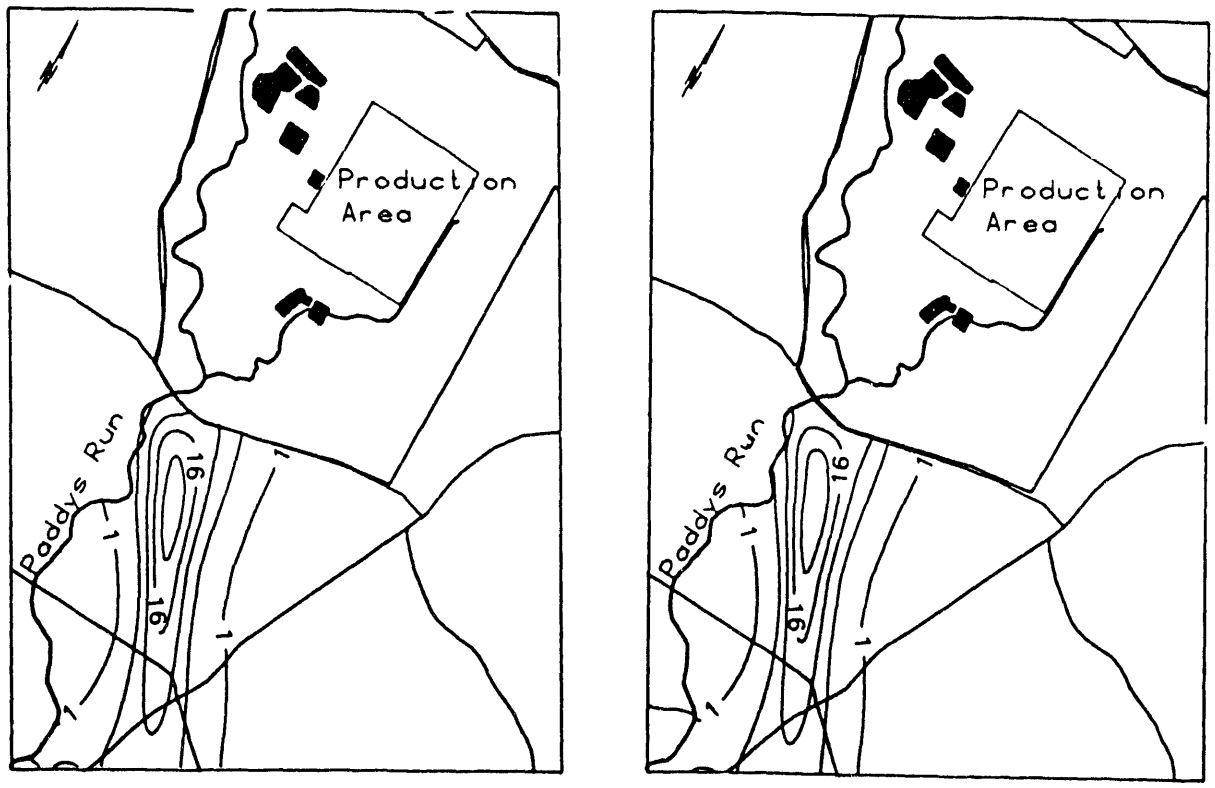
The median individual lifetime cancer risks for potential future wells in the impacted area are shown in Figure 6-18. The largest predicted median individual lifetime cancer risk for both the stop source and continue source scenarios was 2.2×10^{-5}. The distribution of risks are similar for the two scenarios because most of the exposure is received in the first years of life when the water concentrations associated with uranium discharges prior to 1989.

Table 6-12. Representative residential and centerline wells -- individual lifetime cancer risk distributions.

WELL	STOP SOURCE MEAN	STOP SOURCE MEDIAN	STOP SOURCE 95% CL	CONTINUE SOURCE MEAN	CONTINUE SOURCE MEDIAN	CONTINUE SOURCE 95% CL
RESIDENTIAL						
1	1.2E-5	7.7E-6	3.4E-5	1.3E-5	8.7E-6	3.8E-5
2	2.2E-6	1.6E-6	5.7E-6	2.2E-6	1.7E-7	5.9E-6
3	7.2E-12	5.0E-12	2.0E-11	5.6E-11	4.0E-11	1.6E-10
4	2.2E-6	1.6E-6	6.4E-6	2.2E-6	1.6E-6	6.7E-6
5	8.6E-6	6.0E-6	2.4E-5	8.7E-7	6.1E-6	2.4E-5
CENTERLINE						
1	9.4E-6	7.0E-6	2.6E-5	1.1E-5	8.5E-6	3.1E-5
2	4.4E-6	3.2E-6	1.3E-5	4.6E-6	3.3E-6	1.3E-5
3	7.9E-6	5.7E-6	2.3E-5	8.1E-6	5.8E-6	2.3E-5
4	9.0E-6	6.7E-6	2.3E-5	9.1E-6	6.8E-6	2.4E-5

Figure 6-18. Median individual lifetime cancer risk (x 10^{-6}) for potential future wells in the impacted area, stop source and continue source scenarios (contours shown are 1 x 10^{-6} to 21 x 10^{-6}, contour interval is 2.5 x 10^{-6}).

STOP SOURCE **CONTINUE SOURCE**



6.9 Assumptions and Uncertainties

This assessment included a number of assumptions and associated uncertainties which should be considered in interpreting the results of this analysis. The major assumptions incorporated into the assessment are described below.

6.9.1 Toxicity Assessment

Parameters for Pharmacokinetic Model and Toxicity Threshold

The values for the parameters used in the pharmacokinetic model and the toxicity threshold for uranium in the kidney are sources of uncertainty in the toxicity assessment. This uncertainty was incorporated into the risk assessment by describing these parameters as distributions derived from the range of published data.

Model Predictions

Model Formulation Uncertainties -- Although uncertainty in individual parameters of the intake-to-kidney concentration model has been characterized, there is no direct method to estimate uncertainties associated with the specific formulation of the model used. This uncertainty was accounted for in a separate uncertainty analysis (Section 6.8.1) by assuming that the range of uncertainty from this source is about a factor of two.

Ground Water Transport Model -- The ground water concentration estimates are also uncertain. The uncertainties associated with the ground water transport model were incorporated in a separate uncertainty analysis (section 6.8.1) by assuming that the ground water model had an uncertainty of less than 1.5. Uranium in food and milk contributed less than 10% of the uranium exposure (and risk) and the uncertainty in the food chain model was ignored.

Intake Rates

Intake rates for water, food and milk were derived from published data. The variation in intake rates in a population was accounted for by describing these rates as distributions.

6.9.2 Cancer Risk Assessment

Dose Estimation

The estimation of dose associated with a unit intake (1 $Bq \cdot yr^{-1}$) was derived from ICRP (1991b) Annual Limits on Intake Values. These values were derived to protect workers, and are probably conservative estimates which result in an overestimate of the risk associated with the ingestion of uranium from the FEMP facility. The importance of the uncertainty in the dose estimates was investigated for a single well (Residential Well 1) by assuming that the dose estimates had an uncertainty of about a factor of 2. We included this uncertainty in the analysis as a lognormal distribution with geometric mean one and a 95% range of + or - two times, leading to a geometric standard deviation of 1.4 (arithmetic mean = 1.06, standard deviation = 0.37). Including this uncertainty in the assessment had only a small effect on the resulting risk distribution (Table 6-13).

Model Predictions

The ground water concentration estimates are also uncertain. The importance of the uncertainties associated with the ground water transport models were assessed for a single well (Residential Well 1) by assuming that the ground water model had an uncertainty of less than 1.5. This was included in the analysis as a lognormal distribution with a geometric mean of one and a 95% range of + or - 1.5, leading to a geometric standard deviation of 1.2 (arithmetic mean = 1.02, standard deviation = 0.19). Including this uncertainty in the assessment had only a small effect on the resulting risk distribution (Table 6-13). Uranium in food and milk contributed less than 10% of the uranium exposure (and risk) and the uncertainty in the food chain model was ignored.

Table 6-13. Uncertainty analysis for cancer risk assessment -- dose estimates and transport model uncertainty.

	INDIVIDUAL LIFETIME CANCER RISK* ADDITIONAL SOURCES OF UNCERTAINTY			
	NONE	DOSE ESTIMATES	TRANSPORT MODELS	BOTH DOSE** AND MODELS
Mean	1.3E-5	1.4E-5	1.3E-5	1.4E-5
Median	8.7E-6	9.3E-6	8.9E-6	9.0E-6
Maximum	1.1-4	1.3E-4	1.3E-4	1.5E-4
95% CL	3.8E-5	4.0E-5	3.8E-5	4.0E-5

*Residential Well 1, continuing source scenario
**analysis incorporating uncertainty in both dose estimates and transport model predictions

Risk Factor

The uncertainty in the risk factor for low dose radiation was incorporated into the assessment by deriving a distribution about the value recommended by ICRP (0.05 deaths·Sv^{-1}, ICRP, 1991a).

Intake Rates

Intake rates for water, food and milk were derived from published data. The variation in intake rates for a population was accounted for by describing these rates as distributions.

6.10 Summary -- Risk Assessment for Uranium in Ground Water

This risk assessment for uranium in ground water south of the FEMP facility was performed for 70 years into the future and estimates risks to currently located wells, to potential wells located along the centerline of the developing plume and to future wells located in the potentially impacted area south of the site.

No toxic effects were predicted for any individual well. An assessment incorporating the uncertainty in transport and food chain model predictions resulted in a predicted probability of toxic effects of 4×10^{-6} for any well located south of the FEMP in the next 70 years.

All estimated cancer risks were small. The largest predicted individual lifetime risk was for Residential Well 1 (1.3×10^{-5}). The predicted individual lifetime cancer fatality risks for wells located anywhere south of the facility were small (always less than 2.2×10^{-5}). Predicted risks for both the stop and continue source scenarios were similar because most exposure is associated with uranium discharged before 1989.

7 SUMMARY AND CONCLUSIONS

Two important environmental problems at the USDOE Fernald Environmental Management Project facility in Fernald, Ohio were studied in this human health risk assessment. The problems were chosen for assessment based on: 1) USDOE, public and regulatory concern, 2) potential for offsite exposure and 3) high score in the USDOE Environmental Survey's implementation of the MEPAS (Multi Media Environmental Pollution Assessment System) code.

The problems studied in this assessment were radon emissions from the K-65 waste silos, and offsite contamination of ground water with uranium.

The radon assessment involved the development of a source term (based on available monitoring data), and the prediction of exposure and risk to fenceline residents, residents within 1 and 5 miles of the silos, and residents of Hamilton and Cincinnati, Ohio. A gaussian plume model and site metrological data were used to make these predictions. Two release scenarios were studied: the continuing, routine release of radon from the silos and an accidental loss of one silo dome integrity. Time indoors, time living at a residence and the risk factor for radon were described as distributions based on available data. The exposures and risks associated with background radon concentrations were also estimated.

Radon and radon progeny associated with the K-65 silos were estimated to result in individual lifetime risks greater than 1×10^{-4} only for indoor workers (median: 4.3×10^{-4}) and fenceline residents (median: 1.8×10^{-4}) under the routine release scenario. Population risks associated with the routine and accidental release scenarios were less than 1.0 for all identified receptor populations. The individual and population risks associated with background radon concentrations (median individual risk: 7.3×10^{-3}) were 1 to 4 orders of magnitude larger than the risks associated with radon from the silos.

The uranium risk assessment was based on model predictions performed by IT Corporation, using a calibrated, three dimensional ground water flow and transport model. Exposures and risks were estimated 70 years into the future for currently located residential wells, potential future wells located along the center of the developing plume, and all possible future well locations in the impacted area.

Exposure routes included in the assessment were: water ingestion, intake of homegrown food and intake of homeproduced milk. Intake rates for water, food and milk were based on distributions derived from published data. The cancer risk assessment was based only on exposures associated with uranium in ground water

emanating from the FEMP (direct water ingestion, ingestion in food irrigated with contaminated water, and ingestion in milk from cows watered with contaminated water). Because toxicity is a threshold effect, the toxicity assessment included contributions to milk from soil and vegetation that were not associated with ground water contamination, as well as uranium associated with the use of ground water.

The toxicity risk assessment was based on a pharmacokinetic model and parameter distributions derived from the published literature. A threshold distribution for effects from uranium in the kidney was developed from data available in the literature, and the probability of a toxic effect was defined as the probability of kidney uranium concentration exceeding the threshold.

The cancer risk assessment was based on ALI values (annual limits on intake) published by ICRP (1979), modified to reflect a distribution of gut uptake factors and allowing dose to be committed only for 70 years.

No toxic effects were predicted for any individual well. An assessment incorporating the uncertainty in transport and food chain model predictions and the pharmacokinetic model formulation resulted in a predicted probability of toxic effects of 4×10^{-6} for any randomly located well south of the FEMP in the next 70 years.

All estimated cancer risks were small. The largest predicted individual lifetime risk was for Residential Well 1 (1.3×10^{-5}). The predicted individual lifetime cancer fatality risks for wells located anywhere south of the facility were small (always less than 2.2×10^{-5}). Predicted risks for both the stop and continue source scenarios were similar because most exposure is associated with uranium discharged before 1989. Including additional uncertainties (from ground water and food chain model and dose estimate) had little effect on the risk estimates.

8 REFERENCES

Alexander, R., 1988, comments in *Ultrasensitive Techniques for Measurement of Uranium in Biological Samples and the Nephrotoxicity of Uranium*, Kathren, R.L. and J.R. Webber, eds., NUREG/CP-0093, United States Nuclear Regulatory Commission, Washington, DC.

Ames, L.L. and D. Rai, 1978, *Radionuclide Interactions with Soil and Rock Media: Vol. 1, Processes Influencing Radionuclide Mobility and Retention, Element Chemistry and Geochemistry, Conclusions and Evaluation*, EPA 520/6-78-007, U.S. Environmental Protection Agency, Office of Radiation Programs, Las Vegas, NV.

Baes, T.S., C.F., III, and R.D. Sharp, 1983, "A Proposal for Estimation of Soil Leaching and Leaching Constants for Use in Assessment Models," *Journal of Environmental Quality* 12:17-28.

Bassett, S.H., A. Frenkel, N. Cedars, et al., 1948, *The Excretion of Hexavalent Uranium Following Intravenous Administration. II. Studies on Human Subjects,* USAEC Report UR-37.

BNI, 1990, *K-65 Silos Removal Action at the Feed Materials Production Center, Fernald, Ohio for the United States Department of Energy*, FMPCREV2.WP, Bechtel National, Inc.

Boback, M.W., T.A. Dugan, D.A. Fleming, R.B. Grant and R.W. Keys, 1987, *History of FEMP Radionuclide Discharges*, Feed Materials Production Center, Westinghouse Materials Company of Ohio. FMPC-2082.

Borak, T.B., 1985, *Calculation of Radon Emission, Dispersion and Dosimetry from K-65 Storage Tanks at the Feed Materials Production Center*, Feed Materials Production Center, Fernald, Ohio.

Brenk, H.D., J.E. Fairobent and E.H. Markee, Jr., 1983, "Transport of Radionuclides in the Atmosphere", in: *Radiological Assessment*, J.E. Till and H.R. Meyer, eds., NUREG/CR-3332, United States Nuclear Regulatory Commission, Washington, D.C.

BRH, 1970, *Radiological Health Handbook*, Bureau of Radiological Health, United States Department of Health, Education and Welfare, Rockville, Md., 20852.

Brookins, D.G., 1984, *Geochemical Aspects of Radioactive Waste Disposal*, Wiley Interscience, 420 pp.

Butterworth, A., 1958, "Human Data on Uranium Exposure", in *Symposium on Occupational Health Experience and Practices in the Uranium Industry*, (HASL-58), Health and Safety Laboratory, New York, N.Y., pp. 41-46.

Callahan, M.A., M.W. Slinak, N.W. Gabel, I.P. May, C.F. Fowler, J.R. Freed, P. Jennings, R.L. Durfee, F.C. Whitmore, B. Maestri, W.R Mabey, B.R. Holt, and C. Gould, 1979, *Water-Related Environmental Fate of 129 Priority Pollutants: Vol. 1, Introduction and Technical Background, Metals and Inorganics, Pesticides and PCB's*, EPA 440/4-79-029, U.S. Environmental Protection Agency, Office of Water Planning and Standards, Washington, D.C.

Camargo Assoc., 1986, *K-65 Silos Study and Evaluation Report*, Camargo Associates Limited, Cincinnati, Ohio.

Chapin, S., 1974, *Human Activity Patterns in the City: Things People Do in Time and Space*, Wiley Interscience, New York.

Cohen, B.L., 1990, "Seasonal Variations of Radon Levels in Homes", *Radiat. Prot. Mgmt.* 7:62-67.

Cohen, B.L., 1991, "Variation of Radon Levels in U.S. Homes Correlated With House Characteristics, Location and Socioeconomic Factors", *Health Physics* 60:631-642.

Cohen, B.L., and R.S. Shah, 1991, "Radon Levels in United States Homes by States and Counties", *Health Physics* 60:243-259.

Cohrssen, J.J. and V.T. Covello, 1989, *Risk Analysis: A Guide to Principles and Methods for Analyzing Health and Environmental Risks*, United States Council on Environmental Quality, Executive Office of the President.

Colle, R., R.J. Rubin, L.I. Knab and J.M. R. Hutchinson, 1981, *Radon Transport Through and Exhalation from Building Materials*, NBS Technical Note 1139, U.S. Department of Commerce, Washington, D.C.

Cookfair, D.L., W.L. Beck, C. Shy, C.C. Lyshbaugh and C.L. Sowder., 1983, "Lung Cancer Among Workers at a Uranium Processing Plant", Presented at Epidemiology Applied to Health Physics, in *Proceedings of the Health Physics Society*, pp. 398-406.

Dames & Moore, 1985, *Department of Energy, Feed Materials Production Center, Groundwater Study, Task C Report*, Prepared for NLO, Inc., July 1985.

Dang, H.S., V.R. Pullat and K.C. Pillai, 1992, "Gastrointestinal Absorption Factor for Uranium Incorporated in Diet", *Radiation Protection Dosimetry* 40: 195-197.

Diamond, G. L. 1989, "Biological Consequences of Exposure to Soluble Forms of Natural Uranium", *Radiation Protection Dosimetry* 26(1/4): 23-33.

DDHS, 1990, *Vital Statistics of the United States, 1987*, United States Department of Health and Human Services, Public Health Service, Hyattsville, Md.

Diamond, G.L., P.E. Morrow, B.J. Panner, R.M. Gelein, and R.B. Baggs, 1989, "Reversible Uranyl Fluoride Nephrotoxicity in the Long Evans Rat", *Fundamental and Applied Toxicology* 13: 65-78.

Domenico P.A., and Schwartz, F.W., 1990, *Physical and Chemical Hydrogeology*, John Wiley and Sons, New York, 824 pp.

Draxler, R.R., and J.L. Hefter, 1981, *Workbook for Estimating the Climatology of Regional Continental Scale Atmospheric Dispersion and Deposition over the United States,* NOAA Technical Memorandum ERL-ARL-96, Air Resources Laboratory, Silver Springs, Md.

Droppo, J.G., Jr., J.W. Buck, D.L. Strenge and M.R. Siegel, 1990, *Analysis of Health Impact Inputs to the U.S. Department of Energy's Risk Information System.* PNL-7432, Pacific Northwest Laboratory, Richland Washington.

Ershow, A.G. and K.P. Cantor, 1989, *Total Water and Tapwater Intake in the United States: Population-Based Estimates of Quantities and Sources*, Life Sciences Research Office, Federation of American Societies for Experimental Biology, Rockville, MD.

Evans, R.D., 1969, "Engineer's Guide to the Elementary Behavior of Radon Daughters", *Health Physics* 17:229-252.

Federal Register, 1991, "United States Environmental Protection Agency, National Primary Drinking Water Regulations; Radionuclides", *40CFR Parts* 141, 142. 56:138.

Fish, B.R., J.A. Payne, and J.L. Thompson, 1960, "Ingestion of Uranium Compounds", in *Oak Ridge National Laboratory, Health Physics Division, Annual Report*, Oak Ridge National Laboratory, Oak Ridge , TN. ORNL-2994, pp. 269-272.

Fries, G.F., 1986, "Assessment of Potential Residues in Foods Derived From Animals Exposed to TCDD Contaminated Soil", presented at DIOXIN87: *Sixth International Symposium on Chlorinated Dioxins and Related Compounds*, Fukuoka, Japan, as cited in USEPA, 1990b.

Garrels, R.M., and Christ, C.G., 1965, *Solutions, Minerals and Equilibria*, Harper & Row, New York.

Geomet, 1981., *Comparison of Indoor and Outdoor Air Quality*, PRI EA-1733, Project 1309, GEOMET, Inc., Gaithersburg, Maryland.

Geotrans, 1988, *SWIFT III Quality Assurance Benchmark Problem Execution Fiche*, Geotrans, Inc., Herndon, VA.

Goldschmidt, V.M., 1954, *Geochemistry*, Clarendon Press, Oxford.

Grumski, J.T., 1987, *Feasibility Investigation for Control of Radon Emissions from the K-65 Silos*, Westinghouse Materials Company of Ohio, Fernald Ohio.

Hagee, G.R., P.H. Jenkins, P.J. Gephart and C.R. Rudy, 1985, *Radon and Radon Flux Measurements at the Feed Materials Production Center, Fernald, Ohio*, Monsanto research Corporation, Miamisburg, Ohio, MlM-MU-85-68-001, August, 1985.

Hamilton, J.G., 1948, "The Metabolic Properties of The Fission Products and Actinide Elements", *Rev. Mod. Phys.* 20:718-723.

Hamilton, L.D., 1989, "Problems in Estimating the Risk of Low Dose Environmental Radiation", in *Radiation Protection: Past and Future*, D.J. TerMarsch and D.K. Myers, eds., Workshop/Symposium held at Chalk River Laboratories, Chalk River, Ontario, Canada, Atomic Energy of Canada Limited, AECL-9959.

Harrington, C.D. and A.E. Ruehle, 1959, *Uranium Production Technology*, D. Van Nostrand Company Inc., Princeton, N.J., as cited in USDOE 1990a.

Harrison, R.M. and H.A. McCartney, 1980, " A Comparison of the Predictions of a Simple Gaussian Plume Dispersion Model with Measurements of Pollution Concentration at Ground-Level and Aloft", *Atmos. Environ.* 14:589-596.

Harrison, J.D. and J.W. Stather, 1981, "The Gastrointestinal Absorption of Proactinium, Uranium and Neptunium in the Hamster", *Rad. Res.* 88:47-55.

Hem, J.D., 1982, "Study and Interpretation of the Chemical Characteristics of Natural Water", *Geological Survey Water Supply Paper 1473*, 1982.

Hill, M.S., 1985, "Patterns of Time Use", in: *Time, Goods and Well-Being.*, Juster, F.T., Stafford, F.P., eds. Survey Research Center, Institute for Social Research University of Michigan, Ann Arbor, MI.

Honeyman, B.D., and Santchi, P.H., 1988. "Metals in Aquatic Systems," *Environmental Science & Technology*, 22:862-871.

Hopper, R.D., R.A. Levy, R.C. Rankin and M.A. Boyd, 1990, *National Ambient Radon Study*, United States Environmental Protection Agency, Las Vegas, NV.

Hursh, J.B., W.F. Newman, T. Toribara et al., 1969, "Oral Ingestion of Uranium by Man", *Health Physics* 17:619-621.

Hursh, J.B. and N.L. Spoor, 1973, "Data on Man", in *Uranium, Plutonium, Transplutonic Elements, Handbook of Experimental Pharmacology*, H.C. Hodge, J.N. Stannard, and J.B.Hursh, eds., Springer-Verlag, New York, pp. 197-239.

ICRP, 1960, *Report of Committee II on Permissible Dose for Internal Radiation; ICRP Publication 2*, International Commission on Radiological Protection, Pergamon Press, Oxford.

ICRP, 1979, "Limits for Intakes of Radionuclides by Workers", International Commission on Radiological Protection, ICRP Publication 30, Part 1. *Annals of the ICRP Committee 2*, No. 3/4, Pergamon Press, Oxford.

ICRP, 1987, "Lung Cancer Risk From Indoor Exposures to Radon Daughters", ICRP Publication 50, *Annals of the ICRP* 17(1), Pergamon Press, Oxford.

ICRP, 1991a. "1990 Recommendations of the International Commission on Radiological Protection", Publication 60, *Annals of the ICRP*, 21:1-3, Pergamon Press, Oxford.

ICRP, 1991b. "Annual Limits on Intake of Radionuclides by Workers Based on the 1990 Recommendations", Publication 61, *Annals of the ICRP*, Volume , No. . Pergamon Press, Oxford.

Ijaz, T., Janke, R., and Janke, R., 1990, *A Baseline Risk Assessment for the K-65 Silos Using EPA Methodology For Applicability to the EE/CA*, Revision 1, University of Cincinnati, R-008-204.2.

Israeli, M., and C.B. Nelson, 1992, "Distribution and Expected Time of Residence for U.S. Households", *Risk Analysis*, 12:(1):65-72.

IT, 1989, *Assessment of Radiation Dose and Cancer Risk for Emissions from 1951 Through 1984, Feed Materials Production Center, Fernald, Ohio*, IT Corporation, Knoxville, Tennessee, for United States Department of Energy, Project No. 303063.

IT, 1990, *Quantitative Analysis Report of Alternatives for Interim Remediation of the K-65 Silos*, International Technology Corporation, Knoxville, Tennessee.

IT, 1991a, *Fernald Groundwater Report*, U.S. Department of Energy, Oak Ridge Operations Office, December 1990.

IT, 1991b, *FINAL DRAFT -- Technical Position Paper No. 3, For the Remedial Investigation/Feasibility Study at the Feed Materials Production Center, Fernald Ohio., Risk Assessment Intake, Exposure and Transport Models and Parameters.* Prepared for United States Department of Energy, Fernald, Ohio.

James, A.C., 1988, "Lung Dosimetry", in *Radon and its Decay Products in Indoor Air*, W.W. Nazaroff and A.V. Nero Jr., eds., John Wiley & Sons, New York, NY.

Kathren, R.L. and J.R. Webber, (eds), 1988, *Ultrasensitive Techniques for Measurement of Uranium in Biological Samples and the Nephrotoxicity of Uranium*, NUREG/CP-0093, United States Nuclear Regulatory Commission, Washington, DC.

Katz, J.J., and E. Rabinovitch, 1951, *The Chemistry of Uranium: Part I, The Element, Its Binary and Related Compounds*, McGraw-Hill, Inc., N.Y.

Kigoshi, K., 1971, "Alpha-recoil Thorium-234: Dissolution Into Water and the Uranium-234/Uranium-238 Disequilibrium in Nature", *Science* 173:47-48.

Kocher, D.C. 1989, "Relationship Between Kidney Burden and Radiation Dose From Chronic Ingestion of U: Implications for Radiation Standards for the Public", *Health Physics* 57:9-15.

Krauskopf, K.B., 1979, *Introduction to Geochemistry,* 2nd ed., McGraw-Hill, New York.

Larsen, R.P., M.H. Bhattacharrya, R.D. Oldham et al., 1984, "Gastrointestinal Absorption and Retention of Plutonium and Uranium in the Baboon", in *Argonne National Laboratory, Environmental Research Division, Annual Report, July 1982-June 1983,* Argonne National Laboratory, Argonne, IL. ANL-83-100, pp. 51-60.

LaTouche Y.D., D.L. Willis, and O.I. Dawydiak, 1987, "Absorption and Biokinetics of U in Rats Following an Oral Administration of Uranyl Nitrate Solution, *Health Physics 53*: 147-162.

Lerch, N.K., W.F. Hale, and D.D. Lemaster, 1982, *Soil Survey of Hamilton County, Ohio,* Soil Conservation Service, National Cooperative Soil Survey, United States Department of Agriculture, Washington, D.C., 219 pp.

Liggett, R.W. 1989, "The Behavior and Chemical Toxicity of U in the Kidney: a Reassessment", *Health Physics* 57: 365-383.

Looney, B.B., M.W. Grant, and C.M. King, 1987, *Estimation of Geochemical Parameters for Assessing Subsurface Transport at the Savannah River Plant,* DPST-85-904, E.I. duPont de Nemours & Co., Savannah River Laboratory, Aiken, SC.

Lubin, J.H., Y.L. Qiao, P.R. Taylor, S.X. Yao, A. Schatzkin, B.L. Mao, J.Y. Rao, X.Z. Xian and Y.Y. Li, 1990, "Quantitative Evaluation of the Radon and Lung Cancer Association in a Case Control Study of Chinese Tin Miners", *Cancer Research* 50:174-180.

Luessenhop, A.J., J.C. Gallimore, W.H. Sweet, E.G. Struxness, and J. Robinson., 1958, "The Toxicity in Man of Hexavalent Uranium Following Intravenous Administration", *Am. J. Roentgenol.* 79:83-100.

Masuda, K. 1971, "Intake and Urinary Excretion of Uranium in Non-occupationally Exposed Persons, IV. Discussions on Dietary Intake and Urinary Excretion of Uranium", *Japanese Journal of Hygiene 26*(5): 447-450.

Maynard, E.A., W.L. Downs, and H.C. Hodge, 1953, "Oral Toxicity of Uranium Compounds", in *Pharmacology and Toxicology of Uranium Compounds,* C. Voegtlin and H.C. Hodge, eds., National Nuclear Energy Series, Div., VI, Vol I., McGraw Hill, New York, pp. 309-376.

Mays, C.W., R.E. Roland and A.F. Stehney, 1985, "Cancer Risk from the Lifetime Intake of Radium and Uranium Isotopes", *Health Physics* 48: 635-647.

McGuire, S.A. 1991, *Chemical Toxicity of Uranium Hexafluoride Compared to Acute Effects of Radiation,* NUREG-1391, United States Nuclear Regulatory Commission, Washington, D.C.

Meglen, R.R., 1985, *Dietary Intake Computations for Radionuclides,* Prepared from monitoring data provided by Science Applications International Corporation, Prepared by Robert R. Meglen, Center for Environmental Sciences, University of Colorado at Denver(cited in USEPA 1991b).

Moore, R., 1988, comments in *Ultrasensitive Techniques for Measurement of Uranium in Biological Samples and the Nephrotoxicity of Uranium*, Kathren, R.L. and Webber, J.R., eds., NUREG/CP-0093, United States Nuclear Regulatory Commission, Washington, DC., pp. 107-113.

Moore, R.E., C.F. Baes III, L.M. McDowell-Boyer, A.P Watson, F.O. Hoffman, J.C. Pleasant and C.W. Miller, 1979, *AIRDOS-EPA: A Computerized Methodology for Estimating Environmental Concentrations and Dose to Man From Airborne Releases of Radionuclides*, Reprint of ORNL-5532, EPA 520/1-79-009, United States Environmental Protection Agency Office of Radiation Programs, Washington, D.C. 20460.

Moore, R.H., ed., 1984, *Biokinetics and Analysis of Uranium in Man*, Proceedings of a Colloquium held at Richland, WA, August 8-9, 1984, Prepared by the Hanford Environmental Health Foundation for the United States Department of Energy, Available from NTIS, U.S. Department of Commerce, Springfield, VA.

Morris, S.C. 1990, *Baseline Risk Information for Formal Priority System Fiscal Year 1992 Interim System Application*, Brookhaven National Laboratory, Upton, NY 11973.

Morris, S.C. and A.F. Meinhold, 1988, *Report of Technical Support for the Hazardous Waste Remedial Program on Health and Environmental Risks of Inactive Hazardous Waste Sites*, BNL-42339, Informal Report, Brookhaven National Laboratory, Upton, NY 11973.

Myrich, T.E., B.A. Berven, and F.F. Haywood, 1983, "Determination of Concentrations of Selected Radionuclides in Surface Soil in the U.S.", *Health Physics*, 45(3).

NCP, 1990, *National Oil and Hazardous Pollution Contingency Plan*, 40 CFR 300.

NCRP, 1984, *Radiological Assessment: Predicting the Transport, Bioaccumulation and Uptake By Man of Radionuclides Released to The Environment*, NCRP Report No. 76, National Council on Radiation Protection and Measurements, Bethesda, MD.

NCRP, 1986, *Screening Techniques for Determining Compliance with Environmental Standards*, NCRP Commentary No. 3., National Council on Radiation Protection and Measurements, Bethesda, MD.

NOAA, 1984, *Local Climatological Data Annual Summary With Comparative Data, Cincinnati (Greater Cincinnati Airport), Ohio. National Climactic Data Center*, National Oceanic and Atmospheric Administration, Asheville, North Carolina.

Novikov. Y.V., 1972., "Hygenic Standardization of the Permissible Amount of Uranium in Drinking Water", *Gig. Sanit.* 37:13-17.

Novikov, Y.V., 1967, "The Effect of Low Concentrations of Uranium in Drinking Water on the Health of the Population", *Hyg. Sanit.* (translation of Gig. Sanit.) 33:340-344.

Novikov, Y.V., N.N. Pushkina and A.M. Tambovtseva, 1968, "The Effects on Man of Low Concentrations of Uranium", *Hyg. Sanit.* (translation of Gig. Sanit.) 33:340-344.

NRC, 1980, *The Effects on Populations of Exposure to Low Levels of Ionizing Radiation*, BEIR III, Committee on the Biological Effects of Ionizing Radiation, National Research Council, National Academy Press, Washington, DC, 1980.

NRC, 1988, *Health Risks of Radon and Other Internally Deposited Alpha Emitters*, BEIR IV, Committee on the Biological Effects of Ionizing Radiation, National Research Council, National Academy Press, Washington, D.C.

NRC, 1990, *Health Effects of Exposure to Low Levels of Ionizing Radiation*, BEIR V, Committee on the Biological Effects of Ionizing Radiation, National Research Council, National Academy Press, Washington, D.C.

NRC, 1991, *Comparative Dosimetry of Radon in Homes and Mines*, National Research Council, National Academy Press, Washington, D.C., 1991.

NUS, 1990, *Internal Correspondence, United States Department of Energy Environmental Survey Phase III Prioritization, Releases from the K-65 Silos Ranking Unit -- Feed Materials production Center (FMPC)*, Record Of Assumptions Memorandum 1, Revision 1.

Osmond, J.K., and J.B. Cowart, 1976, "The Theory and Uses of Natural Uranium Isotopic Variations in Hydrology," *Atomic Energy Review*, 14:(4):621-679.

Parks, B.S., 1992, *User's Guide for CAP88-PC, Version 1.0*, Office of Radiation Programs, United States Environmental Protection Agency, Las Vegas, NV., Report NO: 402-B-92-001.

Paustenbach, D.J., D.M. Meyer, P.J. Sheenan and V. Lau, 1991, "An Assessment and Quantative Uncertainty Analysis of the Health Risks to Workers Exposed to Chromium-Contaminated Soils", *Toxicol. Indust. Health* 7:159-196.

Pertlik, F., J.J.W. Rogers, and J.A.S. Adams, 1974, "Uranium", in *Handbook of Geochemistry*, Vol. 4, K.H. Wedepohl, ed, Springer-Verlag, Berlin.

Puskin, J.S., 1992, "An Analysis of the Uncertainties in Estimates of Radon-Induced Lung Cancer", *Risk Analysis*, 12:(2):277-285.

Puskin, J.S. and Y. Yang., 1988, "A Retrospective Look at Rn-induced Lung Cancer Mortality From the Viewpoint of a Relative Risk Model", *Health Physics* 54: 635-643.

Puskin, J.S. and C.B. Nelson, 1989, "EPA's Perspective on the Risks From Residential Radon Exposure", *J. Am. Pollut. Control Assoc.* 39:915-920.

Reeves, M., 1985, *Theory and Implementation for SWIFT II The Sandia Waste-Isolation Flow and Transport Model for Fractured Media*, Sandia National Laboratories, Albuquerque, New Mexico. 87185, United States Nuclear Regulatory Commission Report, NUREG/CR-3328.

Rich, B.L., S.L. Hinnefeld, C.R. Lagerquist, W.G. Mansfield, L.H. Munson, and E.R. Wagner, 1988, *Health Physics Manual of Good Practices for Uranium Facilities* (EGG-2530), Idaho National Engineering Laboratory, Idaho Falls, Idaho.

Richardson, A.H., 1964, "Thorium, Uranium, and Potassium in the Conway Granite, New Hampshire, U.S.A." in *The Natural Radiation Environment*, University of Chicago Press, pp. 39-50.

Roseberry, A.M., and D.E. Burmaster, 1992, "Lognormal Distributions for Water Intake by Children and Adults", *Risk Analysis*, 12:(1):99-104.

Rosholt, J.N., 1964, "Isotopic Fractionation of Uranium Related to Roll Features in Sandstone - Shirley Basin, Wyoming," *Economic Geology*, 59:570-585.

Rosholt, J.N., and M. Totsumoto, 1971, "Isotopic Composition of Thorium and Uranium in Apollo 12 Samples," *Proceedings of the 2nd Lunar Science Conference, Geochimica et Cosmochimica Acta*, Suppl. No. 2, p. 1577-1584.

Rowe, M.D., 1991, *Exposure to Radon Daughters in Plumes*, Brookhaven National Laboratory, Upton, New York, In Press.

SAIC, 1987, *Populations Within 5 miles of the Feed Materials Production Center at Fernald, Ohio*, Prepared for Westinghouse Materials Company of Ohio, Feed Materials Production Center Emergency Operations Center, Science Applications International Corporation, Oak Ridge Tennessee, November, 1987.

Sheppard, M.I., D.I. Beals, D.H. Thibault and P. O'Conner, 1984, *Nuclide Distribution Coefficients and Their Statistical Distribution*, AECL-8346, Atomic Energy of Canada Limited, Whiteshell Nuclear Research Establishment, Pinawa, Manitoba.

Sheppard, M.I., T.T. Vandergraaf, D.H. Thibault, and J. A. Keith Reid, 1983, "Technetium and Uranium: Sorption by and Plant Uptake From Peat and Sand", *Health Physics* 44:635-644.

Shor, R.W., and D.E. Fields, 1979, "The Fraction of Total Feed Composed of Fresh Forage, Fs, and the Fraction of the Year Fresh Forage is Utilized", in *A Statistical Analysis of Selected Parameters for Predicting Food Chain Transport and Internal Dose of Radionuclides*, Hoffman, F.O., and Baes, C.F., III, eds., NUREG/CR-1004, ORNL.NUREG/TM-282, Oak Ridge National Laboratory, Oak Ridge Tennessee.

Singh, N.P., D.P. Burleigh, H.M. Ruth, and M.E. Wrenn, 1990, "Daily U Intake in Utah Residents From Food and Drinking Water", *Health Physics* 59:333-337.

Singh, N.P., 1988, in *Ultrasensitive Techniques for Measurement of Uranium in Biological Samples and the Nephrotoxicity of Uranium*, R.L. Kathren, J.R. Webber eds., (NUREG/CP-0093), U.S. Nuclear Regulatory Commission, Washington, D.C. pp. 144-146.

Somayajula, B.L.K., C.A. Meghani and D.V. Borole, 1980, "Uranium in Drinking Waters: Rejection by Human Body", in, B. Patel, ed., *Management of the Environment*, Wiley Eastern, New Delhi, pp. 528-532.

Spencer, H., D. Osis, I.M. Fisenne, P.M. Perry and N.H. Harley, 1990, "Measured Intake and Excretion Patterns of Naturally Occurring ^{234}U, ^{238}U, and Calcium in Humans", *Radiation Research* 124:90-95.

Spoor, N.L., and J.B. Hursh, 1973, "Protection Criteria", in *Uranium, Plutonium and the Transplutonic Elements, Handbook of Experimental Pharmacology*, H.C. Hodge, J.N,. Stannard and J.B. Hursh, eds., Vol 36, pp 241-270, Springer-Verlag, New York.

Stannard, J.N., 1988, *Radioactivity and Health, A History* (DOE/RL/01830-T59), Pacific Northwest Laboratory, Richland, WA.

Sullivan, M. F., 1983, "Gut-related Studies of Radionuclide Toxicity" in *Pacific Northwest Laboratories Annual Report for 1982*, Pacific Northwest Laboratories, Richland, Washington, PNL 4600, pp. 95-99.

Sullivan, M.F.; P.S. Ruemmler, J.L. Ryan, and R.L Buschbom, 1986, "Influence of Oxidizing or Reducing Agents on Gastrointestinal Absorption of U, Pu, Am, Cm and Pm by Rats", *Health Physics* 50:223-232.

Syracuse Research Corp., 1990, *Toxicological Profile for Uranium*, Agency for Toxic Substances and Disease Registry, TP-90-29, Washington, D.C.

Szali, 1972, *The Use of Time: Daily Activities of Urban and Suburban Populations in Twelve Countries.* Paris: Mouton, The Hague.

Tracy, B.L., J.M. Quinn, J. Lahey, A.P. Gilman, K. Mancuso, A.P. Yagminas and D.C. Villeneuve, 1992, "Absorption and Retention of Uranium From Drinking Water by Rats and Rabbits", *Health Physics* 62:65-73, 1992.

Thun, M.J., D.B. Baker, K. Steenland et al., 1985, "Renal Toxicity in Uranium Mill Workers", *Scand. J. Work. Environ. Health* 11:83-90.

Till, J.E. and H.R. Meyer, 1983, *Radiological Assessment: A Textbook on Envirionmental Risk Analysis*, NUREG/CR-3332, ORNL-5968, United States Nuclear Regulatory Commission, Washington, D.C.

UNSCEAR, 1977, *Sources and Effects of Ionizing Radiation*, United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR), Report E.77.IX.1, United Nations, New York, NY, 1977.2

USDA, 1970, *Ohio Irrigation Guide*, United States Department of Agriculture Soil Conservation Service and Agricultural Research Service (cited in IT, 1991b).

USDA, 1979, *Soil Survey of Hamilton County, Ohio*, United States Department of Agriculture, Soil Conservation Service.

USDA, 1984, *Nutrient Intakes: Individuals in 48 States, Years 1977-1978*, Nationwide Food Consumption Survey 1977-1978, Report No. I-2. Human Nutrition Information Service, United States Department of Agriculture, Hyattsville, MD.

USDA, 1989, *Food and Nutrient Intakes of Individuals in One Day in the United States*, U.S. Department of Agriculture, Spring 1977, Nationwide Food Consumption Survey 1977-1978, Preliminary Report No. 2.

U.S. Department of Commerce, 1988, *American Housing Survey for the United States, 1985, Current Housing Reports* (H-150-85, 28-29), United States Department of Commerce, Bureau of Census and U.S. Department of Housing and Urban Development.

U.S. Department of Commerce, 1989, *American Housing Survey for the United States, 1987, Current Housing Reports* (H-150-87, 52-53), United States Department of Commerce, Bureau of Census and U.S. Department of Housing and Urban Development.

USDOE, 1987, *Remedial Investigation and Feasibility Study, Feed Materials Production Center, Fernald, Ohio, Task 1 Report: Description of Current Situation*. U.S. Department of Energy, Oak Ridge Operations.

USDOE, 1988, *Environmental Survey Preliminary Summary Report of the Defense Production Facilities*, DOE/EH-0072, Environment, Safety, and Health, Office of Environmental Audit. U.S. Department of Energy, Washington, D.C.

USDOE, 1990a, *Remedial Investigation Report for Operable Unit 4. Task 6 Report. Draft Final*, Feed Materials Production Center, Fernald, Ohio, U.S. Department of Energy.

USDOE, 1990b, *Engineering Evaluation/Cost Analysis South Plume. Feed Materials Production Center, Fernald Ohio*, United States Department of Energy. FMPC-0003-6.

USDOE, 1990c, *Feed Materials Production Center Annual Environmental Report for Calendar Year 1989*, U.S. Department of Energy.

USDOE, 1990d, *Remedial Investigation/Feasibility Study - Ground Water Report, Feed Materials Production Center, Fernald, Ohio*, United States Department of Energy, Oak Ridge, TN.

USDOE, 1990e, *Draft Environmental Impact Statement. FEMP Renovation and Site Evaluation*. United States Department of Energy, G-000-107.3/441

USEPA, 1984, *An Estimation of the Daily Average Food Intake by Age and Sex for Use in Assessing the Radionuclide Intake of Individuals in the General Population*, Office of Radiation Programs, United States Environmental Protection Agency, Washington, D.C., EPA/520/1-84-021.

USEPA, 1986, *Radon Reduction Methods: A Homeowner's Guide*, EPA-86-005 United States Environmental Protection Agency, Washington, D.C.

USEPA, 1989, *Risk Assessment Guidance for Superfund. Volume I Human Health Evaluation Manual (Part A), Interim Final,* EPA/540/1-89/002, Office of Emergency and Remedial Response, United States Environmental Protection Agency, Washington, DC.

USEPA, 1990a, *Occurrence and Exposure Assessment for Uranium in Public Drinking Water Supplies (draft)*, Prepared by Wade Miller Associates, Inc. (April 26, 1990)

USEPA, 1990b, *Exposure Factors Handbook*, Office of Health and Environmental Assessment, United States Environmental Protection Agency, Washington, D.C., EPA/600/8-89/043.

USEPA, 1991a, *Occurrence and Exposure Assessment for Uranium in Public Drinking Water Supplies. Revision 2.* Prepared by Wade Miller Associates, Inc.

USEPA, 1991b, *Drinking Water Criteria Document for Uranium, DRAFT,* Criteria and Standards Division, Office of Drinking Water, U.S. Environmental Protection Agency, Washington, D.C., 20460.

USNRC, 1977, *Regulatory Guide 1.109: Calculation of Annual Doses to Man from Routine Releases of Reactor Effluents for the Purpose of Evaluating Compliance with 10CFR Part 50, Appendix I.*

Varchol, B.D., 1990, *Statistical Evaluation of Homeowner Wells,* Westinghouse Materials Company of Ohio, WMCO:R:(EM):90-0296.

Wakskol and Yasov, 1974, "$^{234}U/^{238}U$ Disequilibrium in Water of the Judea Group (Cenomianian-Turonian) Aquifer in Galilee, Northern Israel," in *Isotope Techniques in Groundwater Hydrology*, International Atomic Energy Agency Proceedings, Vienna, SM-182/34.

Welford, G.A., and R. Baird, 1967, "Uranium Levels in Human Diet and Biological Materials", *Health Physics* 13:1321-1324.

Weston, 1987, *Characterization Investigation Study: Volume 3, Radiological Survey of Surface Soils,* Roy. F. Weston for United States Department of Energy, Feed Materials Production Center, Fernald, Ohio.

Wrenn, M.E., P.W. Durbin, B. Howard, J. Lipsztein, J. Rundo, E.T. Still and D.L. Willis., 1985, "Metabolism of Ingested U and Ra", *Health Physics* 48(5):601-633.

Wrenn, M.E., P.W. Durbin, D.L.Willis and N.P. Singh, 1987, "The Potential Toxicity of Uranium in Water", *American Water Works Association J.* 79: 117-184.

Wrenn, M.E., N.P.Singh, H.Ruth, M.L.Rallison, and D.P.Burleigh, 1990, "Gastrointestinal Absorption of Soluble Uranium From Drinking Water in Humans", in Cothern, C.R. and Rebers, R.A. (eds), *Radon, Radium and Uranium in Drinking Water*, Lewis Publishers, Chelsea, MI, pp. 159-163.

Wrenn, M.E., N.P. Singh, H.Ruth, M.L.Rallison and D.P.Burleigh, 1989, "Gastrointestinal Absorption of Soluble U from Drinking Water by Man", *Radiation Protection Dosimetry* 26: 119-122.

Yamamoto, T., K. Masuda and N. Onishi, 1968, "Studies on Environmental Contamination by Uranium. 1. Environmental Survey of Uranium in Kamisaibara Village, Okayama" *J. Rad. Res.* (Japan) 9:92-99.

Zach, R., and K.R. Mayoh, 1984, "Soil Ingestion by Cattle: A Neglected Pathway", *Health Physics*, 46(2):426-431.

APPENDIX A

COMPARISON OF URANIUM TOXICITY ASSESSMENT TO USEPA APPROACH

The USEPA recently proposed regulations for limits on the concentration of uranium in drinking water (Federal Register, 1991). The limit (20 μg/L) was based on the toxic effects of uranium to the kidney. Here we compare on a point-by-point basis the analysis supporting the USEPA limit with the analysis presented in this report. As will be noted, the underlying philosophies for the two approaches differ. The USEPA selects single values for the parameters that make up their analysis -- these values are usually conservatively high, although sometimes USEPA terms them "best estimates." In our analysis we attempt to explicitly include uncertainty by characterizing parameters as probability distributions.

Metabolic/Pharmacokinetic Parameters

In the USEPA approach, the organ doses for uranium were modeled for the carcinogenic effects of uranium but not for the toxicologic effects. In the toxicity assessment, they compare administered dose to rabbits directly with toxic effect, extrapolating to humans on a mg U/kg body weight basis. We use a pharmacokinetic model for both radiocarcinogenic and toxic effects. In the toxicity assessment, we compare kidney concentration of uranium with a threshold level. We can calculate backwards, from the toxic threshold concentration in the kidney, through the pharmacokinetic model and the amount of water drunk per day, to obtain the equivalent threshold level in drinking water that can be compared with the USEPA value.

Uptake (gut to blood)

We followed the approach of Wrenn et al. (1985) and LaTouche et al. (1987). We included more recent reports and analyzed them in three groups: overnight fasting (animals and humans), ad lib feeding, and human studies with environmental doses. The ad lib feeding group had considerably lower uptake, but the two other groups were similar. We based our estimate on the environmental group and characterized the results as lognormal (geometric mean = 1.03, standard deviation = 1.78, 95% bounds 0.0033 to 0.0317 and total range 0-0.05).

USEPA used a value of 0.05, which they called a "best estimate." They review values from the literature ranging from < 0.01 to 0.30 (Federal Register, 1991, p. 33076; USEPA, 1991, pp. IV-1 to IV-12). Although not noted by USEPA, the original source of the value 0.05 was ICRP (1979), which suggested a range of 0.005 to 0.05. Somewhere along the way, the value went from the upper end of a range to a best estimate.

The USEPA estimate is the maximum of our range, and above the 95% confidence interval of the distribution we use. Examination of the literature, however, shows that all of the absorption values cited by USEPA that are over 5% are inappropriate for inclusion in the analysis. Some of these values come from studies in which the authors themselves estimated much lower absorption rates. A committee of internationally known experts, commissioned by USEPA to review the metabolism of ingested uranium, analyzed the literature available in the mid-1980s and recommended a value of 1.4%. They noted: "None of the available experimental or environmental data support a fractional U absorption greater than

about 5%, even at intakes of the order of 1 to 2 mg/day for Reference Man. ..." (Wrenn et al., 1985, pp. 626-627). The studies cited by USEPA as indicating absorption rates above 5% are discussed individually below:

1. Spencer et al., 1990 (26%). The authors' own conclusion in the paper was that, for total intake, absorption of uranium was 1.5% (looking at water intake alone, the authors estimate an intake of 5%. USEPA's estimate was based on fecal excretion (assume [intake-fecal excretion] / intake = absorption). This involves a very high error rate. The authors examine this approach, but state that, "Although the net absorption of the two uranium isotopes determined from the intake and fecal excretions ... averaging 26 and 23% respectively, the large error for the balance ... includes zero net absorption. *It is well known that it is not possible to determine uptake of trace elements quantitatively in this manner...*" [emphasis added]. As a minor point, it can be noted that, although Spencer et al. made the calculation for ^{234}U and ^{238}U separately, USEPA reported only the higher number, even though it applied to ^{234}U, which constituted only about 1/10,000th of the total mass concentration of the intake.

2. Somayajulo et al., 1980 (26%): In USEPA's review of this study (USEPA, 1991 p. IV-7) we learn that this is a study of one man. Intake estimates were based on measurements of food in the cafeteria where he usually ate, not on what he specifically ate. The number of days for which urine and feces were collected was not given. Blood levels of uranium were measured, not in the subject, but in another man who presumably consumed the same amount of uranium. The 26% figure was determined by the ratio of food and water input to the urine and feces output as was the case with the Spencer et al. study above. USEPA notes that if the calculation were made using the Wrenn et al. method, the absorption would be 3.8%.

3. Hamilton, 1972 (31%). This study combined measured content of sampled foods in the UK with an estimated European diet to estimate intake (uranium from drinking water was not considered). Concentrations of uranium in various body organs were measured. Hursh and Spoor (1973) combined these data with urinary excretion measurements from a different study to estimate absorption. The potential error involved in applying "market basket" studies of uranium in food with organ and urine samples of different populations is so great that we believe such studies should be rejected in the face of more detailed studies on specific subjects.

4. Welford and Baird, 1967 (0.7%, 12% and 7.7%). Similar to Hamilton (1972, above) this is a market basket study of uranium in food combined with excretion data from other studies. In one case (12%) this was Hursh and Spoor (1973), the same analysis from which the estimates of the Hamilton data came. The second analysis of the Welford and Baird data was done by Wrenn et al. (1985). The latter authors rejected the Hursh and Spoor calculation, produced their own (7.7%), but in the end excluded it from their overall analysis. It is interesting that USEPA's own re-calculation of the Welford and Baird data yielded a value of only 0.7%. Again, however, we believe that the potential error involved in applying "market basket" studies of uranium in food with organ and urine samples of different populations is so great that we believe such studies should be rejected in the face of more detailed studies on specific subjects.

The effect of this on the results is that the arithmetic mean of our estimate for uptake is 1.22% compared to USEPA's estimate of 5%, about a factor of four higher.

Blood to Kidney

We take the general approach of Wrenn et al. (1985), drawing on their results but supplementing them with new results from rats and rabbits by Tracy et al. (1992). The latter authors report that, although there are differences in uptake from the gut, once in the blood uranium uptake and retention by the kidney are similar among rats, rabbits, and humans. We model the two distributions as lognormal distributions with mean and standard deviations taken directly from the Wrenn et al. and Tracy et al. data. The 95% confidence intervals of our distributions for kidney uptake and biological half-life are 1.84-21.4% and 4.70-26.2 days, respectively. We assume the two distributions are inversely correlated with a correlation coefficient of 0.7.

USEPA uses the ICRP (1979) model with two elimination rates. Fractional uptake from blood to kidney is assumed to be 12%, of which 99.6% is retained with a 6 day half time and 0.4% with a 1500 day half-time (USEPA, 1991, p. IV-23). The 12% uptake is at the 85th percentile level of our distribution.

Model Formulation Uncertainties

Although uncertainty in individual parameters of the intake-to-kidney concentration model has been characterized, there is no direct method to estimate uncertainties associated with the specific formulation of the model used. As a practical approach to account for this uncertainty, we assume that the range of uncertainty from this source is about a factor of two. We achieve this by specifying a lognormal distribution with geometric mean 1 and geometric standard deviation 1.4.

Water Intake

We took the distribution of water intake from Ershow and Cantor (1989), based on the 1977-1978 Nationwide Food Consumption Survey of the U.S. Department of Agriculture (USDA, 1984). We used a lognormal distribution with arithmetic mean and standard deviation of 1.203 and 0.689 l/d representing intake of tap water in the mid-west.

USEPA uses a value of 2 l/d. This is at the 89th percentile of our distribution and introduces a mean overestimate of 1.7.

Kidney Concentration Threshold

Wrenn et al. (1985) suggest a threshold of 1 mg U/g. Kocher (1989) applied a safety factor of 10 to this to protect maximally exposed individuals in the public, using 0.1 mg U/g. Several reports of animal experiments demonstrate effects in the range 0.5-1 mg U/g. These effects are perhaps not as severe as those on which the original occupational standard of 3 mg U/g was based, but may be more appropriate end-points for chronic exposure to the public, whose members do not have the additional protection of routine medical surveillance and bioassay. A range of 0.1-1 mg U/g appears appropriate. The threshold value was characterized as a Weibull distribution with location parameter 0.1 to reflect a lower limit of 0.1 mg U/g, a scale parameter of 0.6 and shape parameter of 4. These produce a maximum value of about 1 mg U/g with a mean of 0.6.

EPA bases their toxicity threshold on a 1949 study of rabbits which was clearly a preliminary study, incompletely described in the original source (Maynard and Hodge, 1949). They determine the threshold level to be an intake of 2.8 mg

uranium/kg/day, higher than any other estimate found in the literature. For a 70 kg adult, this is 196 mg/day. Back-calculating from our kidney-concentration threshold, this USEPA value is at the 98th percentile level. We believe this is consistent with an upper bound estimate. We note that USEPA imposes a further safety factor of 1000 on this value.

Water Concentration Toxic Threshold

Results derived from backcalculating from our kidney uranium concentration threshold distribution to daily intake and concentration in drinking water are shown in Table A-1. USEPA's LOAEL (Lowest Adverse Effect Level) value of 196 mg/L in water is at the 97.5 percentile level of the distribution. It produces a mean overestimate of 3.5.

Table A-1. Toxic threshold estimates.

	Daily Uranium Intake (mg/kg/d)	Water Concentration (mg/L)
Mean	0.73	56.7
Upper 95%	1.93	165.

Safety Factor

USEPA introduces a safety factor of 1000 on their water concentration toxic threshold. This is presumably due to its being a LOAEL rather than a NOAEL (No Adverse Effect Level). They ignore NOAEL effects found in other studies. Using the USEPA guidelines for selection of a safety factor, the NAS/NRC Safe Drinking Water Committee selected an uncertainty factor of 100 (NAS, 1983, p. 96). The Committee on Metabolism and Dosimetry of High LET Radionuclides for the National Workshop on Radioactivity in Drinking Water (Wrenn et al., 1985), recommended, "that based on the NAS definition, U should be assigned an uncertainty factor of 10-100" (pp. 612-632). It then selected 50 as a factor "that should provide a high margin of safety" (pp. 632-633).

We do not apply an arbitrary safety factor, but express uncertainties explicitly in the input and results, allowing the degree of safety to be chosen as an explicit level of confidence.

Conclusions

USEPA selected values for each parameter that fall in the upper end of a distribution designed to explicitly show the range of uncertainty. The combination of these can lead to estimates considerably far out at the extremes of the resulting distribution. The largest difference, however, comes from adding a safety factor of 1000 on top of all this.

REFERENCES

Ershow, A.G. and K.P. Cantor, 1989, *Total Water and Tapwater Intake in the United States: Population-Based Estimates of Quantities and Sources*, Life Sciences Research Office, Federation of American Societies for Experimental Biology, Rockville, MD.

Federal Register, 1991. Environmental Protection Agency, National Primary Drinking Water Regulations; Radionuclides", *40CFR Parts* 141, 142. 56:138.

Hamilton, E. I., 1972, "The Concentration of Uranium in Man and His Diet", *Health Physics* 22:149-153.

Hursh, J.B. and N.L. Spoor, 1973, "Data on Man", in *Uranium Plutonium Transplutonic Elements*, H.C. Hodge, J.N. Stannard and J.B. Hursh, eds., Springer-Verlag, New York, pp. 197-239.

ICRP, 1979, "Limits for Intakes of Radionuclides by Workers", International Commission on Radiological Protection, ICRP Publication 30, part 1. *Annals of the ICRP Committee 2*, No. 3/4, Pergamon Press, Oxford.

Kocher, D.C. 1989, "Relationship Between Kidney Burden and Radiation Dose From Chronic Ingestion of U: Implications for Radiation Standards for the Public", *Health Physics* 57:9-15.

LaTouche Y.D., D.L., Willis, and O.I. Dawydiak, 1987, "Absorption and Biokinetics of U in Rats Following an Oral Administration of Uranyl Nitrate Solution, *Health Physics 53*: 147-162.

Maynard, E.A. and H.C. Hodge, 1949, "Study of Toxicity of Various Uranium Compounds When Fed to Experiment Animals", in *Pharmacology and Toxicology of Uranium Compounds*, C. Voegtlin and H.C. Hodge, eds., National Nuclear Energy Series, Div. VI, Vol. 1, Mcgraw Hill, New York, pp. 309-376.

NAS, 1983, *Drinking Water and Health*, National Academy of Sciences, Vol. 5, National Academy Press, Washington, D.C.

Somayajula, B.L.K., C.A. Meghani and D.V. Borole, 1980, "Uranium in Drinking Waters: Rejection by Human Body", in, *Management of the Environment*, B. Patel, ed., Wiley Eastern, New Delhi, pp. 528-532.

Spencer, H., D. Osis, I.M. Fisenne, P.M. Perry and N.H. Harley, 1990, "Measured Intake and Excretion Patterns of Naturally Occurring ^{234}U, ^{238}U, and Calcium in Humans", *Radiation Research* 124:90-95.

Tracy, B.L., J.M. Quinn, J. Lahey, A.P. Gilman, K. Mancuso, A.P. Yagminas and D.C. Villeneuve, 1992, "Absorption and Retention of Uranium From Drinking Water by Rats and Rabbits", *Health Physics* 62:65-73.

USDA, 1984, *Nutrient Intakes: Individuals in 48 States, Years 1977-1978*, Nationwide Food Consumption Survey 1977-1978, Report No. I-2. Human Nutrition Information Service, United States Department of Agriculture, Hyattsville, MD.

USEPA, 1991, *Drinking Water Criteria Document for Uranium, DRAFT*, Criteria and Standards Division, Office of Drinking Water, U.S. Environmental Protection Agency, Washington, D.C., 20460.

Welford, G.A., and R. Baird, 1967, "Uranium levels in Human Diet and Biological Materials", *Health Physics* 13:1321-1324.

Wrenn, M.E., P.W. Durbin, B. Howard, J. Lipsztein, J. Rundo, E.T. Still and D.L. Willis., 1985, "Metabolism of Ingested U and Ra", *Health Physics* 48(5): 601-633.

APPENDIX B
TABLES -- PREDICTED URANIUM CONCENTRATIONS IN WATER, FOOD AND MILK

Table B-1. Predicted uranium concentrations in ground water at five representative wells for the stop source and continue source scenarios.

well	year	STOP SOURCE water (ug/l)	CONTINUE SOURCE water (ug/l)
1	0	241.19	241.19
	10	107.1	107.1
	20	40.52	42.06
	30	15.04	21.88
	40	5.47	16.12
	50	2.12	14.67
	60	1.05	14.01
	70	0.5	13.82
2	0	0.14	0.14
	10	1.379	1.379
	20	5.67	5.67
	30	11.91	11.97
	40	16.85	17.24
	50	17.94	19.45
	60	16.09	20
	70	12.22	18.96
3	0	0	0
	10	0	0
	20	0	0
	30	0	0
	40	0	0
	50	0	0
	60	0.0001	0.0001
	70	0.001	0.01
4	0	8.6	8.6
	10	25.25	25.25
	20	15.68	15.71
	30	6.81	6.77
	40	2.32	2.83
	50	0.72	1.71
	60	0.33	1.25
	70	0.13	1.1
5	0	2.44	2.44
	10	21.58	21.58
	20	59.03	58.95
	30	63.61	63.1
	40	43.5	43.15
	50	25.82	29.48
	60	18.38	20.14
	70	12.53	15.66

Table B-2. Predicted uranium concentrations in ground water at four centerline wells for the stop source and continue source scenarios.

well	year	STOP SOURCE water (ug/l)	CONTINUE SOURCE water (ug/l)
1	0	8.3	8.3
	10	30.9	30.9
	20	45.4	45.4
	30	44.7	45.9
	40	34.6	42.5
	50	21.9	39.5
	60	13.6	36.4
	70	7.3	34.5
2	0	1.9	1.9
	10	11.3	11.3
	20	25	25
	30	29	29
	40	25.7	25.8
	50	21.1	23
	60	17.7	21
	70	13.5	19.9
3	0	4.2	4.1
	10	26.3	26.3
	20	56.5	56.5
	30	54.8	54.3
	40	37.1	36.9
	50	23.2	26.3
	60	17.3	19.3
	70	12.3	16
4	0	0.6	0.6
	10	12.9	12.9
	20	57.1	56.9
	30	78.1	77.8
	40	57.6	56.8
	50	31.5	36.7
	60	20.1	21.8
	70	11.7	14.4

Table B-3 - Estimated concentrations in water, food and milk for five residential wells -- data for toxicity assessment.

		STOP SOURCE			CONTINUE SOURCE		
well	year	water (ug/l)	food (ug/kg)	milk (ug/l)	water (ug/l)	food (ug/kg)	milk (ug/l)
r1	0	242.1	1473.227	11.30618	242.1	1473.227	11.30618
	10	108.0	657.3166	6.478936	108.0	657.3166	6.478936
	20	41.4	252.1906	4.082056	43.0	261.5612	4.137496
	30	16.0	97.14992	3.164776	22.8	138.7699	3.411016
	40	6.4	38.91838	2.820256	17.0	103.7215	3.203656
	50	3.0	18.5343	2.699656	15.6	94.89854	3.151456
	60	2.0	12.02356	2.661136	14.9	90.88257	3.127696
	70	1.4	8.676925	2.641336	14.7	89.72646	3.120856
r2	0	1.1	6.486397	2.628376	1.1	6.486397	2.628376
	10	2.3	14.02546	2.67298	2.3	14.02546	2.67298
	20	6.6	40.13534	2.827456	6.6	40.13534	2.827456
	30	12.8	78.10449	3.052096	12.9	78.46958	3.054256
	40	17.8	108.1634	3.229936	18.2	110.5365	3.243976
	50	18.9	114.7958	3.269176	20.4	123.9839	3.323536
	60	17.0	103.539	3.202576	20.9	127.3305	3.343336
	70	13.1	79.99078	3.063256	19.9	121.0023	3.305896
r3	0	0.9	5.634525	2.623336	0.9	5.634525	2.623336
	10	0.9	5.634525	2.623336	0.9	5.634525	2.623336
	20	0.9	5.634525	2.623336	0.9	5.634525	2.623336
	30	0.9	5.634525	2.623336	0.9	5.634525	2.623336
	40	0.9	5.634525	2.623336	0.9	5.634525	2.623336
	50	0.9	5.634525	2.623336	0.9	5.634525	2.623336
	60	0.9	5.635133	2.62334	0.9	5.635133	2.62334
	70	0.9	5.64061	2.623372	0.9	5.695373	2.623696
r4	0	9.5	57.9638	2.932936	9.5	57.9638	2.932936
	10	26.2	159.2757	3.532336	26.2	159.2757	3.532336
	20	16.6	101.0442	3.187816	16.6	101.2267	3.188896
	30	7.7	47.07201	2.868496	7.7	46.82862	2.837056
	40	3.2	19.75126	2.706856	3.8	22.85451	2.725216
	50	1.6	10.01558	2.649256	2.6	16.03953	2.684896
	60	1.3	7.642509	2.635216	2.2	13.24052	2.668336
	70	1.1	6.425549	2.628016	2.0	12.3278	2.662936
r5	0	3.4	20.48144	2.711176	3.4	20.48144	2.711176
	10	22.5	136.9445	3.400216	22.5	136.9445	3.400216
	20	60.0	364.8203	4.748416	59.9	364.3335	4.745536
	30	64.5	392.6887	4.913296	64.0	389.5854	4.894936
	40	44.4	270.3233	4.189336	44.1	268.1936	4.176736
	50	26.7	162.7441	3.552856	30.4	185.0144	3.684616
	60	19.3	117.4731	3.285016	21.1	128.1824	3.348376
	70	13.5	81.87707	3.074416	16.6	100.9225	3.187096

Table B-4 - Estimated concentrations in water, food and milk for five residential wells -- data for cancer risk assessment.

well	year	STOP SOURCE water (ug/l)	STOP SOURCE food (mg/kg)	STOP SOURCE milk (mg/l)	CONTINUE SOURCE water (ug/l)	CONTINUE SOURCE food (mg/kg)	CONTINUE SOURCE milk (mg/l)
1	0	241.2	1.468	0.009	241.2	1.468	0.009
	1	207.7	1.264	0.007	207.7	1.264	0.007
	2	207.7	1.264	0.007	207.7	1.264	0.007
	3	207.7	1.264	0.007	207.7	1.264	0.007
	4	207.7	1.264	0.007	207.7	1.264	0.007
	5	174.1	1.060	0.006	174.1	1.060	0.006
	6	140.6	0.856	0.005	140.6	0.856	0.005
	7	140.6	0.856	0.005	140.6	0.856	0.005
	8	140.6	0.856	0.005	140.6	0.856	0.005
	9	140.6	0.856	0.005	140.6	0.856	0.005
	10	107.1	0.652	0.004	107.1	0.652	0.004
	11	90.5	0.550	0.003	90.8	0.553	0.003
	12	90.5	0.550	0.003	90.8	0.553	0.003
	13	90.5	0.550	0.003	90.8	0.553	0.003
	14	90.5	0.550	0.003	90.8	0.553	0.003
	15	73.8	0.449	0.003	74.6	0.454	0.003
	16	57.2	0.348	0.002	58.3	0.355	0.002
	17	57.2	0.348	0.002	58.3	0.355	0.002
	18	57.2	0.348	0.002	58.3	0.355	0.002
	19	57.2	0.348	0.002	58.3	0.355	0.002
	20	40.5	0.247	0.001	42.1	0.256	0.002
	21	34.2	0.208	0.001	37.0	0.225	0.001
	22	34.2	0.208	0.001	37.0	0.225	0.001
	23	34.2	0.208	0.001	37.0	0.225	0.001
	24	34.2	0.208	0.001	37.0	0.225	0.001
	25	27.8	0.169	0.001	32.0	0.195	0.001
	26	21.4	0.130	0.001	26.9	0.164	0.001
	27	21.4	0.130	0.001	26.9	0.164	0.001
	28	21.4	0.130	0.001	26.9	0.164	0.001
	29	21.4	0.130	0.001	26.9	0.164	0.001
	30	15.0	0.092	0.001	21.9	0.133	0.001
	31	12.6	0.077	0.000	20.4	0.124	0.001
	32	12.6	0.077	0.000	20.4	0.124	0.001
	33	12.6	0.077	0.000	20.4	0.124	0.001
	34	12.6	0.077	0.000	20.4	0.124	0.001
	35	10.3	0.062	0.000	19.0	0.116	0.001
	36	7.9	0.048	0.000	17.6	0.107	0.001
	37	7.9	0.048	0.000	17.6	0.107	0.001
	38	7.9	0.048	0.000	17.6	0.107	0.001
	39	7.9	0.048	0.000	17.6	0.107	0.001
	40	5.5	0.033	0.000	16.1	0.098	0.001
	41	4.6	0.028	0.000	15.8	0.096	0.001
	42	4.6	0.028	0.000	15.8	0.096	0.001
	43	4.6	0.028	0.000	15.8	0.096	0.001
	44	4.6	0.028	0.000	15.8	0.096	0.001
	45	3.8	0.023	0.000	15.4	0.094	0.001
	46	3.0	0.018	0.000	15.0	0.091	0.001
	47	3.0	0.018	0.000	15.0	0.091	0.001
	48	3.0	0.018	0.000	15.0	0.091	0.001
	49	3.0	0.018	0.000	15.0	0.091	0.001
	50	2.1	0.013	0.000	14.7	0.089	0.001
	51	1.9	0.011	0.000	14.5	0.088	0.001
	52	1.9	0.011	0.000	14.5	0.088	0.001
	53	1.9	0.011	0.000	14.5	0.088	0.001

	54	1.9	0 011	0 000	14.5	0.088	0 001
	55	1.6	0 010	0.000	14.3	0.087	0.001
	56	1.3	0 008	0 000	14.2	0.086	0 001
	57	1.3	0 008	0.000	14.2	0.086	0 001
	58	1.3	0.008	0.000	14.2	0.086	0.001
	59	1.3	0.008	0.000	14.2	0.086	0.001
	60	1.1	0.006	0.000	14.0	0.085	0.001
	61	0.9	0.006	0.000	14.0	0.085	0.001
	62	0.9	0.006	0.000	14.0	0.085	0.001
	63	0.9	0.006	0.000	14.0	0.085	0.001
	64	0.9	0.006	0.000	14.0	0.085	0.001
	65	0.8	0.005	0.000	13.9	0.085	0.001
	66	0.6	0.004	0.000	13.9	0.084	0.000
	67	0.6	0.004	0.000	13.9	0.084	0.000
	68	0.6	0.004	0.000	13.9	0.084	0.000
	69	0.6	0.004	0.000	13.9	0.084	0.000
	70	0.5	0.003	0.000	13.8	0.084	0.000
2	0	0.1	0.001	0.000	0.1	0.001	0.000
	1	0.4	0.003	0.000	0.4	0.003	0.000
	2	0.4	0.003	0.000	0.4	0.003	0.000
	3	0.4	0.003	0.000	0.4	0.003	0.000
	4	0.4	0.003	0.000	0.4	0.003	0.000
	5	0.8	0.005	0.000	0.8	0.005	0.000
	6	1.1	0.007	0.000	1.1	0.007	0.000
	7	1.1	0.007	0.000	1.1	0.007	0.000
	8	1.1	0.007	0.000	1.1	0.007	0.000
	9	1.1	0.007	0.000	1.1	0.007	0.000
	10	1.4	0.008	0.000	1.4	0.008	0.000
	11	2.5	0.015	0.000	2.5	0.015	0.000
	12	2.5	0.015	0.000	2.5	0.015	0.000
	13	2.5	0.015	0.000	2.5	0.015	0.000
	14	2.5	0.015	0.000	2.5	0.015	0.000
	15	3.5	0.021	0.000	3.5	0.021	0.000
	16	4.6	0.028	0.000	4.6	0.028	0.000
	17	4.6	0.028	0.000	4.6	0.028	0.000
	18	4.6	0.028	0.000	4.6	0.028	0.000
	19	4.6	0.028	0.000	4.6	0.028	0.000
	20	5.7	0.035	0.000	5.7	0.035	0.000
	21	7.2	0.044	0.000	7.2	0.044	0.000
	22	7.2	0.044	0.000	7.2	0.044	0.000
	23	7.2	0.044	0.000	7.2	0.044	0.000
	24	7.2	0.044	0.000	7.2	0.044	0.000
	25	8.8	0.053	0.000	8.8	0.054	0.000
	26	10.4	0.063	0.000	10.4	0.063	0.000
	27	10.4	0.063	0.000	10.4	0.063	0.000
	28	10.4	0.063	0.000	10.4	0.063	0.000
	29	10.4	0.063	0.000	10.4	0.063	0.000
	30	11.9	0.072	0.000	12.0	0.073	0.000
	31	13.1	0.080	0.000	13.3	0.081	0.000
	32	13.1	0.080	0.000	13.3	0.081	0.000
	33	13.1	0.080	0.000	13.3	0.081	0.000
	34	13.1	0.080	0.000	13.3	0.081	0.000
	35	14.4	0.087	0.001	14.6	0.089	0.001
	36	15.6	0.095	0.001	15.9	0.097	0.001
	37	15.6	0.095	0.001	15.9	0.097	0.001
	38	15.6	0.095	0.001	15.9	0.097	0.001
	39	15.6	0.095	0.001	15.9	0.097	0.001
	40	16.9	0.103	0.001	17.2	0.105	0.001
	41	17.1	0.104	0.001	17.8	0.108	0.001
	42	17.1	0.104	0.001	17.8	0.108	0.001
	43	17.1	0.104	0.001	17.8	0.108	0.001

	44	17.1	0.104	0.001	17.8	0.108	0.001
	45	17.4	0.106	0.001	18.3	0.112	0.001
	46	17.7	0.108	0.001	18.9	0.115	0.001
	47	17.7	0.108	0.001	18.9	0.115	0.001
	48	17.7	0.108	0.001	18.9	0.115	0.001
	49	17.7	0.108	0.001	18.9	0.115	0.001
	50	17.9	0.109	0.001	19.5	0.118	0.001
	51	17.5	0.106	0.001	19.6	0.119	0.001
	52	17.5	0.106	0.001	19.6	0.119	0.001
	53	17.5	0.106	0.001	19.6	0.119	0.001
	54	17.5	0.106	0.001	19.6	0.119	0.001
	55	17.0	0.104	0.001	19.7	0.120	0.001
	56	16.6	0.101	0.001	19.9	0.121	0.001
	57	16.6	0.101	0.001	19.9	0.121	0.001
	58	16.6	0.101	0.001	19.9	0.121	0.001
	59	16.6	0.101	0.001	19.9	0.121	0.001
	60	16.1	0.098	0.001	20.0	0.122	0.001
	61	15.1	0.092	0.001	19.7	0.120	0.001
	62	15.1	0.092	0.001	19.7	0.120	0.001
	63	15.1	0.092	0.001	19.7	0.120	0.001
	64	15.1	0.092	0.001	19.7	0.120	0.001
	65	14.2	0.086	0.001	19.5	0.119	0.001
	66	13.2	0.080	0.000	19.2	0.117	0.001
	67	13.2	0.080	0.000	19.2	0.117	0.001
	68	13.2	0.080	0.000	19.2	0.117	0.001
	69	13.2	0.080	0.000	19.2	0.117	0.001
	70	12.2	0.074	0.000	19.0	0.115	0.001
3	0	0.0	0.000	0.000	0.0	0.000	0.000
	1	0.0	0.000	0.000	0.0	0.000	0.000
	2	0.0	0.000	0.000	0.0	0.000	0.000
	3	0.0	0.000	0.000	0.0	0.000	0.000
	4	0.0	0.000	0.000	0.0	0.000	0.000
	5	0.0	0.000	0.000	0.0	0.000	0.000
	6	0.0	0.000	0.000	0.0	0.000	0.000
	7	0.0	0.000	0.000	0.0	0.000	0.000
	8	0.0	0.000	0.000	0.0	0.000	0.000
	9	0.0	0.000	0.000	0.0	0.000	0.000
	10	0.0	0.000	0.000	0.0	0.000	0.000
	11	0.0	0.000	0.000	0.0	0.000	0.000
	12	0.0	0.000	0.000	0.0	0.000	0.000
	13	0.0	0.000	0.000	0.0	0.000	0.000
	14	0.0	0.000	0.000	0.0	0.000	0.000
	15	0.0	0.000	0.000	0.0	0.000	0.000
	16	0.0	0.000	0.000	0.0	0.000	0.000
	17	0.0	0.000	0.000	0.0	0.000	0.000
	18	0.0	0.000	0.000	0.0	0.000	0.000
	19	0.0	0.000	0.000	0.0	0.000	0.000
	20	0.0	0.000	0.000	0.0	0.000	0.000
	21	0.0	0.000	0.000	0.0	0.000	0.000
	22	0.0	0.000	0.000	0.0	0.000	0.000
	23	0.0	0.000	0.000	0.0	0.000	0.000
	24	0.0	0.000	0.000	0.0	0.000	0.000
	25	0.0	0.000	0.000	0.0	0.000	0.000
	26	0.0	0.000	0.000	0.0	0.000	0.000
	27	0.0	0.000	0.000	0.0	0.000	0.000
	28	0.0	0.000	0.000	0.0	0.000	0.000
	29	0.0	0.000	0.000	0.0	0.000	0.000
	30	0.0	0.000	0.000	0.0	0.000	0.000
	31	0.0	0.000	0.000	0.0	0.000	0.000
	32	0.0	0.000	0.000	0.0	0.000	0.000
	33	0.0	0.000	0.000	0.0	0.000	0.000

	34	0 0	0 000	0 000	0 0	0 000	0 000
	35	0 0	0 000	0 000	0 0	0 000	0 000
	36	0 0	0 000	0 000	0 0	0 000	0 000
	37	0 0	0.000	0.000	0.0	0.000	0 000
	38	0.0	0.000	0.000	0.0	0.000	0.000
	39	0.0	0.000	0.000	0.0	0.000	0.000
	40	0.0	0.000	0.000	0.0	0.000	0.000
	41	0.0	0.000	0.000	0.0	0.000	0.000
	42	0.0	0.000	0.000	0.0	0.000	0.000
	43	0.0	0.000	0.000	0.0	0.000	0.000
	44	0.0	0.000	0.000	0.0	0.000	0.000
	45	0.0	0.000	0.000	0.0	0.000	0.000
	46	0.0	0.000	0.000	0.0	0.000	0.000
	47	0.0	0.000	0.000	0.0	0.000	0.000
	48	0.0	0.000	0.000	0.0	0.000	0.000
	49	0.0	0.000	0.000	0.0	0.000	0.000
	50	0.0	0.000	0.000	0.0	0.000	0.000
	51	0.0	0.000	0.000	0.0	0.000	0.000
	52	0.0	0.000	0.000	0.0	0.000	0.000
	53	0.0	0.000	0.000	0.0	0.000	0.000
	54	0.0	0.000	0.000	0.0	0.000	0.000
	55	0.0	0.000	0.000	0.0	0.000	0.000
	56	0.0	0.000	0.000	0.0	0.000	0.000
	57	0.0	0.000	0.000	0.0	0.000	0.000
	58	0.0	0.000	0.000	0.0	0.000	0.000
	59	0.0	0.000	0.000	0.0	0.000	0.000
	60	0.0	0.000	0.000	0.0	0.000	0.000
	61	0.0	0.000	0.000	0.0	0.000	0.000
	62	0.0	0.000	0.000	0.0	0.000	0.000
	63	0.0	0.000	0.000	0.0	0.000	0.000
	64	0.0	0.000	0.000	0.0	0.000	0.000
	65	0.0	0.000	0.000	0.0	0.000	0.000
	66	0.0	0.000	0.000	0.0	0.000	0.000
	67	0.0	0.000	0.000	0.0	0.000	0.000
	68	0.0	0.000	0.000	0.0	0.000	0.000
	69	0.0	0.000	0.000	0.0	0.000	0.000
	70	0.0	0.000	0.000	0.0	0.000	0.000
4	0	8.6	0.052	0.000	8.6	0.052	0.000
	1	12.8	0.078	0.000	12.8	0.078	0.000
	2	12.8	0.078	0.000	12.8	0.078	0.000
	3	12.8	0.078	0.000	12.8	0.078	0.000
	4	12.8	0.078	0.000	12.8	0.078	0.000
	5	16.9	0.103	0.001	16.9	0.103	0.001
	6	21.1	0.128	0.001	21.1	0.128	0.001
	7	21.1	0.128	0.001	21.1	0.128	0.001
	8	21.1	0.128	0.001	21.1	0.128	0.001
	9	21.1	0.128	0.001	21.1	0.128	0.001
	10	25.3	0.154	0.001	25.3	0.154	0.001
	11	22.9	0.139	0.001	22.9	0.139	0.001
	12	22.9	0.139	0.001	22.9	0.139	0.001
	13	22.9	0.139	0.001	22.9	0.139	0.001
	14	22.9	0.139	0.001	22.9	0.139	0.001
	15	20.5	0.125	0.001	20.5	0.125	0.001
	16	18.1	0.110	0.001	18.1	0.110	0.001
	17	18.1	0.110	0.001	18.1	0.110	0.001
	18	18.1	0.110	0.001	18.1	0.110	0.001
	19	18.1	0.110	0.001	18.1	0.110	0.001
	20	15.7	0.095	0.001	15.7	0.096	0.001
	21	13.5	0.082	0.000	13.5	0.082	0.000
	22	13.5	0.082	0.000	13.5	0.082	0.000
	23	13.5	0.082	0.000	13.5	0.082	0.000

	24	13.5	0.082	0.000	13.5	0.082	0.000
	25	11.2	0.068	0.000	11.2	0.068	0.000
	26	9.0	0.055	0.000	9.0	0.055	0.000
	27	9.0	0.055	0.000	9.0	0.055	0.000
	28	9.0	0.055	0.000	9.0	0.055	0.000
	29	9.0	0.055	0.000	9.0	0.055	0.000
	30	6.8	0.041	0.000	6.8	0.041	0.000
	31	5.7	0.035	0.000	5.8	0.035	0.000
	32	5.7	0.035	0.000	5.8	0.035	0.000
	33	5.7	0.035	0.000	5.8	0.035	0.000
	34	5.7	0.035	0.000	5.8	0.035	0.000
	35	4.6	0.028	0.000	4.8	0.029	0.000
	36	3.4	0.021	0.000	3.8	0.023	0.000
	37	3.4	0.021	0.000	3.8	0.023	0.000
	38	3.4	0.021	0.000	3.8	0.023	0.000
	39	3.4	0.021	0.000	3.8	0.023	0.000
	40	2.3	0.014	0.000	2.8	0.017	0.000
	41	1.9	0.012	0.000	2.6	0.016	0.000
	42	1.9	0.012	0.000	2.6	0.016	0.000
	43	1.9	0.012	0.000	2.6	0.016	0.000
	44	1.9	0.012	0.000	2.6	0.016	0.000
	45	1.5	0.009	0.000	2.3	0.014	0.000
	46	1.1	0.007	0.000	2.0	0.012	0.000
	47	1.1	0.007	0.000	2.0	0.012	0.000
	48	1.1	0.007	0.000	2.0	0.012	0.000
	49	1.1	0.007	0.000	2.0	0.012	0.000
	50	0.7	0.004	0.000	1.7	0.010	0.000
	51	0.6	0.004	0.000	1.6	0.010	0.000
	52	0.6	0.004	0.000	1.6	0.010	0.000
	53	0.6	0.004	0.000	1.6	0.010	0.000
	54	0.6	0.004	0.000	1.6	0.010	0.000
	55	0.5	0.003	0.000	1.5	0.009	0.000
	56	0.4	0.003	0.000	1.4	0.008	0.000
	57	0.4	0.003	0.000	1.4	0.008	0.000
	58	0.4	0.003	0.000	1.4	0.008	0.000
	59	0.4	0.003	0.000	1.4	0.008	0.000
	60	0.3	0.002	0.000	1.3	0.008	0.000
	61	0.3	0.002	0.000	1.2	0.007	0.000
	62	0.3	0.002	0.000	1.2	0.007	0.000
	63	0.3	0.002	0.000	1.2	0.007	0.000
	64	0.3	0.002	0.000	1.2	0.007	0.000
	65	0.2	0.001	0.000	1.2	0.007	0.000
	66	0.2	0.001	0.000	1.1	0.007	0.000
	67	0.2	0.001	0.000	1.1	0.007	0.000
	68	0.2	0.001	0.000	1.1	0.007	0.000
	69	0.2	0.001	0.000	1.1	0.007	0.000
	70	0.1	0.001	0.000	1.1	0.007	0.000
5	0	2.4	0.015	0.000	2.4	0.015	0.000
	1	7.2	0.044	0.000	7.2	0.044	0.000
	2	7.2	0.044	0.000	7.2	0.044	0.000
	3	7.2	0.044	0.000	7.2	0.044	0.000
	4	7.2	0.044	0.000	7.2	0.044	0.000
	5	12.0	0.073	0.000	12.0	0.073	0.000
	6	16.8	0.102	0.001	16.8	0.102	0.001
	7	16.8	0.102	0.001	16.8	0.102	0.001
	8	16.8	0.102	0.001	16.8	0.102	0.001
	9	16.8	0.102	0.001	16.8	0.102	0.001
	10	21.6	0.131	0.001	21.6	0.131	0.001
	11	30.9	0.188	0.001	30.9	0.188	0.001
	12	30.9	0.188	0.001	30.9	0.188	0.001
	13	30.9	0.188	0.001	30.9	0.188	0.001

14	30 9	0.188	0 001	30 9	0.188	0 001
15	40 3	0 245	0 001	40 3	0 245	0.001
16	49 7	0 302	0 002	49 6	0.302	0.002
17	49.7	0 302	0 002	49.6	0.302	0.002
18	49.7	0.302	0.002	49.6	0.302	0.002
19	49.7	0.302	0.002	49.6	0.302	0.002
20	59.0	0.359	0.002	59.0	0.359	0.002
21	60.2	0.366	0.002	60.0	0.365	0.002
22	60.2	0.366	0.002	60.0	0.365	0.002
23	60.2	0.366	0.002	60.0	0.365	0.002
24	60.2	0.366	0.002	60.0	0.365	0.002
25	61.3	0.373	0.002	61.0	0.371	0.002
36	62.5	0.380	0.002	62.1	0.378	0.002
37	62.5	0.380	0.002	62.1	0.378	0.002
28	62.5	0.380	0.002	62.1	0.378	0.002
29	62.5	0.380	0.002	62.1	0.378	0.002
30	63.6	0.387	0.002	63.1	0.384	0.002
31	58.6	0.356	0.002	58.1	0.354	0.002
32	58.6	0.356	0.002	58.1	0.354	0.002
33	58.6	0.356	0.002	58.1	0.354	0.002
34	58.6	0.356	0.002	58.1	0.354	0.002
35	53.6	0.326	0.002	53.1	0.323	0.002
36	48.5	0.295	0.002	48.1	0.293	0.002
37	48.5	0.295	0.002	48.1	0.293	0.002
38	48.5	0.295	0.002	48.1	0.293	0.002
39	48.5	0.295	0.002	48.1	0.293	0.002
40	43.5	0.265	0.002	43.2	0.263	0.002
41	39.1	0.238	0.001	39.7	0.242	0.001
42	39.1	0.238	0.001	39.7	0.242	0.001
43	39.1	0.238	0.001	39.7	0.242	0.001
44	39.1	0.238	0.001	39.7	0.242	0.001
45	34.7	0.211	0.001	36.3	0.221	0.001
46	30.2	0.184	0.001	32.9	0.200	0.001
47	30.2	0.184	0.001	32.9	0.200	0.001
48	30.2	0.184	0.001	32.9	0.200	0.001
49	30.2	0.184	0.001	32.9	0.200	0.001
50	25.8	0.157	0.001	29.5	0.179	0.001
51	24.0	0.146	0.001	27.1	0.165	0.001
52	24.0	0.146	0.001	27.1	0.165	0.001
53	24.0	0.146	0.001	27.1	0.165	0.001
54	24.0	0.146	0.001	27.1	0.165	0.001
55	22.1	0.134	0.001	24.8	0.151	0.001
56	20.2	0.123	0.001	22.5	0.137	0.001
57	20.2	0.123	0.001	22.5	0.137	0.001
58	20.2	0.123	0.001	22.5	0.137	0.001
59	20.2	0.123	0.001	22.5	0.137	0.001
60	18.4	0.112	0.001	20.1	0.123	0.001
61	16.9	0.103	0.001	19.0	0.116	0.001
62	16.9	0.103	0.001	19.0	0.116	0.001
63	16.9	0.103	0.001	19.0	0.116	0.001
64	16.9	0.103	0.001	19.0	0.116	0.001
65	15.5	0.094	0.001	17.9	0.109	0.001
66	14.0	0.085	0.001	16.8	0.102	0.001
67	14.0	0.085	0.001	16.8	0.102	0.001
68	14.0	0.085	0.001	16.8	0.102	0.001
69	14.0	0.085	0.001	16.8	0.102	0.001
70	12.5	0.076	0.000	15.7	0.095	0.001

Table B-5 - Estimated concentrations in water, food and milk for four centerline wells -- data for toxicity assessment.

		STOP SOURCE			CONTINUE SOURCE		
well	year	water (ug/l)	food (ug/kg)	milk (ug/l)	water (ug/l)	food (ug/kg)	milk (ug/l)
c1	0	9.2	56.13836	3	9.2	56.13836	3
	10	31.8	193.6548	3.735736	31.8	193.6548	3.735736
	20	46.3	281.8844	4.257736	46.3	281.8844	4.257736
	30	45.6	277.6251	4.232536	46.8	284.9268	4.275736
	40	35.5	216.1686	3.868936	43.4	264.2385	4.153336
	50	22.8	138.8916	3.411736	40.4	245.9841	4.045336
	60	14.5	88.3878	3.112936	37.3	227.1212	3.933736
	70	8.2	50.05356	2.886136	35.4	215.5601	3.865336
c2	0	2.8	17.19564	2.691736	2.8	17.19564	2.691736
	10	12.2	74.39276	3.030136	12.2	74.39276	3.030136
	20	25.9	157.7545	3.523336	25.9	157.7545	3.523336
	30	29.9	182.0937	3.667336	29.9	182.0937	3.667336
	40	26.6	162.0139	3.548536	26.7	162.6224	3.552136
	50	22.0	134.0238	3.382936	23.9	145.5849	3.451336
	60	18.6	113.3355	3.260536	21.9	133.4153	3.379336
	70	14.4	87.77932	3.109336	20.8	126.722	3.339736
c3	0	5.1	31.19068	2.774536	5.0	30.5822	2.770936
	10	27.2	165.6648	3.570136	27.2	165.6648	3.570136
	20	57.4	349.4257	4.657336	57.4	349.4257	4.657336
	30	55.7	339.0816	4.596136	55.2	336.0392	4.578136
	40	38.0	231.3806	3.958936	37.8	230.1636	3.951736
	50	24.1	146.8019	3.458536	27.2	165.6648	3.570136
	60	18.2	110.9016	3.246136	20.2	123.0712	3.318136
	70	13.2	80.47756	3.066136	16.9	102.9913	3.199336
c4	0	1.5	9.285405	2.644936	1.5	9.285405	2.644936
	10	13.8	84.12844	3.087736	13.8	84.12844	3.087736
	20	58.0	353.0766	4.678936	57.8	351.8596	4.671736
	30	79.0	480.8574	5.424936	78.7	479.032	5.424136
	40	58.5	356.119	4.696936	57.7	351.2512	4.668136
	50	32.4	197.3057	3.757336	37.6	228.9467	3.944536
	60	21.0	127.939	3.346936	22.7	138.2832	3.408136
	70	12.6	76.82668	3.044536	15.3	93.25564	3.141736

Table B-6 - Estimated concentrations in water, food and milk for four centerline wells -- data for cancer risk Assessment.

well	year	STOP SOURCE water (ug/l)	STOP SOURCE food (mg/kg)	STOP SOURCE milk (mg/l)	CONTINUE SOURCE water (ug/l)	CONTINUE SOURCE food (mg/kg)	CONTINUE SOURCE milk (mg/l)
1	0	8.3	0.051	0.000	8.3	0.051	0.000
	1	14.0	0.085	0.001	14.0	0.085	0.001
	2	14.0	0.085	0.001	14.0	0.085	0.001
	3	14.0	0.085	0.001	14.0	0.085	0.001
	4	14.0	0.085	0.001	14.0	0.085	0.001
	5	19.6	0.119	0.001	19.6	0.119	0.001
	6	25.3	0.154	0.001	25.3	0.154	0.001
	7	25.3	0.154	0.001	25.3	0.154	0.001
	8	25.3	0.154	0.001	25.3	0.154	0.001
	9	25.3	0.154	0.001	25.3	0.154	0.001
	10	30.9	0.188	0.001	30.9	0.188	0.001
	11	34.5	0.210	0.001	34.5	0.210	0.001
	12	34.5	0.210	0.001	34.5	0.210	0.001
	13	34.5	0.210	0.001	34.5	0.210	0.001
	14	34.5	0.210	0.001	34.5	0.210	0.001
	15	38.2	0.232	0.001	38.2	0.232	0.001
	16	41.8	0.254	0.002	41.8	0.254	0.002
	17	41.8	0.254	0.002	41.8	0.254	0.002
	18	41.8	0.254	0.002	41.8	0.254	0.002
	19	41.8	0.254	0.002	41.8	0.254	0.002
	20	45.4	0.276	0.002	45.4	0.276	0.002
	21	45.2	0.275	0.002	45.5	0.277	0.002
	22	45.2	0.275	0.002	45.5	0.277	0.002
	23	45.2	0.275	0.002	45.5	0.277	0.002
	24	45.2	0.275	0.002	45.5	0.277	0.002
	25	45.1	0.274	0.002	45.7	0.278	0.002
	26	44.9	0.273	0.002	45.8	0.279	0.002
	27	44.9	0.273	0.002	45.8	0.279	0.002
	28	44.9	0.273	0.002	45.8	0.279	0.002
	29	44.9	0.273	0.002	45.8	0.279	0.002
	30	44.7	0.272	0.002	45.9	0.279	0.002
	31	42.2	0.257	0.002	45.1	0.274	0.002
	32	42.2	0.257	0.002	45.1	0.274	0.002
	33	42.2	0.257	0.002	45.1	0.274	0.002
	34	42.2	0.257	0.002	45.1	0.274	0.002
	35	39.7	0.241	0.001	44.2	0.269	0.002
	36	37.1	0.226	0.001	43.4	0.264	0.002
	37	37.1	0.226	0.001	43.4	0.264	0.002
	38	37.1	0.226	0.001	43.4	0.264	0.002
	39	37.1	0.226	0.001	43.4	0.264	0.002
	40	34.6	0.211	0.001	42.5	0.259	0.002
	41	31.4	0.191	0.001	41.8	0.254	0.002
	42	31.4	0.191	0.001	41.8	0.254	0.002
	43	31.4	0.191	0.001	41.8	0.254	0.002
	44	31.4	0.191	0.001	41.8	0.254	0.002
	45	28.3	0.172	0.001	41.0	0.249	0.001
	46	25.1	0.153	0.001	40.3	0.245	0.001
	47	25.1	0.153	0.001	40.3	0.245	0.001
	48	25.1	0.153	0.001	40.3	0.245	0.001
	49	25.1	0.153	0.001	40.3	0.245	0.001
	50	21.9	0.133	0.001	39.5	0.240	0.001
	51	19.8	0.121	0.001	38.7	0.236	0.001

	52	19 8	0.121	0 001	38 7	0 236	0 001
	53	19.8	0.121	0.001	38 7	0.236	0.001
	54	19.8	0.121	0 001	38 7	0.236	0.001
	55	17.8	0.108	0 001	38 0	0.231	0.001
	56	15.7	0.095	0.001	37.2	0.226	0.001
	57	15.7	0.095	0.001	37.2	0.226	0.001
	58	15.7	0.095	0.001	37.2	0.226	0.001
	59	15.7	0.095	0.001	37.2	0.226	0.001
	60	13.6	0.083	0.000	36.4	0.221	0.001
	61	12.0	0.073	0.000	35.9	0.219	0.001
	62	12.0	0.073	0.000	35.9	0.219	0.001
	63	12.0	0.073	0.000	35.9	0.219	0.001
	64	12.0	0.073	0.000	35.9	0.219	0.001
	65	10.5	0.064	0.000	35.5	0.216	0.001
	66	8.9	0.054	0.000	35.0	0.213	0.001
	67	8.9	0.054	0.000	35.0	0.213	0.001
	68	8.9	0.054	0.000	35.0	0.213	0.001
	69	8.9	0.054	0.000	35.0	0.213	0.001
	70	7.3	0.044	0.000	34.5	0.210	0.001
2	0	1.9	0.012	0.000	1.9	0.012	0.000
	1	4.3	0.026	0.000	4.3	0.026	0.000
	2	4.3	0.026	0.000	4.3	0.026	0.000
	3	4.3	0.026	0.000	4.3	0.026	0.000
	4	4.3	0.026	0.000	4.3	0.026	0.000
	5	6.6	0.040	0.000	6.6	0.040	0.000
	6	9.0	0.054	0.000	9.0	0.054	0.000
	7	9.0	0.054	0.000	9.0	0.054	0.000
	8	9.0	0.054	0.000	9.0	0.054	0.000
	9	9.0	0.054	0.000	9.0	0.054	0.000
	10	11.3	0.069	0.000	11.3	0.069	0.000
	11	14.7	0.090	0.001	14.7	0.090	0.001
	12	14.7	0.090	0.001	14.7	0.090	0.001
	13	14.7	0.090	0.001	14.7	0.090	0.001
	14	14.7	0.090	0.001	14.7	0.090	0.001
	15	18.2	0.110	0.001	18.2	0.110	0.001
	16	21.6	0.131	0.001	21.6	0.131	0.001
	17	21.6	0.131	0.001	21.6	0.131	0.001
	18	21.6	0.131	0.001	21.6	0.131	0.001
	19	21.6	0.131	0.001	21.6	0.131	0.001
	20	25.0	0.152	0.001	25.0	0.152	0.001
	21	26.0	0.158	0.001	26.0	0.158	0.001
	22	26.0	0.158	0.001	26.0	0.158	0.001
	23	26.0	0.158	0.001	26.0	0.158	0.001
	24	26.0	0.158	0.001	26.0	0.158	0.001
	25	27.0	0.164	0.001	27.0	0.164	0.001
	26	28.0	0.170	0.001	28.0	0.170	0.001
	27	28.0	0.170	0.001	28.0	0.170	0.001
	28	28.0	0.170	0.001	28.0	0.170	0.001
	29	28.0	0.170	0.001	28.0	0.170	0.001
	30	29.0	0.176	0.001	29.0	0.176	0.001
	31	28.2	0.171	0.001	28.2	0.172	0.001
	32	28.2	0.171	0.001	28.2	0.172	0.001
	33	28.2	0.171	0.001	28.2	0.172	0.001
	34	28.2	0.171	0.001	28.2	0.172	0.001
	35	27.4	0.166	0.001	27.4	0.167	0.001
	36	26.5	0.161	0.001	26.6	0.162	0.001
	37	26.5	0.161	0.001	26.6	0.162	0.001
	38	26.5	0.161	0.001	26.6	0.162	0.001
	39	26.5	0.161	0.001	26.6	0.162	0.001
	40	25.7	0.156	0.001	25.8	0.157	0.001
	41	24.6	0.149	0.001	25.1	0.153	0.001

	42	24 6	0 149	0 001	25 1	0.153	0 001
	43	24.6	0 149	0 001	25 1	0.153	0 001
	44	24 6	0 149	0 001	25.1	0.153	0.001
	45	23.4	0.142	0 001	24.4	0.148	0 001
	46	22.3	0.135	0.001	23.7	0.144	0.001
	47	22.3	0.135	0.001	23.7	0.144	0.001
	48	22.3	0.135	0.001	23.7	0.144	0.001
	49	22.3	0.135	0 001	23.7	0.144	0.001
	50	21.1	0.128	0.001	23.0	0.140	0 001
	51	20.3	0.123	0.001	22.5	0.137	0.001
	52	20.3	0.123	0.001	22.5	0.137	0.001
	53	20.3	0.123	0.001	22.5	0.137	0.001
	54	20.3	0.123	0.001	22.5	0.137	0.001
	55	19.4	0.118	0.001	22.0	0.134	0.001
	56	18.6	0.113	0.001	21.5	0.131	0.001
	57	18.6	0.113	0.001	21.5	0.131	0.001
	58	18.6	0.113	0.001	21.5	0.131	0.001
	59	18.6	0.113	0.001	21.5	0.131	0.001
	60	17.7	0.108	0.001	21.0	0.128	0.001
	61	16.7	0.101	0.001	20.7	0.126	0.001
	62	16.7	0.101	0.001	20.7	0.126	0.001
	63	16.7	0.101	0.001	20.7	0.126	0.001
	64	16.7	0.101	0.001	20.7	0.126	0.001
	65	15.6	0.095	0.001	20.5	0.124	0.001
	66	14.6	0.089	0.001	20.2	0.123	0.001
	67	14.6	0.089	0.001	20.2	0.123	0.001
	68	14.6	0.089	0.001	20.2	0.123	0.001
	69	14.6	0.089	0.001	20.2	0.123	0.001
	70	13.5	0.082	0.000	19.9	0.121	0.001
3	0	4.2	0.026	0.000	4.1	0.025	0.000
	1	9.7	0.059	0.000	9.7	0.059	0.000
	2	9.7	0.059	0.000	9.7	0.059	0.000
	3	9.7	0.059	0.000	9.7	0.059	0.000
	4	9.7	0.059	0.000	9.7	0.059	0.000
	5	15.3	0.093	0.001	15.2	0.092	0.001
	6	20.8	0.126	0.001	20.8	0.126	0.001
	7	20.8	0.126	0.001	20.8	0.126	0.001
	8	20.8	0.126	0.001	20.8	0.126	0.001
	9	20.8	0.126	0.001	20.8	0.126	0.001
	10	26.3	0.160	0.001	26.3	0.160	0.001
	11	33.9	0.206	0.001	33.9	0.206	0.001
	12	33.9	0.206	0.001	33.9	0.206	0.001
	13	33.9	0.206	0.001	33.9	0.206	0.001
	14	33.9	0.206	0.001	33.9	0.206	0.001
	15	41.4	0.252	0.001	41.4	0.252	0.001
	16	49.0	0.296	0.002	49.0	0.298	0.002
	17	49.0	0.298	0.002	49.0	0.298	0.002
	18	49.0	0.298	0.002	49.0	0.298	0.002
	19	49.0	0.298	0.002	49.0	0.298	0.002
	20	56.5	0.344	0.002	56.5	0.344	0.002
	21	56.1	0.341	0.002	56.0	0.340	0.002
	22	56.1	0.341	0.002	56.0	0.34J	0.002
	23	56.1	0.341	0.002	56.0	0.340	0.002
	24	56.1	0.341	0.002	56.0	0.340	0.002
	25	55.7	0.339	0.002	55.4	0.337	0.002
	26	55.2	0.336	0.002	54.9	0.334	0.002
	27	55.2	0.336	0.002	54.9	0.334	0.002
	28	55.2	0.336	0.002	54.9	0.334	0.002
	29	55.2	0.336	0.002	54.9	0.334	0 002
	30	54.8	0.333	0.002	54.3	0.330	0.002
	31	50.4	0.307	0.002	50.0	0.304	0.002

	32	50.4	0 307	0 002	50 0	0.304	0 002
	33	50.4	0 307	0 002	50.0	0.304	0 002
	34	50.4	0 307	0 002	50.0	0.304	0 002
	35	46.0	0.280	0 002	45.6	0.277	0 002
	36	41.5	0.253	0.001	41.3	0.251	0.001
	37	41.5	0.253	0.001	41.3	0.251	0.001
	38	41.5	0.253	0.001	41.3	0.251	0.001
	39	41.5	0.253	0 001	41.3	0.251	0.001
	40	37.1	0.226	0.001	36.9	0.225	0.001
	41	33.6	0.205	0.001	34.3	0.208	0.001
	42	33.6	0.205	0 001	34.3	0.208	0.001
	43	33.6	0.205	0.001	34.3	0.208	0.001
	44	33.6	0.205	0.001	34.3	0.208	0.001
	45	30.2	0.183	0.001	31.6	0.192	0.001
	46	26.7	0.162	0.001	29.0	0.176	0.001
	47	26.7	0.162	0.001	29.0	0.176	0.001
	48	26.7	0.162	0.001	29.0	0.176	0.001
	49	26.7	0.162	0.001	29.0	0.176	0.001
	50	23.2	0.141	0.001	26.3	0.160	0.001
	51	21.7	0.132	0.001	24.6	0.149	0.001
	52	21.7	0.132	0.001	24.6	0.149	0.001
	53	21.7	0.132	0.001	24.6	0.149	0.001
	54	21.7	0.132	0.001	24.6	0.149	0.001
	55	20.3	0.123	0.001	22.8	0.139	0.001
	56	18.8	0.114	0.001	21.1	0.128	0.001
	57	18.8	0.114	0.001	21.1	0.128	0.001
	58	18.8	0.114	0.001	21.1	0.128	0.001
	59	18.8	0.114	0.001	21.1	0.128	0.001
	60	17.3	0.105	0.001	19.3	0.117	0.001
	61	16.1	0.098	0.001	18.5	0.112	0.001
	62	16.1	0.098	0.001	18.5	0.112	0.001
	63	16.1	0.098	0.001	18.5	0.112	0.001
	64	16.1	0.098	0.001	18.5	0.112	0.001
	65	14.8	0.090	0.001	17.7	0.107	0.001
	66	13.6	0.082	0.000	16.8	0.102	0.001
	67	13.6	0.082	0.000	16.8	0.102	0.001
	68	13.6	0.082	0.000	16.8	0.102	0.001
	69	13.6	0.082	0.000	16.8	0.102	0.001
	70	12.3	0.075	0.000	16.0	0.097	0.001
4	0	0.6	0.004	0.000	0.6	0.004	0.000
	1	3.7	0.022	0.000	3.7	0.022	0.000
	2	3.7	0.022	0.000	3.7	0.022	0.000
	3	3.7	0.022	0.000	3.7	0.022	0.000
	4	3.7	0.022	0.000	3.7	0.022	0.000
	5	6.8	0.041	0.000	6.8	0.041	0.000
	6	9.8	0.060	0.000	9.8	0.060	0.000
	7	9.8	0.060	0.000	9.8	0.060	0.000
	8	9.8	0.060	0.000	9.8	0.060	0.000
	9	9.8	0.060	0.000	9.8	0.060	0.000
	10	12.9	0.078	0.000	12.9	0.078	0.000
	11	24.0	0.146	0.001	23.9	0.145	0.001
	12	24.0	0.146	0.001	23.9	0.145	0.001
	13	24.0	0.146	0.001	23.9	0.145	0.001
	14	24.0	0.146	0.001	23.9	0.145	0.001
	15	35.0	0.213	0.001	34.9	0.212	0.001
	16	46.1	0.280	0.002	45.9	0.279	0.002
	17	46.1	0.280	0.002	45.9	0.279	0.002
	18	46.1	0.280	0.002	45.9	0.279	0.002
	19	46.1	0.280	0.002	45.9	0.279	0.002
	20	57.1	0.347	0.002	56.9	0.346	0.002
	21	62.4	0.379	0.002	62.1	0.378	0.002

22	62.4	0.379	0.002
23	62.4	0.379	0.002
24	62.4	0.379	0.002
25	67.6	0.411	0.002
26	72.9	0.443	0.003
27	72.9	0.443	0.003
28	72.9	0.443	0.003
29	72.9	0.443	0.003
30	78.1	0.475	0.003
31	73.0	0.444	0.003
32	73.0	0.444	0.003
33	73.0	0.444	0.003
34	73.0	0.444	0.003
35	67.9	0.413	0.002
36	62.7	0.382	0.002
37	62.7	0.382	0.002
38	62.7	0.382	0.002
39	62.7	0.382	0.002
40	57.6	0.350	0.002
41	51.1	0.311	0.002
42	51.1	0.311	0.002
43	51.1	0.311	0.002
44	51.1	0.311	0.002
45	44.6	0.271	0.002
46	38.0	0.231	0.001
47	38.0	0.231	0.001
48	38.0	0.231	0.001
49	38.0	0.231	0.001
50	31.5	0.192	0.001
51	28.7	0.174	0.001
52	28.7	0.174	0.001
53	28.7	0.174	0.001
54	28.7	0.174	0.001
55	25.8	0.157	0.001
56	23.0	0.140	0.001
57	23.0	0.140	0.001
58	23.0	0.140	0.001
59	23.0	0.140	0.001
60	20.1	0.122	0.001
61	18.0	0.110	0.001
62	18.0	0.110	0.001
63	18.0	0.110	0.001
64	18.0	0.110	0.001
65	15.9	0.097	0.001
66	13.8	0.084	0.000
67	13.8	0.084	0.000
68	13.8	0.084	0.000
69	13.8	0.084	0.000
70	11.7	0.071	0.000

62.1	0.378	0 002
62.1	0.378	0 002
62.1	0.378	0 002
67.4	0.410	0 002
72.6	0.442	0 003
72.6	0.442	0.003
72.6	0.442	0 003
72.6	0.442	0 003
77.8	0.473	0 003
72.6	0.441	0.003
72.6	0.441	0.003
72.6	0.441	0.003
72.6	0.441	0.003
67.3	0.410	0.002
62.1	0.378	0.002
62.1	0.378	0.002
62.1	0.378	0.002
62.1	0.378	0.002
56.8	0.346	0.002
51.8	0.315	0.002
51.8	0.315	0.002
51.8	0.315	0.002
51.8	0.315	0.002
46.8	0.284	0.002
41.7	0.254	0.002
41.7	0.254	0.002
41.7	0.254	0.002
41.7	0.254	0.002
36.7	0.223	0.001
33.0	0.201	0.001
33.0	0.201	0.001
33.0	0.201	0.001
33.0	0.201	0.001
29.3	0.178	0.001
25.5	0.155	0.001
25.5	0.155	0.001
25.5	0.155	0.001
25.5	0.155	0.001
21.8	0.133	0.001
20.0	0.121	0.001
20.0	0.121	0.001
20.0	0.121	0.001
20.0	0.121	0.001
18.1	0.110	0.001
16.3	0.099	0.001
16.3	0.099	0.001
16.3	0.099	0.001
16.3	0.099	0.001
14.4	0.088	0.001

END

DATE
FILMED
7/9/93