ANL/EVS/TM-14/4

Data Collection Handbook to Support Modeling Impacts of Radioactive Material in Soil and Building Structures

Environmental Science Division

About Argonne National Laboratory
Argonne is a U.S. Department of Energy laboratory managed by UChicago Argonne, LLC under contract DE-AC02-06CH11357. The Laboratory's main facility is outside Chicago, at 9700 South Cass Avenue, Argonne, Illinois 60439. For information about Argonne and its pioneering science and technology programs, see www.anl.gov.

DOCUMENT AVAILABILITY

Online Access: U.S. Department of Energy (DOE) reports produced after 1991 and a growing number of pre-1991 documents are available free via DOE's SciTech Connect (http://www.osti.gov/scitech/).

Reports not in digital format may be purchased by the public from the National Technical Information Service (NTIS):
U.S. Department of Commerce
National Technical Information Service
5301 Shawnee Rd
Alexandria, VA 22312
www.ntis.gov
Phone: (800) 553-NTIS (6847) or (703) 605-6000
Fax: (703) 605-6900
Email: **orders@ntis.gov**

Reports not in digital format are available to DOE and DOE contractors from the Office of Scientific and Technical Information (OSTI):
U.S. Department of Energy
Office of Scientific and Technical Information
P.O. Box 62
Oak Ridge, TN 37831-0062
www.osti.gov
Phone: (865) 576-8401
Fax: (865) 576-5728
Email: **reports@osti.gov**

Disclaimer
This report was prepared as an account of work sponsored by an agency of the United States Government. Neither the United States Government nor any agency thereof, nor UChicago Argonne, LLC, nor any of their employees or officers, makes any warranty, express or implied, or assumes any legal liability or responsibility for the accuracy, completeness, or usefulness of any information, apparatus, product, or process disclosed, or represents that its use would not infringe privately owned rights. Reference herein to any specific commercial product, process, or service by trade name, trademark, manufacturer, or otherwise, does not necessarily constitute or imply its endorsement, recommendation, or favoring by the United States Government or any agency thereof. The views and opinions of document authors expressed herein do not necessarily state or reflect those of the United States Government or any agency thereof, Argonne National Laboratory, or UChicago Argonne, LLC.

ANL/EVS/TM-14/4

Data Collection Handbook to Support Modeling Impacts of Radioactive Material in Soil and Building Structures

---

prepared by
Charley Yu, Sunita Kamboj, Cheng Wang, and Jing-Jy Cheng
Environmental Science Division, Argonne National Laboratory

September 2015

CONTENTS

CONTENTS (CONT.)

CONTENTS (CONT.)

CONTENTS (CONT.)

CONTENTS (CONT.)

CONTENTS (CONT.)

FIGURES

FIGURES (CONT.)

TABLES

TABLES (CONT.)

TABLES (CONT.)

TABLES (CONT.)

TABLES (CONT.)

TABLES (CONT.)

This page intentionally left blank.

NOTATION

The following is a list of acronyms, initialisms, and abbreviations (including units of measure) used in this document.

ACRONYMS, INITIALISMS, AND ABBREVIATIONS

AHS	American Housing Survey
ANL	Argonne National Laboratory
ASTM	American Society for Testing and Materials
CSFII	Continuing Survey of Food Intake by Individuals
CSSM	Center for Survey Statistics and Methodology (Iowa State University)
DCF	dose conversion factor
DOA	U.S. Department of the Army
DOE	U.S. Department of Energy
DOI	U.S. Department of the Interior
DRF	dose reduction factor
EPA	U.S. Environmental Protection Agency
FDR	frequency domain reflectometer
FUSRAP	Formerly Utilized Sites Remedial Action Program
FW	freshwater
IAEA	International Atomic Energy Agency
ICRP	International Commission on Radiological Protection
LLD	lower limit of detection
L/S	liquid-to-solid ratio
NAS	National Academy of Sciences
NCDC	National Climate Data Center
NCI	National Cancer Institute
NCRP	National Council on Radiation Protection and Measurements
NHANES	National Health and Nutrition Examination Survey
NMFS	National Marine Fisheries Service
NMM	neutron moisture meter
NOAA	National Oceanic and Atmospheric Administration
NRC	U.S. Nuclear Regulatory Commission
NRCS	National Resources Conservation Service
PNNL	Pacific Northwest National Laboratory

RECS	Residential Energy Consumption Survey
REV	representative elementary volume
SCS	U.S. Soil Conservation Service
SW	saltwater
TCDD	tetrachlorodibenzo-p-dioxin
TDR	time domain reflectometer
USDA	U.S. Department of Agriculture
USLE	Universal Soil Loss Equation

UNITS OF MEASURE

Bq	becquerel(s)
°C	degree(s) Celsius
cm	centimeter(s)
cm^3	cubic centimeter(s)
d	day(s)
ft^2	square foot (feet)
g	gram(s)
gal	gallon(s)
h	hour(s)
in.	inch(es)
keV	kiloelectron volt(s)
kg	kilogram(s)
km^2	square kilometer(s)
L	liter(s)
l	length
l^2	length squared
l^3	length cubed
lb	pound(s)
M	mass
m	meter(s)
m^2	square meter(s)
m^3	cubic meter(s)
mi	mile(s)
mi^2	square mile(s)
mm	millimeter(s)
mol	mole(s)
mrem	millirem(s)
mSv	millisievert(s)
pCi	picocurie(s)
s	second(s)
T	time
V	volume
yr	year(s)

DATA COLLECTION HANDBOOK TO SUPPORT MODELING THE IMPACTS OF RADIOACTIVE MATERIAL IN SOIL AND BUILDING STRUCTURES

by

Charley Yu, Sunita Kamboj, Cheng Wang, and Jing-Jy Cheng

ABSTRACT

This handbook is an update of the 1993 version of the Data Collection Handbook and the Radionuclide Transfer Factors Report to support modeling the impact of radioactive material in soil. Many new parameters have been added to the RESRAD Family of Codes, and new measurement methodologies are available. A detailed review of available parameter databases was conducted in preparation of this new handbook. This handbook is a companion document to the user manuals when using the RESRAD (onsite) and RESRAD-OFFSITE code. It can also be used for RESRAD-BUILD code because some of the building-related parameters are included in this handbook. The RESRAD (onsite) has been developed for implementing U.S. Department of Energy Residual Radioactive Material Guidelines. Hydrogeological, meteorological, geochemical, geometrical (size, area, depth), crops and livestock, human intake, source characteristic, and building characteristic parameters are used in the RESRAD (onsite) code. The RESRAD-OFFSITE code is an extension of the RESRAD (onsite) code and can also model the transport of radionuclides to locations outside the footprint of the primary contamination. This handbook discusses parameter definitions, typical ranges, variations, and measurement methodologies. It also provides references for sources of additional information. Although this handbook was developed primarily to support the application of RESRAD Family of Codes, the discussions and values are valid for use of other pathway analysis models and codes.

1 INTRODUCTION

The RESRAD Family of Codes is a suite of software tools developed by Argonne National Laboratory (ANL) for the evaluation of radiologically contaminated sites (Yu et al. 2013a; Yu 2007, 2006, Yu 1999). In 1993, a Data Collection Handbook (Yu et al. 1993) and a Radionuclide Transfer Factors Report (Wang et al. 1993) were published to support the use of RESRAD (onsite) code for modeling the impacts of radioactive materials in soil. The RESRAD (onsite) computer code evaluates the radiological dose and excess cancer risk to an individual who is exposed while residing and/or working in an area where soil is contaminated with radionuclides (Yu et al. 2001). RESRAD (onsite) code was developed by ANL in the 1980s (Gilbert et al. 1989) in support of the U.S. Department of Energy (DOE) Order establishing residual radioactive material guidelines (DOE Order 5400.5, now superseded by Order 458.1). The DOE and other agencies and their contractors have used the RESRAD (onsite) code and its manual to derive cleanup criteria and dose calculations. The DOE Office of Environment, Health, Safety and Security and the Office of Environmental Management provide periodic guidance regarding any significant changes to the code and manual. Since its first release in June 1989, many new features and pathways have been added to the RESRAD (onsite) code in response to feedback from users and sponsors. The RESRAD team has participated in many national and international model intercomparison studies in which both hypothetical and actual contaminated site-based scenarios were analyzed using the RESRAD (onsite) code. The development of the RESRAD-OFFSITE code started in the 1990s, and it was recently improved with a new source term model on the request of U.S. Nuclear Regulatory Commission (Yu et al. 2013b). Because many new parameters have been added to the RESRAD codes and new measurement methodologies are available as well as parameter database are available now, it is time to update the 1993 Data Collection Handbook and Radionuclide Transfer Factors Report. Both RESRAD (onsite) and RESRAD-OFFSITE codes consider a building or house located on the contaminated soil. Therefore, many building-related parameters are discussed in this handbook. These building-related parameters are also used in the RESRAD-BUILD computer code, which is designed for the evaluation of radiologically contaminated buildings (Yu et al. 2003). Hence this handbook is useful when RESRAD-BUILD code is being applied to evaluate radiologically contaminated buildings and structures. Additional information on the parameters used in RESRAD-BUILD is documented in previous work (Kamboj et al. 2000, Yu et al. 2000, and Biwer et al. 2002).

Fifty-six parameters are discussed in this handbook. The definition, typical range, default value used in RESRAD (onsite) and RESRAD-OFFSITE, relation to other parameters, and measurement methodology are given for most of the measurable parameters. Table 1.1 lists the default values and the lower and upper bounds set in the RESRAD (onsite) code for each parameter used in the code. Table 1.2 lists the default values and the lower and upper bounds set in the RESRAD-OFFSITE code for each parameter used in the code. The intent of this handbook is to provide users with a better understanding of each input parameter in terms of its typical range, variation, and use in the RESRAD (onsite) and RESRAD-OFFSITE codes.

The default parameter values listed in Tables 1.1 and 1.2 have been carefully selected, and use of these values, in most cases, will not result in significant underestimation of the dose or risk. Site-specific parameters should be used whenever possible, especially for sensitive parameters.

The topics discussed in each section of this handbook are as follows: Section 2, hydrogeological parameters; Section 3, meteorological parameters; Section 4, radon emanation parameters; Section 5, building characteristic parameters; Section 6, crop and livestock parameters; Section 7, human intake parameters; Section 8, source characteristic parameters; and Section 9, miscellaneous parameters to estimate derived concentration guideline levels. References are given in Section 10.

TABLE 1.1 Default Values, Lower Bounds, and Upper Bounds for RESRAD (onsite) Input Parameters

Parameter	Unit	Default Value	Lower[a] Bound	Upper[a] Bound
Soil bulk density				
Cover material	g/cm^3	1.5	0.001	22.5
Contaminated zone	g/cm^3	1.5	0.001	22.5
Unsaturated zone	g/cm^3	1.5	0.001	22.5
Saturated zone	g/cm^3	1.5	0.001	22.5
Building foundation material	g/cm^3	2.4	0.001	22.5
Total porosity				
Cover material	-[b]	0.4	0.00001	1
Contaminated zone	-	0.4	0.00001	1
Unsaturated zone	-	0.4	0.00001	1
Saturated zone	-	0.4	0.00001	1
Building foundation material	-	0.1	0.00001	1
Effective porosity				
Saturated zone	-	0.2	1×10^{-34}	1
Unsaturated zone	-	0.2	1×10^{-34}	1
Hydraulic conductivity				
Contaminated zone	m/yr	10	0.001	1×10^{10}
Unsaturated zone	m/yr	10	0.001	1×10^{10}
Saturated zone	m/yr	100	0.001	1×10^{10}
Volumetric water content				
Cover material	-	0.05	0	1
Building foundation material	-	0.03	0	1
Effective radon diffusion coefficient				
Cover material	m^2/s	2×10^{-6}	-1[c]	1
Contaminated zone	m^2/s	2×10^{-6}	-1[c]	1
Building foundation material	m^2/s	3×10^{-7}	-1[c]	1
Radon emanation coefficient (Radon-222/Radon-220)	-	0.25/0.15	0.01	1
Precipitation rate	m/yr	1	0	10
Runoff coefficient	-	0.2	0	1
Irrigation rate	m/yr	0.2	0	10
Evapotranspiration coefficient	-	0.5	0	0.999

TABLE 1.1 (Cont.)

Parameter	Unit	Default Value	Lower[a] Bound	Upper[a] Bound
Soil-specific *b* parameter				
Contaminated zone	-	5.3	0	15
Unsaturated zone	-	5.3	0	15
Saturated zone	-	5.3	0	15
Erosion rate				
Cover material	m/yr	0.001	0	5
Contaminated zone	m/yr	0.001	0	5
Hydraulic gradient	-	0.02	1×10^{-10}	10
Length of contaminated zone parallel to the aquifer flow	m	100	0.0001	1×10^{6}
Watershed area for nearby stream or pond	m^2	1×10^{6}	0.0001	1×10^{34}
Water table drop rate	m/yr	0.001	0	5
Well-pump intake depth	m	10	0.00001	1,000
Radon vertical dimension of mixing	m	2	0.001	1,000
Average annual wind speed	m/s	2	0.0001	20
Average building air exchange rate	1/h	0.5	0	1,000
Building room height	m	2.5	0.0001	100
Building indoor area factor	-	0	0	100
Thickness of uncontaminated unsaturated zone	m	4	0	10,000
Building foundation thickness	m	0.15	0	10
Foundation depth below ground surface	m	-1[c]	-100	100
Fraction of time spent indoors on-site	-	0.5	0	1
Fraction of time spent outdoors on-site	-	0.25	0	1
Area of contaminated zone	m^2	10,000	0.0001	1×10^{15}
Cover depth	m	0	0	100

TABLE 1.1 (Cont.)

Parameter	Unit	Default Value	Lower[a] Bound	Upper[a] Bound
Distribution coefficient	cm^3/g	d	0	∝
Fraction of annular areas within contaminated area	-	0	0	1
Radionuclide concentration in groundwater	pCi/L	0	0	∝
Leach rate	1/yr	0	0	∝
Livestock fodder intake				
Meat	kg/d	68	0	300
Milk	kg/d	55	0	300
Mass loading for inhalation	g/m^3	1×10^{-4}	0	1
Milk consumption rate	L/yr	92	0	1,000
Filtration factor for inhalation	-	0.4	0	1
Depth of roots	m	0.9	0	100
Soil ingestion rate	g/yr	36.5	0	10,000
Thickness of contaminated zone	m	2	1×10^{-5}	1,000
Radiation dose limit	mrem/yr	25	0	∝
Seafood consumption rate				
Fish	kg/yr	5.4	0	1,000
Other seafood	kg/yr	0.9	0	100
Fruit, vegetable, and grain consumption rate	kg/yr	160	0	1,000
Inhalation rate	m^3/yr	8,400	0	20,000
Leafy vegetable consumption rate	kg/yr	14	0	100
Livestock water intake rate				
Meat	L/d	50	0	500
Milk	L/d	160	0	500
Meat and poultry consumption rate	kg/yr	63	0	300
Shielding factor for external gamma	-	0.7	0	1

TABLE 1.1 (Cont.)

Parameter	Unit	Default Value	Lower[a] Bound	Upper[a] Bound
Elapsed time of waste placement	yr	0	0	100
Initial concentration of principal radionuclide	pCi/g	d	0	∝
Drinking water intake rate	L/yr	510	0	10,000
Fraction of drinking water from site	-	1	0	1
Fraction of aquatic food from site	-	0.5	0	1
Mass loading for foliar deposition	g/m^3	1×10^{-4}	0	1
Depth of soil mixing layer	m	0.15	0	1
Fraction from groundwater				
Drinking water	-	1	0	1
Livestock water	-	1	0	1
Irrigation water	-	1	0	1

[a] The lower and upper bound values represent the lower and upper limit of an input value that can be entered to RESRAD (onsite) code. They do not represent the actual limit of the parameter value physically. For some secondary (derived) parameters (e.g., leach rate), the upper and lower bounds are derived from other primary (basic) parameters (e.g., thickness of contaminated zone).

[b] A hyphen indicates that the parameter is dimensionless.

[c] A negative value of “-1” for this parameter serves as a flag in RESRAD (onsite) code. See the section in the handbook on the particular parameter for details.

[d] The default value is radionuclide-dependent.

TABLE 1.2 Default Values, Lower Bounds, and Upper Bounds for RESRAD-OFFSITE Input Parameters

Parameter	Unit	Default Value	Lower[a] Bound	Upper[a] Bound
Soil bulk density				
Cover material	g/cm^3	1.5	0.001	22.5
Contaminated zone	g/cm^3	1.5	0.001	22.5
Unsaturated zone	g/cm^3	1.5	0.001	22.5
Saturated zone	g/cm^3	1.5	0.001	22.5
Building foundation material	g/cm^3	2.4	0	22.5
Fruit, grain, nonleafy agriculture area	g/cm^3	1.5	0.001	22.5
Leafy vegetable agriculture area	g/cm^3	1.5	0.001	22.5
Pasture, silage growing area	g/cm^3	1.5	0.001	22.5
Grain growing area	g/cm^3	1.5	0.001	22.5
Offsite dwelling area	g/cm^3	1.5	0.001	22.5
Total porosity				
Cover material	-[b]	0.4	0	1
Contaminated zone	-	0.4	0.00001	1
Unsaturated zone	-	0.4	0.00001	1
Saturated zone	-	0.4	0.00001	1
Building foundation material	-	0.1	0.0001	1
Fruit, grain, nonleafy agriculture area	-	0.4	0.00001	1
Leafy vegetable agriculture area	-	0.4	0.00001	1
Pasture, silage growing area	-	0.4	0.00001	1
Grain growing area	-	0.4	0.00001	1
Offsite dwelling area	-	0.4	0.0001	1
Effective porosity				
Contaminated zone	-	0.4	0.00001	1
Saturated zone	-	0.2	0.00001	1
Unsaturated zone	-	0.2	0.00001	1
Field capacity				
Contaminated zone	-	0.3	0.00001	1
Unsaturated zone	-	0.3	0.00001	1
Hydraulic conductivity				
Contaminated zone	m/yr	10	0.001	1×10^{10}
Unsaturated zone	m/yr	10	0.001	1×10^{10}
Saturated zone	m/yr	100	0.001	1×10^{10}
Effective radon diffusion coefficient				
Cover material	m^2/s	2×10^{-6}	-1	1
Contaminated zone	m^2/s	2×10^{-6}	-1	1
Building foundation material	m^2/s	3×10^{-7}	-1	1
Fruit, grain, nonleafy agriculture area	m^2/s	2×10^{-6}	0	1
Leafy vegetable agriculture area	m^2/s	2×10^{-6}	0	1
Pasture, silage growing area	m^2/s	2×10^{-6}	0	1
Grain growing area	m^2/s	2×10^{-6}	0	1
Offsite dwelling area	m^2/s	2×10^{-6}	0	1

TABLE 1.2 (Cont.)

Parameter	Unit	Default Value	Lower[a] Bound	Upper[a] Bound
Radon emanation coefficient (Radon-222/Radon-220)	-	0.25/0.15	0.01	1
Precipitation rate	m/yr	1	0	10
Runoff coefficient (contaminated zone, agricultural areas, livestock feed growing areas, and ~~off~~site dwelling area)	-	0.2	0	1
Irrigation rate (contaminated zone, agricultural areas, and livestock feed growing areas)	m/yr	0.2	0	10
Evapotranspiration coefficient (contaminated zone, agricultural areas, and livestock feed growing areas)	-	0.5	0	0.999
Soil-specific *b* parameter				
Contaminated zone	-	5.3	0	15
Unsaturated zone	-	5.3	0	15
Hydraulic gradient of saturated zone to well and surface water body	-	0.02	1×10^{-10}	10
Length of contaminated zone parallel to the aquifer flow	m	100	0.0001	1,000,000
Well-pump intake depth[c]	m	10	0.00001	1,000
Surface water body intake depth[c]	m	10	0	1,000
Radon vertical dimension of mixing	m	2	0.001	1,000
Average annual wind speed	m/s	2	0.001	20
Average building air exchange rate	1/h	0.5	0	1,000
Building room height	m	2.5	0.001	100
Building indoor area factor	-	0	0	100
Thickness of uncontaminated unsaturated zone	m	4	0.01	10,000
Building foundation thickness	m	0.15	0	10
Foundation depth below ground surface	m	-1	-100	100
Fraction of time spent indoors on-site	-	0	0	1

TABLE 1.2 (Cont.)

Parameter	Unit	Default Value	Lower[a] Bound	Upper[a] Bound
Fraction of time spent outdoors on-site	-	0	0	1
Fraction of time spent indoors in off-site dwelling	-	0.5	0	1
Fraction of time spent outdoors off-site	-	0.1	0	1
Fraction of time spent in fruit, grain, nonleafy fields	-	0.1	0	1
Fraction of time spent in leafy vegetable fields	-	0.1	0	1
Fraction of time spent in pasture and silage fields	-	0.1	0	1
Fraction of time spent in livestock grain fields	-	0.1	0	1
Area of primary contamination and other areas	m^2	d	e	e
Cover depth	m	0	0	100
Distribution coefficient	cm^3/g	e	0	1×10^{34}
Shape of the primary contaminated area[f]	-	polygon	-	-
Radionuclide concentration in groundwater[g]	pCi/L	NA	NA	NA
Leach rate	1/yr	0	0	1×10^{34}
Mass loading for inhalation	g/m^3	1×10^{-4}	0	2
Milk consumption rate	L/yr	92	0	1,000
Shielding factor for inhalation	-	0.4	0	1
Soil ingestion rate	g/yr	36.5	0	10,000
Thickness of contaminated zone	m	2	1×10^{-5}	1,000
Radiation dose limit	mrem/yr	25	-	-
Seafood consumption rate				
Fish	kg/yr	5.4	0	1,000
Other seafood	kg/yr	0.9	0	100
Fruit, vegetable, and grain consumption rate	kg/yr	160	0	1,000

TABLE 1.2 (Cont.)

Parameter	Unit	Default Value	Lower[a] Bound	Upper[a] Bound
Inhalation rate	m^3/yr	8,400	0	20,000
Leafy vegetable consumption rate	kg/yr	14	0	100
Meat and poultry consumption rate	kg/yr	63	0	300
Shielding factor for external gamma	-	0.7	0	1
Initial concentration of principal radionuclide	pCi/g	d	0	1×10^{20}
Drinking water intake rate	L/yr	510	0	1,000
Fraction of drinking water from site	-	1	0	1
Fraction of aquatic food from site	-	0.5	0	1
Livestock pasture and silage intake				
Meat	kg/d	14	0	300
Milk	kg/d	44	0	300
Livestock grain intake				
Meat	kg/d	54	0	300
Milk	kg/d	11	0	300
Livestock water intake rate				
Meat	L/d	50	0	500
Milk	L/d	160	0	500
Livestock soil intake from pasture and silage				
Meat	kg/d	0.1	0	10
Milk	kg/d	0.4	0	10
Livestock soil intake from grain				
Meat	kg/d	0.4	0	10
Milk	kg/d	0.1	0	10
Mass loading for foliar deposition	g/m^3	1×10^{-4}	0	1
Depth of soil mixing layer	m	0.15	0	1
Depth of roots				
Fruit, grain, nonleafy	m	1.2	0	10
Leafy vegetables	m	0.9	0	3
Pasture and silage	m	0.9	0	3
Grain	m	1.2	0	10

TABLE 1.2 (Cont.)

Parameter	Unit	Default Value	Lower[a] Bound	Upper[a] Bound
Fraction from groundwater				
Drinking water	-	1	0	1
Household water	-	1	0	1
Livestock water	-	1	0	1
Irrigation water	-	1	0	1

[a] The lower and upper bound values represent the lower and upper limit of an input parameter that can be used in RESRAD-OFFSITE. For some secondary (derived) parameters (e.g., primary contaminated area), the upper and lower bounds are derived from other primary (basic) parameters (e.g., *x* and *y* coordinates).

[b] A hyphen indicates that the parameter is dimensionless.

[c] Defined as the depth of the aquifer contributing to the well or the surface water body.

[d] The default value is calculated from other input parameters (*x* coordinates and *y* coordinates).

[e] The default value is radionuclide dependent.

[f] See the section in this handbook on the particular parameter for details.

[g] Initially there is groundwater contamination.

This page intentionally left blank.

2 HYDROGEOLOGICAL PARAMETERS

The parameters discussed in this section include soil density, total porosity, effective porosity, hydraulic conductivity, soil-specific exponential b parameter, erosion rate, hydraulic gradient, length of contaminated zone parallel to the aquifer flow, watershed area for nearby stream or pond, water table drop rate, well-pump intake depth, thickness of uncontaminated unsaturated zone, distribution coefficient, and leach rate.

2.1 SOIL DENSITY

Bulk density of dry soil (often referred to as dry density in the text) is used in RESRAD (onsite) and RESRAD-OFFSITE codes. Bulk density is related to the soil particle density and total porosity. This section provides descriptions of the techniques and procedures for measuring both types of densities (dry density and soil particle density) because both are required for calculating total porosity of the soil material.

2.1.1 Definition

Density, as applied to any kind of homogeneous monophasic material of mass *M* and volume *V*, is expressed as the ratio of *M* to *V*. Under specified conditions, this definition leads to unique values that represent a well-defined property of the material. For heterogeneous and multiphasic materials, such as porous media, however, application of this definition can lead to different results, depending on the exact way the mass and volume of the system are defined.

Soil is a typical heterogeneous multiphasic porous system which, in its general form, contains three natural phases: (1) the solid phase or the soil matrix (formed by mineral particles and solid organic materials); (2) the liquid phase, which is often represented by water and which could more properly be called the soil solution; and (3) the gaseous phase, which contains air and other gases. In this three-phase soil system, the concept of average density can be used to define the following densities: (1) density of solids or soil particle density, ρ_s; (2) bulk or dry density, ρ_b; and (3) total or wet density, ρ_t.

The masses and volumes associated with the three soil phases must be defined before the definitions of the different densities that characterize the soil system can be formalized. Thus, consider a representative elementary volume (REV) of soil that satisfies the following criteria (Bear 1972; Marsily 1986):

1. A sufficiently large volume of soil, containing a large number of pores, so that the concept of mean global properties is applicable, and

2. A sufficiently small volume of soil so that the variation of any parameter of the soil from one part of the domain to another can be approximated by a continuous function.

Within a REV, the masses of the phases composing the soil can be defined as follows:

M_s = the mass of solids,

M_l = the mass of liquids,

M_g = the mass of gases (negligible compared with the masses of the solid and liquid phases), and

$M_t = M_s + M_l$ = the total mass.

Similarly, within the REV, the volumes associated with the soil phases can be defined as follows:

V_s = the volume of solids,

V_l = the volume of liquids,

V_g = the volume of gases,

$V_p = V_l + V_g$ = the volume of pore space, and

$V_t = V_s + V_l + V_g$ = the total volume.

These mass and volume definitions can be used to define the concepts of soil particle density, bulk (dry) soil density, and total (wet) soil density. The dimensional unit of soil density is mass per unit of cubic length ($M1^{-3}$).

2.1.1.1 Soil Particle Density

The soil particle density, ρ_s, or the density of solids, represents the density of the soil (i.e., mineral) particles collectively and is expressed as the ratio of the solid phase mass to the volume of the solid phase of the soil. Soil particle density is defined as follows:

$$\rho_s = \frac{M_s}{V_s} \tag{2.1.1}$$

The soil particle density in soil minerals ranges from about 1.8 to 3.2 g/cm^3 (United States Department of Agriculture [USDA] 2009). However, in most mineral soils, the soil particle density has a narrow range of 2.6–2.7 g/cm^3 (Hillel 1980b). This density is close to that of quartz, which is usually the predominant constituent of sandy soils. A typical value of 2.65 g/cm^3 has been suggested to characterize the soil particle density of a general mineral soil (Freeze and Cherry 1979; USDA 2009). Aluminosilicate clay minerals have particle density variations in the same range. The presence of iron oxides and other heavy minerals increases the value of the soil particle density. Goethite, a common iron oxyhydroxide soil mineral, has a particle density of 4.2 g/cm^3 (USDA 2009). The presence of solid organic materials in the soil decreases the value.

2.1.1.2 Bulk (Dry) Density

The soil bulk or dry density, ρ_b, is the ratio of the mass of the solid phase of the soil (i.e., dried soil) to its total volume (solid and pore volumes together) and is defined as follows:

$$\rho_b = \frac{M_s}{V_t} = \frac{M_s}{V_s + V_l + V_g} \tag{2.1.2}$$

The bulk density, ρ_b, is related to the soil particle density, ρ_s, by the total soil porosity, p_t, according to the following equation:

$$\rho_b = (1 - p_t)\rho_s \, , \tag{2.1.3}$$

where $(1-p_t)$ is the ratio of the solid volume (V_s) to the total volume ($V_l + V_g + V_s$). Section 2.2 of this report discusses total porosity.

From the above definition, it should be obvious that the value of the dry density is always smaller than the value of the soil particle density. For example, if the volume of the pores ($V_l + V_g$) accounts for half of the total volume, the value of dry density is half the value of the soil particle density.

Typical values of dry density in different types of soils and in concrete are shown in Table 2.1.1. Dry density depends on the structure of the soil matrix (or its degree of compaction or looseness) and on the soil matrix's swelling/shrinkage characteristics. Sandy soils, because of less pore space compared to silt or clay soils, have relatively high bulk density. Soils that are loose, porous, or rich in organic matter have lower bulk density (USDA Natural Resources Conservation Service [NRCS] 2008). High bulk density is an indicator of low soil porosity and soil compaction. It may cause restrictions to plant growth and poor movement of air and water through the soil (USDA NRCS 2008). Because of its high degree of aggregation (i.e., small total porosity), concrete has, in general, a higher dry density than soil.

To use Table 2.1.1 to estimate dry bulk density (or any other soil properties discussed in this handbook), the user needs to know the soil texture type. The common method used in the field to classify a soil is the "feel" method (Brady 1984). This method consists of merely rubbing the soil between the thumb and fingers. Usually it is helpful to wet the sample to estimate plasticity more accurately. The way a wet soil "slicks out," that is, develops a continuous ribbon when pressed between the thumb and fingers, gives a good idea of the amount of clay present. The slicker the wet soil, the higher the clay content. The sand particles are gritty, and the silt has a floury or talcum-powder feel when dry and is only slightly plastic and sticky when wet. Persistent cloddiness is generally the result of the presence of silt and clay. The accuracy of the feel method depends largely on experience. The laboratory method is more accurate but is time-consuming. The laboratory method to classify soil involves particle-size analysis, in which sieves are usually employed for coarser particles and the rate of settling in water for finer particles (Marshall and Holmes 1979). The USDA has developed a method for naming soils on the basis of particle-size analysis. The relationship between such an analysis and soil class names is shown diagrammatically in Figure 2.1.1. The legend in the figure explains the use of this soil texture triangle.

TABLE 2.1.1 Dry Bulk Density (g/cm^3) for Different Soil Types from Different Sources[a]

Soil Type	NUREG/CR-6697 (Yu et al. 2000)	Linsley et al. (1982); Poffijn (1988)
Sand	1.51	1.52
Loamy sand	1.56	NA[b]
Sandy loam	1.56	1.44
Sandy clay loam	1.62	NA
Loam	1.51	1.36
Silt loam	1.46	1.28
Silt	1.43	NA
Clay loam	1.56	1.28
Silty clay loam	1.51	NA
Sandy clay	1.64	NA
Silty clay	1.70	NA
Clay	1.64	1.20
Generic soil type	1.52	NA
Concrete	NA	2.40

[a] The values are rounded to three significant digits.

[b] NA = not applicable.

2.1.1.3 Total (Wet) Density

The total, or wet, density of soil, ρ_t, is the ratio of the total mass of soil (with interstitial liquid) to its total volume and can be defined as follows:

$$\rho_t = \frac{M_t}{V_t} = \frac{M_s+M_l}{V_s+V_l+V_g} \tag{2.1.4}$$

Total density differs from dry density in that it is strongly dependent on the moisture content of the soil. For a dry soil, total density approximates the value of dry density.

2.1.2 Measurement Methodology

For use in RESRAD (onsite), only the dry densities of five distinct materials (cover layer, contaminated zone, unsaturated and saturated zones, and building foundation material) are needed as input parameters. For the RESRAD-OFFSITE code, the dry densities of agricultural areas (fruit, grain, nonleafy, and leafy vegetables) and livestock field areas (pasture, silage, and grain) are also required. However, because information on both soil particle and bulk (i.e., dry) density is required for calculating total porosity of the soil material, descriptions of the techniques and procedures for measuring both types of densities follow.

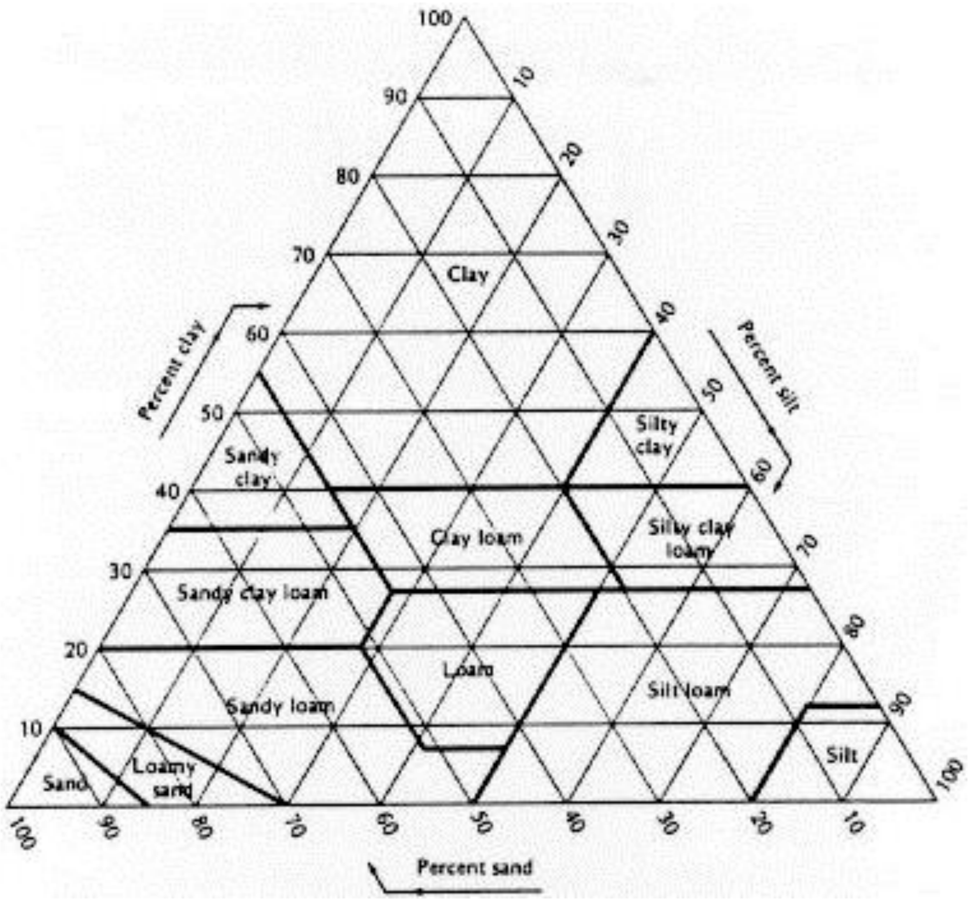


FIGURE 2.1.1 U.S. Department of Agriculture Method for Naming Soils (Note: Percentages of sand, silt, and clay in the major soil textural classes are shown. To use the diagram, locate the percentage of clay first and project inward as shown by the arrow. Do the same for the percentage of silt [or sand]. The point at which the two projections cross will identify the class name.) Source: Brady 1984

The standard methods used on Formerly Utilized Sites Remedial Action Program (FUSRAP) sites for determining the particle density and the dry density in soil materials are those prepared by the American Society of Testing Materials (ASTM 2007; 2008a, b, c; 2009; 2010a, b, c) and the U.S. Department of the Army (DOA 1970), as listed in Table 2.1.2. A general discussion of these measurement methodologies is also provided by Blake and Hartge (1986a,b).

TABLE 2.1.2 Standard Methods for Measuring Particle Density and Bulk (Dry) Density in Soil Materials at FUSRAP Sites

Parameter Measured	Type of Measurement	Standard Test Method	Reference
Soil particle density	Soil sample testing	Appendix IV: Specific Gravity	DOA (1970)
		ASTM D 854–10: Standard Test Method for Specific Gravity of Soil Solids by Water Pycnometer	ASTM (2010a)
Bulk (dry) soil density	Soil sample testing	Appendix II: Unit Weights, Void Ratio, Porosity, and Degree of Saturation	DOA (1970)
		ASTM D 7263–09: Standard Test Methods for Laboratory Determination of Density (Unit Weight) of Soil Specimens	ASTM (2009)
	In situ near-surface testing	ASTM D 1556–07: Standard Test Method for Density and Unit Weight of Soil in Place by the Sand-Cone Method	ASTM (2007)
		ASTM D 2167–08: Standard Test Method for Density and Unit Weight of Soil in Place by the Rubber Balloon Method	ASTM (2008a)
		ASTM D 6938–10 : Standard Test Methods for In-Place Density and Water Content of Soil and Soil-Aggregate by Nuclear Methods (shallow depth)	ASTM (2010b)
		ASTM D 2937–10: Standard Test Method for Density of Soil in Place by the Drive-Cylinder Method	ASTM (2010c)
		ASTM D 4564–08e1: Standard Test Method for Density and Unit Weight of Soil in Place by the Sleeve Method	ASTM (2008b)
	In situ below-surface testing	ASTM D 5195–08: Standard Test Method for Density of Soil and Rock In-Place at Depths Below Surface by Nuclear Methods	ASTM (2008c)

2.1.2.1 Soil Particle Density Measurement

The soil particle density of a soil sample is calculated on the basis of the measurement of two quantities: (1) M_s, the mass of the solid phase of the sample (dried mass), and (2) V_s, the volume of the solid phase (Blake and Hartge 1986b). Assuming that water is the only volatile in a soil sample, the mass (M_s) can be obtained by drying the sample (usually at 110 ± 5°C) until it reaches a constant weight, W_s. This method may not be valid for organic soils or soils with asphalt.

The solid phase volume, V_s, can be measured in different ways. One way is to measure the volume directly by observing the resulting increase in the volume of water as the sample of dried soil is introduced into a graduated flask that initially contains pure water (or another liquid). After making sure that the soil/water mixture is free from air bubbles, the observed expansion in volume (i.e., the replaced volume of water) should be equal to V_s, the solid phase volume. The problem with this approach is that the techniques used to eliminate air bubbles from the mixture (such as heating) can also disturb the total volume and thus introduce errors into the calculations.

Another way to measure the solid phase volume (V_s) is to evaluate the mass and density of water (or another fluid) displaced by the sample (after being oven-dried). This second approach has been used for quite some time and is simple, direct, and accurate if done carefully (Blake and Hartge 1986a). It is based on the fact that if V_{dw}, the volume of water displaced by the solids, is equal to V_s, then

$$V_{dw} = \frac{M_{dw}}{\rho_w} = V_s = \frac{M_s}{\rho_s} \qquad (2.1.5)$$

and

$$\rho_s = \rho_w \frac{M_s}{M_{dw}} \quad , \qquad (2.1.6)$$

where

M_{dw} =mass of the displaced water, and

ρ_w = water density.

Therefore, to obtain the soil particle density, it is necessary to evaluate the water density at the specific pressure and temperature conditions and to measure M_s and M_{dw} (DOA 1970, Appendix IV; ASTM 2010a).

The value of M_{dw} is obtained by using a graduated volumetric flask and taking the following measurements:

M_f = mass of the empty flask,

M_{fs} = mass of the flask plus the dried soil sample,

M_{fsw} = mass of the flask plus the soil after filling with water up to a fixed volume, V_f, and

M_{fw} = mass of the flask filled with pure water up to the fixed volume V_f.

The mass of the displaced water, M_{dw}, can then be calculated as follows:

$$M_{dw} = \left(M_{fs} - M_f\right) - \left(M_{fsw} - M_{fw}\right) \tag{2.1.7}$$

Substituting M_{dw} into the expression for soil particle density, ρ_s, yields

$$\rho_s = \rho_w \left[\frac{M_s}{(M_{fs} - M_f) - (M_{fsw} - M_{fw})}\right] \tag{2.1.8}$$

This method is very precise, but it requires careful measuring of volumes and masses and consideration of the effects of pressure and temperature conditions on the water density. Possible errors can result not only from determining the masses and volumes but from non-representative sampling.

2.1.2.2 Dry Density Measurement

The dry (bulk) density (ρ_b) of a soil sample is evaluated on the basis of two measured values: (1) M_s, the oven-dried mass of the sample, and (2) V_t, the field volume or the total volume of the sample. As stated previously, for the calculation of soil particle density (ρ_s), mass (M_s) is measured after drying the sample at 110 ± 5° C until a near constant weight is reached. This laboratory technique directly determines the dry density of a soil sample (DOA 1970, Appendix II). Possible direct methods of measuring the dry density include the core and excavation methods, which essentially consist of drying and weighing a known volume of soil.

Variations of these methods are related to different ways of collecting the soil sample and measuring volume. In the core method (Blake and Hartge 1986a; ASTM 2010c), a cylindrically shaped metal sampler is introduced into the soil, with care to avoid disturbing the sample. At the desired depth in the soil, a known field volume (V_t) of soil material is collected as it exists in situ. The sample is then oven-dried and weighed to obtain the mass. The value of the dry density is calculated by dividing the mass by the volume. Problems in using this technique include sampling difficulties, such as the presence of gravels in the soil, and the possibility of disturbing the structure of the soil during the sampling process when the sampler is introduced into the ground.

In the excavation method (Blake and Hartge 1986a), the dry density of the soil is determined by excavating a hole in the ground, oven-drying and weighing the amount of soil removed from the ground to determine the mass, and measuring the volume of the excavation. The volume (V_t) can be determined in different ways. One is to use the sand-funnel method (ASTM 2007), in which a selected type of sand with a known volume per unit mass is used to completely fill the hole. Then, by measuring the total mass of sand needed to fill the hole, the volume can be determined. Another possible way to measure the volume (V_t) is to use the rubber-balloon method (ASTM 2008a). In this technique, a balloon is placed within the hole and filled with a liquid (water) up to the borders of the hole. The volume of the excavated soil sample is then equal to the volume of the liquid in the balloon.

An advantage of using the excavation method to measure dry densities of soils other than the core method is that it is more suitable for heterogeneous soils with gravels.

An indirect method of measuring soil density, applicable for in situ rather than laboratory determinations, is called the radiation method or gamma-ray attenuation densitometry (Blake and Hartge 1986a; ASTM 2010b; ASTM 2008c). This method is based on the principle that the amount of gamma radiation attenuated and scattered in the soil depends on the soil properties, including the combined densities of the solid/liquid components of the medium. By measuring the radiation that is transmitted through the medium or that is scattered by soil components and reaches a detector placed away from the source, and by using proper calibration, the wet density of the soil, ρ_t, can be determined. To determine the dry density, ρ_b, a correction of the result is needed to delete the contribution from the liquid phase of the soil.

The radiation method used for measuring soil density has several advantages over other related laboratory techniques: (1) it yields an in situ evaluation of soil density, (2) it causes minimum disturbance of the soil, (3) it requires a relatively short measurement time, (4) it is more applicable for deeper subsoil determinations because it requires minimal excavation, and (5) it is a nondestructive technique because continuous or repeated measurements can be performed at the same spot. The radiation method also has some disadvantages compared with the other methods. Because it is a more sophisticated technique, it requires expensive equipment and highly trained operators who must be able to handle the frequent calibration procedures, the electronics, and the sampling equipment. The system operator must be trained in the radiation aspects and radiological protection procedures of the entire operation.

2.1.3 Data Input Requirements

In the RESRAD (onsite) code, one variable is assigned to represent the dry density, measured in units of grams per cubic centimeter, of each of the following five materials: (1) cover material, (2) contaminated zone, (3) unsaturated zone, (4) saturated zone, and (5) building foundation material (i.e., concrete). Density of soil in the contaminated zone, together with radionuclide concentrations, determines the total radioactive material inventory. For the RESRAD-OFFSITE code, the dry densities of agricultural areas (fruit, grain, nonleafy, and leafy vegetables) and livestock field areas (pasture, silage, and grain) are also required.

For different types of soil, a default value of 1.5 g/cm^3 is assigned for the dry density, a value that is representative of a sandy soil. Although the building foundation material (i.e., concrete) has a solid phase density (i.e., particle density) similar to that of the soil, because of its small total porosity, concrete has, in general, a higher dry density than soils. In RESRAD (onsite) and RESRAD-OFFSITE, a default value of 2.4 g/cm^3 is assigned for the dry density of the foundation building material. This default value is provided for generic use of the codes. For more accurate use of the codes, site-specific values for dry density should be experimentally determined by using one of the methods described in Section 2.1.2.2. If a site-specific value is not available, use knowledge of soil type to obtain a slightly more accurate estimate of dry density with data presented in Table 2.1.1. If neither site-specific value nor soil type is known, then use default value. For the probabilistic analysis, use the distributions developed for density of soil of different soil types in NUREG/CR-6697 (Yu et al. 2000).

2.2 TOTAL POROSITY

2.2.1 Definition

The total porosity of a porous medium is the ratio of the pore volume to the total volume of a representative sample of the medium. Assuming that the soil system comprises three phases— solid, liquid (water), and gas (air)—where V_s is the volume of the solid phase, V_l is the volume of the liquid phase, V_g is the volume of the gaseous phase, $V_p = V_l + V_g$ is the volume of the pores, and $V_t = V_s + V_l + V_g$ is the total volume of the sample, then the total porosity of the soil sample, p_t, is defined as follows:

$$p_t = \frac{V_p}{V_t} = \frac{V_l + V_g}{V_s + V_l + V_g} \quad . \tag{2.2.1}$$

Porosity is a dimensionless quantity and can be reported either as a decimal fraction or as a percentage. Table 2.2.1 lists representative total porosity ranges for various geologic materials. A more detailed list of representative porosity values (total and effective porosities) is provided in Table 2.2.2. In general, total porosity values for unconsolidated materials lie in the range of 0.25–0.7 (25%–70%). Coarse-textured soil materials such as gravel and sand tend to have a lower total porosity than fine-textured soils such as silts and clays. The total porosity in soils is not a constant quantity because the soil, particularly clayey soil, alternately swells, shrinks, compacts, and cracks. The porosity of a soil depends on several factors, such as (1) packing density, (2) the particle size distribution, (3) particle shape, and (4) cementing. Table 2.2.3 shows

TABLE 2.2.1 Range of Porosity Values

Soil Type	Porosity, p_t
Unconsolidated deposits	
Gravel	0.25–0.40
Sand	0.25–0.50
Silt	0.35–0.50
Clay	0.40–0.70
Rocks	
Fractured basalt	0.05–0.50
Karst limestone	0.05–0.50
Sandstone	0.05–0.30
Limestone, dolomite	0.00–0.20
Shale	0.00–0.10
Fractured crystalline rock	0.00–0.10
Dense crystalline rock	0.00–0.05

Source: Freeze and Cherry (1979).

TABLE 2.2.2 Representative Porosity Values

	Total Porosity, p_t		Effective Porosity,[a] p_e	
Material	Range	Arithmetic Mean	Range	Arithmetic Mean
Sedimentary material				
Sandstone (fine)	0.14–0.49	0.33	0.02–0.40	0.21
Sandstone (medium)	0.30–0.44	0.37	0.12–0.41	0.27
Siltstone	0.21–0.41	0.35	0.01–0.33	0.12
Sand (fine)	0.25–0.53	0.43	0.01–0.46	0.33
Sand (medium)	0.29–0.49	0.39	0.16–0.46	0.32
Sand (coarse)	0.31–0.46	0.39	0.18–0.43	0.30
Gravel (fine)	0.25–0.38	0.34	0.13–0.40	0.28
Gravel (medium)	0.24–0.44	0.32	0.17–0.44	0.24
Gravel (coarse)	0.24–0.36	0.28	0.13–0.25	0.21
Silt	0.34–0.61	0.46	0.01–0.39	0.20
Clay	0.34–0.57	0.42	0.01–0.18	0.06
Limestone	0.07–0.56	0.30	~0–0.36	0.14
Wind-laid material				
Loess	0.44–0.57	0.49	0.14–0.22	0.18
Eolian sand	0.40–0.51	0.45	0.32–0.47	0.38
Tuff	0.07–0.55	0.41	0.02–0.47	0.21
Igneous rock				
Weathered granite	0.34–0.57	0.45	-[b]	-
Weathered gabbro	0.42–0.45	0.43	-	-
Basalt	0.03–0.35	0.17	-	-
Metamorphic rock				
Schist	0.04–0.49	0.38	0.22–0.33	0.26

[a] Effective porosity is discussed in Section 2.3 of this handbook.

[b] A hyphen indicates that no data are available.

Sources: Morris and Johnson (1967), McWorter and Sunada (1977).

the relationship of total porosity to physical characteristics for the case of silica sand (Gibb et al. 1984).

2.2.2 Measurement Methodology

A standard method approved by the U.S. Army Corps of Engineers for determining the total porosity of soil materials, and used on FUSRAP sites, is described in Appendix II of DOA (1970). Further discussion of this methodology is presented by Danielson and Sutherland (1986).

TABLE 2.2.3 The Relationship between Physical Characteristics of Grains and Total Porosity

	Total Porosity		
Property	Low	High	Reason
Size	NA[a]	NA	Grain size has no influence on porosity.
Sorting	Poor	Good	Small grains fill in voids between large grains.
Packing	Close	Loose	Close packing leads to fewer voids between grains.
Shape	Spherical	Oblong	Spherical grains tend to pack more closely.
Roundness	Rounded	Angular	Rounded grains tend to pack more closely.

[a] NA = not applicable.

Source: Gibb et al. (1984).

On the basis of the definition of total porosity, a soil sample could be evaluated for total porosity by directly measuring the pore volume (V_p) and the total volume (V_t). The total volume is easily obtained by measuring the total volume of the sample. The pore volume can, in principle, be evaluated directly by measuring the volume of water needed to completely saturate the sample. In practice, however, it is always difficult to saturate the soil sample exactly and completely and, therefore, the total porosity of the sample is rarely evaluated by a direct method. Usually, the total porosity is derived by using the following expression (DOA 1970, Appendix II; Danielson and Sutherland 1986):

$$p_t = \left(1 - \frac{V_s}{V_t} \right) = \left(1 - \frac{\rho_b}{\rho_s} \right) , \tag{2.2.2}$$

where

p_t = a decimal fraction,

V_s = soil particle volume,

V_t = total volume,

ρ_s = solid phase (soil particle) density, and

ρ_b = dry bulk density of the sample.

Equation (2.2.2) can be obtained by rearranging Equation (2.1.3) in the soil density section of this handbook. Using this approach, the values of ρ_s and ρ_b are evaluated by laboratory or in situ measurements (Section 2 in the soil density section [2.1.2] of this handbook) and are then used to calculate the total porosity p_t.

2.2.3 Data Input Requirements

To use the RESRAD (onsite) and RESRAD-OFFSITE codes, the user is required to define, or use the default values of, the total porosity of five distinct materials: (1) cover material, (2) contaminated zone, (3) unsaturated zone, (4) saturated zone, and (5) building foundation material (i.e., concrete). For the RESRAD-OFFSITE code, the total porosity of soil materials in agricultural areas (fruit, grain, nonleafy, and leafy vegetables), livestock field areas (pasture, silage, and grain), and off-site dwelling areas are also required. In both codes, the total porosities are entered as decimal fractions rather than as percentages. RESRAD (onsite) and RESRAD-OFFSITE adopt the following values as defaults: $p_t = 0.4$ for the four soil materials listed above and for agricultural and livestock field areas and dwelling areas in the RESRAD-OFFSITE code; a default value of $p_t = 0.1$ for the building foundation (i.e., concrete) is used in both codes. These default values are listed in Table 1.1 and are provided for generic use of the codes. For more accurate use of the codes, site-specific data should be used. The total porosity is negatively correlated with bulk (dry) density, Equation (2.2.2). For the probabilistic analysis, use the distributions developed for porosity of different soil types in NUREG/CR-6697 (Yu et al. 2000).

Site-specific values for total porosity should be experimentally determined according to the method presented in Section 2.2.2. If a site-specific value is not available, use knowledge of soil type to obtain a slightly more accurate estimate of total porosity with data presented in Tables 2.2.1 and 2.2.2. If neither site-specific value nor soil type is known, then use default value.

2.3 EFFECTIVE POROSITY

2.3.1 Definition

The effective porosity, p_e, also called the kinematic porosity, of a porous medium is defined as the ratio of the part of the pore volume where the water can circulate to the total volume of a representative sample of the medium. In naturally porous systems such as subsurface soil, where the flow of water is caused by the composition of capillary, molecular, and gravitational forces, the effective porosity can be approximated by the specific yield, or drainage porosity, which is defined as the ratio of the volume of water drained by gravity from a saturated representative sample of the soil to the total volume of the sample.

The definition of effective (kinematic) porosity is linked to the concept of pore fluid displacement rather than to the percentage of the volume occupied by the pore spaces. The pore volume occupied by the pore fluid that can circulate through the porous medium is smaller than the total pore space, and, consequently, the effective porosity is always smaller than the total porosity. In a saturated soil system composed of two phases (solid and liquid) where (1) V_s is the volume of the solid phase, (2) $V_w = (V_{iw} + V_{mw})$ is the volume of the liquid phase, (3) V_{iw} is the volume of immobile pores containing the water adsorbed onto the soil particle surfaces and the water in the dead-end pores, (4) V_{mw} is the volume of the mobile pores containing water that is free to move through the saturated system, and (5) $V_t = (V_s + V_{iw} + V_{mw})$ is the total volume, the effective porosity can be defined as follows:

$$P_e = \frac{V_{mw}}{V_t} = \frac{V_{mw}}{V_s + V_{mw} + V_{iw}} \quad (2.3.1)$$

Another soil parameter related to the effective soil porosity is the field capacity, θ_r, also called specific retention, irreducible volumetric water content, or residual water content, which is defined as the ratio of the volume of water retained in the soil sample, after all downward gravity drainage has ceased, to the total volume of the sample. Considering the terms presented above for a saturated soil system, the total porosity p_t and the field capacity θ_r can be expressed, respectively, as follows:

$$P_t = \frac{V_{mw} + V_{iw}}{V_t} \quad (2.3.2)$$

and

$$\theta_r = \frac{V_{iw}}{V_t} \quad (2.3.3)$$

Therefore, the effective porosity is related to the total porosity and the field capacity according to the following expression:

$$P_e = P_t - \theta_r \quad (2.3.4)$$

Several aspects of the soil system influence the value of its effective porosity: (1) the adhesive water on minerals, (2) the absorbed water in the clay-mineral lattice, (3) the existence of unconnected pores, and (4) the existence of dead-end pores. The adhesive water in the soil is that part of the water present in the soil that is attached to the surface of the soil grains through the forces of molecular attraction (Marsily 1986). The sum of the volumes of the adhesive and absorbed water plus the water that fills the unconnected and dead-end pores constitutes the volume of the adsorbed water, V_{iw}, that is unable to move through the system.

Table 2.3.1 lists the representative values of dry bulk density, total porosity, and effective porosity for common aquifer matrix materials. A detailed list of representative porosity values (total porosity and effective porosity) is also presented in Table 2.2.2 (see Section 2.2) in this handbook).

2.3.2 Measurement Methodology

Determination of the effective porosity, p_e, of soils can be accomplished indirectly by measuring the total porosity, p_t, and the field capacity, θ_r, and then calculating p_e from Equation (2.3.4). The total porosity is obtained indirectly by measuring the soil densities according to the method described in Section 2.1. To determine the field capacity of the soils, the soil sample is first saturated with water and is then allowed to drain completely under the action of gravity until it gets to its irreducible saturation. The value of θ_r can then be obtained according to the methods used for measuring volumetric water content (Section 2.15).

TABLE 2.3.1 Representative Values of Dry Bulk Density, Total Porosity, and Effective Porosity for Common Aquifer Matrix Materials

Aquifer Matrix	Dry Bulk Density Range (g/cm^3)	Total Porosity Range	Effective Porosity Range
Clay	1.00–2.40	0.34–0.60	0.01–0.2
Peat	-	-	0.3–0.5
Glacial sediments	1.15–2.10	-	0.05–0.2
Sandy clay	-	-	0.03–0.2
Silt	-	0.34–0.61	0.01–0.3
Loess	0.75–1.60	-	0.15–0.35
Fine sand	1.37–1.81	0.26–0.53	0.1–0.3
Medium sand	1.37–1.81	-	0.15–0.3
Coarse sand	1.37–1.81	0.31–0.46	0.2 0.35
Gravely sand	1.37–1.81	-	0.2–0.35
Fine gravel	1.36–2.19	0.25–0.38	0.2–0.35
Medium gravel	1.36–2.19	-	0.15–0.25
Coarse gravel	1.36–2.19	0.24–0.36	0.1–0.25
Sandstone	1.60–2.68	0.05–0.30	0.1–0.4
Siltstone	-	0.21–0.41	0.01–0.35
Shale	1.54–3.17	0.0–0.10	-
Limestone	1.74–2.79	0.0–0.50	0.01–0.24
Granite	2.24–2.46	-	-
Basalt	2.00–2.70	0.03–0.35	-
Volcanic tuff	-	-	0.02–0.35

Source: Domenico and Schwartz 1990.

2.3.3 Data Input Requirements

To use RESRAD (onsite) and RESRAD-OFFSITE, the user is required to define (or to use the default values of) the effective porosity of two distinct geologic materials: (1) saturated zone and (2) unsaturated zone. For the RESRAD-OFFSITE code, the effective porosity of the contaminated zone is also required. In both codes, the effective porosity values are entered as decimal fractions rather than as percentages. As a default value, the codes adopt the value of $p_e = 0.2$ for all three geologic materials. These default values are listed in Tables 1.1 and 1.2, respectively, for RESRAD (onsite) and RESRAD-OFFSITE and are provided for generic use of the codes. For more accurate use of the codes, site-specific data should be used. The effective porosity is positively correlated with the total porosity and negatively correlated with field capacity, Equation (2.3.4). For the probabilistic analysis, use the distributions developed for effective porosity of different soil types in NUREG/CR-6697 (Yu et al. 2000).

Site-specific values for effective porosity should be experimentally determined according to the method presented in Section 2.3.2. Effective porosity values should not be greater than total porosity values. If a site-specific value is not available, use knowledge of soil type to obtain a slightly more accurate estimate of effective porosity with data presented in Table 2.2.2 (Section 2.2). If neither site specific value nor soil type is known, then use default value.

2.4 HYDRAULIC CONDUCTIVITY

2.4.1 Definition

The hydraulic conductivity of a soil is a measure of the soil's ability to transmit water when subjected to a hydraulic gradient. Hydraulic conductivity is defined by Darcy's law, which, for one-dimensional vertical flow, can be written as follows:

$$\boldsymbol{U} = -\boldsymbol{K}\frac{dh}{dz}, \tag{2.4.1}$$

where ***U*** is Darcy's velocity (or the average velocity of the soil fluid through a geometric cross-sectional area within the soil), h is the hydraulic head, and z is the vertical distance in the soil. The coefficient of proportionality, *K*, in Equation (2.4.1) is called the hydraulic conductivity. The term coefficient of permeability is also sometimes used as a synonym for hydraulic conductivity. On the basis of Equation (2.4.1), the hydraulic conductivity is defined as the ratio of Darcy's velocity to the applied hydraulic gradient. The dimension of K is the same as that for velocity, that is, length per unit of time (LT^{-1}).

Hydraulic conductivity is one of the hydraulic properties of the soil; the other involves the soil's fluid retention characteristics. These properties determine the behavior of the soil fluid within the soil system under specified conditions. More specifically, the hydraulic conductivity determines the ability of the soil fluid to flow through the soil matrix system under a specified hydraulic gradient; the soil fluid retention characteristics determine the ability of the soil system to retain the soil fluid under a specified pressure condition.

The hydraulic conductivity depends on the soil grain size, the structure of the soil matrix, the type of soil fluid, and the relative amount of soil fluid (saturation) present in the soil matrix. The important properties relevant to the solid matrix of the soil include pore size distribution, pore shape, tortuosity, specific surface, and porosity. In relation to the soil fluid, the important properties include fluid density, ρ, and fluid viscosity, μ. For a subsurface system saturated with the soil fluid, the hydraulic conductivity, ***K***, can be expressed as follows (Bear 1972):

$$\boldsymbol{K} = \frac{kpg}{\mu}, \tag{2.4.2}$$

where *k*, the intrinsic permeability of the soil, depends only on properties of the solid matrix, and $\rho g/\mu$, called the fluidity of the liquid, represents the properties of the percolating fluid. The hydraulic conductivity, ***K***, is expressed in terms of length per unit of time (LT^{-1}), the intrinsic permeability, *k*, is expressed in L^2, and the fluidity, $\rho g/\mu$, in $L^{-1}T^{-1}$. By using Equation (2.4.2), Darcy’s law can be rewritten explicitly in terms of its coefficient of proportionality (hydraulic conductivity *K*):

$$\boldsymbol{K} = \frac{kpg}{\mu} = \frac{|U|}{|dh/dz|} \tag{2.4.3}$$

When the fluid properties of density and viscosity are known, Equation (2.4.3) can be used to experimentally determine the value of the intrinsic permeability, *k*, and the hydraulic conductivity, ***K***, as will be shown in the following section.

The values of saturated hydraulic conductivity in soils vary within a wide range of several orders of magnitude, depending on the soil material. Table 2.4.1 lists the range of expected values of ***K*** for various unconsolidated and consolidated soil materials. The expected representative values of ***K*** for soil materials of different textures are presented in Table 2.4.2. A more detailed list of expected representative values of ***K*** based on the grain size distribution, degree of sorting, and silt content of several soil materials is presented in Table 2.4.3 and Table 2.4.4. The default hydraulic conductivity values are listed in Table 2.4.5. Newell et al. (1990) developed a hydrogeologic database from a technical survey of 400 sites across the United States and presented the box plot of hydraulic conductivity for 12 groups of hydrogeologic environments (see Figure 8 in Newell et al. [1990] for details.). Their research indicates that the hydraulic conductivity follows a lognormal distribution. The median of the national distribution of hydraulic conductivity is 0.005 cm/s (or 1.58×10^3 m/yr).

Because of the spatial variability usually found in the geological formation of soils, saturated hydraulic conductivity values also show variations throughout the space domain within a subsurface geological formation. Such a geological formation is said to be heterogeneous. If the properties of the geologic formation are invariable in space, the formation is homogeneous. A geological formation is said to be isotropic if at any point in the medium, the values of the saturated hydraulic conductivity (***K***) are independent of the direction of measurement. Again, because of the usually stratified nature of unconsolidated sedimentary soil materials, soils are usually anisotropic. Within an anisotropic geological formation, the vertical component of the saturated hydraulic conductivity is usually smaller (by one to two orders of magnitude) than the horizontal component.

TABLE 2.4.1 Range of Saturated Hydraulic Conductivity of Various Soil Materials

Soil Type	Saturated Hydraulic Conductivity, *K* (m/yr)
Unconsolidated deposits	
Gravel	1×10^4–1×10^7
Clean sand	1×10^2–1×10^5
Silty sand	1×10^1–1×10^4
Silt, loess	1×10^{-2}–1×10^2
Glacial till	1×10^{-5}–1×10^1

TABLE 2.4.1 (Cont.)

Soil Type	Saturated Hydraulic Conductivity, K (m/yr)
Unweathered marine clay	1×10^{-5}–1×10^{-2}
Rocks	
Shale	1×10^{-6}–1×10^{-2}
Unfractured metamorphic and igneous rocks	1×10^{-7}–1×10^{-3}
Sandstone	1×10^{-3}–1×10^{1}
Limestone and dolomite	1×10^{-2}–1×10^{1}
Fractured metamorphic and igneous rocks	1×10^{-1}–1×10^{3}
Permeable basalt	1×10^{1}–1×10^{5}
Karst limestone	1×10^{1}–1×10^{5}

Source: Adapted from Freeze and Cherry (1979).

TABLE 2.4.2 Representative Values of Saturated Hydraulic Conductivity of Different Soil Textures

Texture	Saturated Hydraulic Conductivity, K (m/yr)
Sand	5.55×10^{3}
Loamy sand	4.93×10^{3}
Sandy loam	1.09×10^{3}
Silty loam	2.27×10^{2}
Loam	2.19×10^{2}
Sandy clay loam	1.99×10^{2}
Silty clay loam	5.36×10
Clay loam	7.73×10
Sandy clay	6.84×10
Silty clay	3.21×10
Clay	4.05×10

Source: Clapp and Hornberger (1978).

TABLE 2.4.3 Estimated Saturated Hydraulic Conductivities for Fine-Grained Materials

Grain-Size Class	Saturated Hydraulic Conductivity, K (10^{3} m/yr)
Clay	<0.0001
Silt, clayey	0.1–0.4

TABLE 2.4.3 (Cont.)

Grain-Size Class	Saturated Hydraulic Conductivity, K (10^3 m/yr)
Silt, slightly sandy	0.5
Silt, moderately sandy	0.8–0.9
Silt, very sandy	1.0–1.2
Sandy silt	1.2
Silty sand	1.4

Source: EPA (1986).

TABLE 2.4.4 Estimated Saturated Hydraulic Conductivities for Sands and Gravels According to Degree of Sorting and Silt Content[a]

Grain-Size Class or Range	Saturated Hydraulic Conductivity, K (10^3m/yr)					
	Degree of Sorting			Silt Content		
	Poor	Moderate	Well	Slight	Moderate	High
Very fine sand	1	2	3	3	2	1
Very fine to fine sand	3	3	-[b]	3	2	1
Very fine to medium sand	4	5	-	4	3	2
Very fine to coarse sand	5	-	-	4	3	3
Very fine to very coarse sand	7	-	-	6	4	3
Very fine sand to fine gravel	8	-	-	7	6	4
Very fine sand to medium gravel	11	-	-	9	7	5
Very fine sand to coarse gravel	14	-	-	12	10	7
Fine sand	3	4	6	4	3	2
Fine to medium sand	6	7	-	5	4	3
Fine to coarse sand	6	8	-	6	5	4
Fine to very coarse sand	8	-	-	7	5	4
Fine sand to fine gravel	10	-	-	8	7	5
Fine sand to medium gravel	13	-	-	10	8	6
Fine sand to coarse gravel	16	-	-	12	10	8
Medium sand	7	9	10	7	6	4
Medium to coarse sand	8	10	-	8	6	5
Medium to very coarse sand	9	12	-	8	7	5
Medium sand to fine gravel	11	-	-	9	8	6
Medium sand to medium gravel	15	-	-	13	9	7
Medium sand to coarse gravel	18	-	-	15	12	9
Coarse sand	9	12	15	10	8	6
Coarse to very coarse sand	10	15	-	10	8	6
Coarse sand to fine gravel	13	16	-	12	10	8
Coarse sand to medium gravel	16	-	-	13	10	8
Coarse sand to coarse gravel	20	-	-	15	11	10
Very coarse sand	12	16	21	13	10	8

TABLE 2.4.4 (Cont.)

	Saturated Hydraulic Conductivity, K (10^3m/yr)					
	Degree of Sorting			Silt Content		
Grain-Size Class or Range	Poor	Moderate	Well	Slight	Moderate	High
Very coarse to fine gravel	15	24	-	13	12	10
Very coarse to medium gravel	19	25	-	16	14	11
Very coarse sand to coarse gravel	23	-	-	18	15	12
Fine gravel	18	24	30	25	16	12
Fine to medium gravel	22	37	-	22	19	15
Fine to coarse gravel	27	37	-	26	21	16
Medium gravel	27	26	45	27	22	18
Medium to coarse gravel	33	52	-	33	27	21
Coarse gravel	37	52	67	37	32	26

[a] Reduce conductivities by 10% if grains are subangular.

[b] A hyphen indicates that no data are available.

Source: EPA(1986).

TABLE 2.4.5 Default Hydraulic Conductivity Values Used in RESRAD (onsite) and RESRAD-OFFSITE

Parameter Name	Unit	Default Value	Code-Accepted Values	References	Description
Contaminated-zone hydraulic conductivity for Cover and Contaminated Zone	m/yr	10	10^{-3}–10^{10}	Yu et al. 1993; EPA 1996	The measure of the soil's ability to conductively transmit water when subjected to a hydraulic gradient. The hydraulic conductivity depends on the soil grain size, the structure of the soil matrix, the type of soil fluid, and the relative amount of soil fluid (saturation) present in the soil matrix.
Saturated-zone hydraulic conductivity	m/yr	100	10^{-3}–10^{10}	Yu et al. 1993; EPA 1996	See contaminated-zone hydraulic conductivity (above).
Unsaturated-zone hydraulic conductivity	m/yr	10	10^{-3}–10^{10}	Yu et al. 1993; EPA 1996	See contaminated-zone hydraulic conductivity (above).

2.4.2 Measurement Methodology

The saturated hydraulic conductivity, *K*, of water in soil (or the intrinsic permeability, *k*, of the soil) can be measured by both field and laboratory experiments. Either way, the experimental measurement of ***K*** (or *k*) consists in determining the numerical value for the coefficient in Darcy's equation.

The methodology used for the experimental determination of ***K*** (or *k*) in either laboratory or field experiments is based on the following procedures (Bear 1972):

1. Assume a flow pattern (such as one-dimensional flow in a porous medium) that can be described analytically by Darcy's law;

2. Perform an experiment reproducing the chosen flow pattern and measure all measurable quantities in Equation (2.4.3), including fluid density, dynamic viscosity, flow velocity, and the gradient of the hydraulic head; and

3. Compute the coefficient *K* (or *k*) by substituting the measured quantities into Equation (2.4.3).

Many different laboratory or field experiments can be used to determine the coefficient *K* (or *k*).

An extensive discussion on the respective measurement methodologies for laboratory and field experiments is presented by Klute and Dirksen (1986) and Amoozegar and Warrick (1986), respectively. For FUSRAP sites, the standard methods used for determining saturated hydraulic conductivity in soil materials are those described by the American Society for Testing and Materials (ASTM 1992a-o), the U.S. Environmental Protection Agency (EPA 1986), the U.S. Department of the Army (DOA 1970), and the U.S. Department of the Interior (DOI 1990a,b). Brief descriptions of these pertinent standard methods are presented in Table 2.4.6.

Laboratory tests are carried out on small samples of soil materials collected during core-drilling programs. Because of the small sizes of the soil samples handled in the laboratory, the results of these tests are considered a point representation of the soil properties. If the soil samples used in the laboratory test are truly undisturbed samples, the measured value of ***K*** (or *k*) should be a true representation of the in situ saturated hydraulic conductivity at that particular sampling point.

Laboratory methods may be used to evaluate the vertical and horizontal hydraulic conductivity in soil samples. For instance, in undisturbed samples of either cohesive or cohesionless soils, the values of ***K*** obtained through laboratory tests correspond to the direction in which the sample was taken, that is, generally vertical. The conductivity of disturbed (remolded) samples of cohesionless soils obtained in the laboratory can be used to approximate the actual value of ***K*** in the undisturbed (natural) soil in the horizontal direction (DOA 1970). For fine-grained soils, the undisturbed cohesive sample can be oriented accordingly, to obtain the hydraulic conductivity in either the vertical or horizontal direction.

In contrast to laboratory methods for measuring conductivity in soil samples, field methods, in general, involve a large region of the soil. Consequently, the results obtained from field methods should reflect the influences of both the vertical and horizontal directions and should represent an average value of ***K***. This situation is especially important in highly stratified soils where the values of ***K*** measured from field methods would reflect the domination of the most permeable layer in the soil profile. However, by appropriately selecting the specific method to be used in the field, the in situ values of the vertical and horizontal components of ***K*** could be determined independently in each layer of stratified soils.

Selection of a specific method for a particular application will depend on the objectives to be achieved. Because of the difficulty in obtaining a perfectly undisturbed sample of unconsolidated soil, the ***K*** value determined by laboratory methods may not accurately reflect the respective value in the field. Therefore, field methods should be used whenever the objective is to characterize the physical features of the subsurface system in question as accurately as possible. Field methods, however, are usually more expensive than laboratory methods and, consequently, when the question of cost becomes decisive, or when actual representation of field conditions is not of fundamental importance and in situ hydraulic conductivity is not available, laboratory methods may be used to determine the saturated hydraulic conductivity ***K***. The inclusion of more factors such as mud content (Rosas et al. 2015) into empirical hydraulic equations has shown improvement of accuracy of the ***K*** estimate. Note that for saturated conditions, field methods are superior to laboratory measurements; however, even field measurements can represent local conditions (e.g., slug tests) and can have errors associated with them (e.g., skin effects). According to Rovey and Niemann (2001), slug tests may be thought of as representing the harmonic mean or one-dimensional flow; tracer tests, the geometric mean or two-dimensional flow, and multi-well pumping tests, the arithmetic mean or the three-dimensional flow field. Depending on the scale being represented and the problem being solved, one method may be preferred over another.

TABLE 2.4.6 Standard Laboratory and Field Methods for Measuring Saturated Hydraulic Conductivity, *K*, in Soil Materials

Method Type	Method Specification	Application	Remarks	References
Laboratory	Constant-head conductivity test with permeameter cylinder	Disturbed (remolded) samples of cohesionless coarse-grained soils with $K > 1.0 \times 10^2$ m/yr.	The conductivity of disturbed (remolded) cohesionless soil is generally used to approximate the conductivity of its original, undisturbed state in a horizontal direction.	DOA (1970) EPA (1986) ASTM (1992m) Klute and Dirkson (1986)
	Falling-head conductivity test with permeameter cylinder	Disturbed (remolded) samples of cohesionless fine-grained soils with $K < 1.0 \times 10^2$ m/yr.	The conductivity of disturbed (remolded) cohesionless soil is generally used to approximate the conductivity of its original, undisturbed state in a horizontal direction.	DOA (1970) EPA (1986) ASTM (1992m) Klute and Dirkson (1986)
	Conductivity test with sampling tubes	Undisturbed samples of cohesionless soil that cannot be removed from the sampling tube without excessive disturbance.	The measured conductivity corresponds to the direction in which the sample was taken (generally vertical); may be performed under constant-head or falling-head flow conditions, depending on the estimated conductivity of the sample.	DOA (1970)
	Conductivity test with pressure chamber	Cohesive fine-grained soil samples in the undisturbed, disturbed (remolded), or compacted state in a fully saturated condition.	Should be used only in soils that are originally fully saturated; can be performed under conditions of loading expected in the field; leakage along the sides of the sample can be prevented; usually performed under falling-head flow conditions.	DOA (1970) EPA (1986)
	Conductivity test with back pressure	Cohesive fine-grained soil samples in the undisturbed, disturbed (remolded), or compacted state that are not fully saturated.	The additional pressure (back pressure) applied to the pore fluid of the soil sample reduces the size of the gas bubbles in the pores, increasing the degree of water saturation; usually performed under constant-head flow conditions.	DOA (1970) EPA (1986) ASTM (1992m)
	Conductivity test with consolidometer	Cohesive fine-grained soil samples in a fully saturated condition.	Can be used as an alternative method to the conductivity test with pressure chamber.	DOA (1970)

TABLE 2.4.6 (Cont.)

Method Type	Method Specification	Application	Remarks	References
Laboratory	Grain-size-based empirical method	To evaluate the intrinsic permeability, *k*, in disturbed samples of soil materials with known grain-size distribution. [After determining k, the saturated hydraulic conductivity, K, can then be evaluated from Equation (2.4.2).]	The intrinsic permeability, *k*, can be predicted from the expression $k = cd^a$, where *c* = constant found through regression analysis; *d* = the mean or particle diameter; and *a* = exponent constant, ranging from 1.65 to 1.85.	ASTM (1992n)
Field	Auger-hole method	Saturated soil materials near the ground surface in the presence of a shallow water table.	The method consists of pumping the water out of an auger-hole extending below the water table and then measuring the rate of the rise of the water in the hole; most widely used procedure to measure the saturated hydraulic conductivity in saturated soils; the measured result is dominated by the average value of the horizontal conductivity of the profile.	Amoozegar and Warrick (1986)
	Piezometer method	Saturated soil materials near the ground surface in the presence of a shallow water table.	The method consists of installing a piezometer tube or pipe into an auger hole with a cavity at the bottom; water is removed from the tube and the rate of the rise of the water in the tube is measured; can be used to measure either horizontal or vertical hydraulic conductivity; in stratified soils, the method can be used to measure *K* in each individual layer.	Amoozegar and Warrick (1986)

TABLE 2.4.6 (Cont.)

Method Type	Method Specification	Application	Remarks	References
	Single-well (slug) test in moderately permeable formations under unconfined conditions	Saturated soil materials of moderate *K* in aquifers under unconfined conditions.	Pump-out test method developed primarily for groundwater systems; the method consists of removing a slug of water instantaneously from a well and measuring the recovery of the water in the well; applicable to wells that fully or partially penetrate the interval of interest in the unconfined aquifer; the measured *K* primarily reflects the value in the horizontal direction.	EPA (1986)
Field	Single-well (slug) test in moderately permeable formations under confined conditions	Saturated soil materials of moderately hydraulic conductivity in testing zones under confined conditions, entirely open to the well screen or open borehole.	Pump-out test method developed primarily for groundwater systems; the method consists of removing a lug of water instantaneously from a well and measuring the recovery of the water in the well; used in confined aquifer (saturated zone of the soil under confined conditions); the method assumes that the tested zone is uniform in all radial directions from the test well.	EPA (1986)
	Single-well (modified slug) test in extremely tight formations under confined conditions	Saturated soil materials with low to extremely low conductivity such as silts, clays, and shales. (For K as low as 1.0 × 10-5 m/yr).	Pump-out test method developed primarily for groundwater systems; the test is conducted by suddenly pressurizing a packed-off zone of the soil in a portion of a borehole or well within the confined zone and then monitoring the pressure decay afterwards; used in confined aquifer (saturated zone of the soil under confined conditions).	EPA (1986)

TABLE 2.4.6 (Cont.)

Method Type	Method Specification	Application	Remarks	References
	Constant-head conductivity test by the well permeameter method (also referred to as shallow-well pumping, or dry-auger hole, method)	To measure field-saturated hydraulic conductivity of soil materials in the unsaturated (vadose) zone near the ground surface. Soil types ranging from sand, silt and clay mixtures, with *K* larger than 1.0×10^2 m/yr, to relatively clean sand or sandy gravel with K <1.0×10^4 m/yr.	Pump-in test consisting of measuring the rate at which water flows out of an uncased well into the soil under constant-head flow conditions; specially used to determine the field-saturated hydraulic conductivity in unsaturated zones of the soil (but can also be used in saturated zones); for a very high groundwater condition, a "pump-out" test for saturated soils is often more satisfactory than any "pump-in" type of test; the calculated K is dominated by the conductivity of the most permeable layer of the soil profile; in uniform soils, the measured K reflects the conductivity in the horizontal direction; requires a large quantity of water and a long time (several days) for execution.	Amoozegar and Warrick (1986) ASTM (1992) DOI (1990a)
Field	Double-tube method	To measure field-saturated hydraulic conductivity of oil materials in the unsaturated (vadose) zone, near the ground surface.	Utilizes two concentric cylinders installed in an auger hole; water is introduced into these cylinders and *K* is evaluated by measuring the flow in the cylinders; can measure field saturated K in the horizontal and vertical directions; the method requires over 200 L of water and two to six hours for completion.	Amoozegar and Warrick (1986) ASTM (1992n)
Field	Cylindrical permeameter method (also referred to as ring infiltrometer test method)	To measure field-saturated hydraulic conductivity of soil materials in the unsaturated (vadose) zone near the ground surface. Soil materials with *K* ranging between 1.0×10^{-3} and 1.0×10^3 m/yr.	The method consists of ponding water within a cylindrical ring placed over the soil surface and measuring the volumetric rate of water needed to maintain a constant head; measures the field-saturated *K* in the vertical direction near the ground surface; a time-consuming procedure, requiring in excess of 100 L of water; variations of the method include the single-ring and double-ring infiltrometers.	Amoozegar and Warrick (1986) ASTM (1992i,n)

TABLE 2.4.6 (Cont.)

Method Type	Method Specification	Application	Remarks	References
	Air-entry permeameter method	To measure field-saturated hydraulic conductivity of soil materials in the unsaturated (vadose) zone near the ground surface.	Fast technique to determine the field saturated *K*; requires approximately 10 L of water; is a variation of the single-ring infiltrometer method.	Amoozegar and Warrick (1986) ASTM (1992n)
	Constant-head conductivity test in single drill hole	To measure field-saturated hydraulic conductivity of soil materials at any depth within the unsaturated (vadose) zone. Soil or rock materials with K ranging between 1.0×10^2 and 1.0×10^4 m/yr.	Pump-in test consisting of injecting water into an isolated interval of a drill hole in soil or rock under constant-head flow conditions; the only currently available test that can measure field-saturated *K* at large depths within the unsaturated zone; designed to determine an approximate value of *K* in a specific interval of a drill hole.	Amoozegar and Warrick (1986) ASTM (1992n) DOI (1990b)

2.4.2.1 Laboratory Methods

In the laboratory, the value of *K* can be determined by several different instruments and methods such as the permeameter, pressure chamber, and consolidometer (DOA 1970). A common feature of all these methods is that a soil sample is placed in a small cylindrical receptacle representing a one-dimensional soil configuration through which the circulating liquid is forced to flow. Depending on the flow pattern imposed through the soil sample, the laboratory methods for measuring hydraulic conductivity are classified as either a constant-head test with a steady-state flow regimen or a falling-head test with an unsteady-state flow regimen.

Constant-head methods are primarily used in samples of soil materials with an estimated ***K*** above 1.0×10^2 m/yr, which corresponds to coarse-grained soils such as clean sands and gravels. Falling-head methods, on the other hand, are used in soil samples with estimated values of ***K*** below 1.0×10^2 m/yr (DOA 1970). A list of standard laboratory methods for determining *K*, with variations of the constant-head and falling-head flow conditions, is presented in Table 2.4.6. Also listed in Table 2.4.6, as a laboratory method for measuring ***K***, is the grain-size-based empirical method, in which the intrinsic permeability, *k*, of the soil sample is empirically determined from the otherwise laboratory-measured grain size distribution of the soil sample.

Important considerations regarding the laboratory methods for measuring ***K*** are related to the soil sampling procedure and preparation of the test specimen and circulating liquid. The sampling process, if not properly conducted, usually disturbs the matrix structure of the soil and results in a misrepresentation of the actual field conditions. Undisturbed sampling of soils is possible, but it requires the use of specially designed techniques and instruments (Klute and Dirksen 1986).

A detailed guide on the standard methods for soil sampling is presented in ASTM D 4700–91, Standard Guide for Soil Sampling from the Vadose Zone (ASTM 1992l). Relatively undisturbed soil samples, suitable for the determination of hydraulic conductivity in the laboratory, could be obtained, for example, by using the thin-walled tube sampling method in ASTM D 1587–83, Standard Practice for the Thin-Walled Tube Sampling of Soils (ASTM 1992c). In this technique, a relatively undisturbed soil sample is obtained by pressing a thin-walled metal tube into the soil, removing the soil-filled tube, and sealing its ends to prevent physical disturbance in the soil matrix.

Selecting the test fluid is also of fundamental importance for the laboratory determination of the saturated hydraulic coefficient. The objective is to have the test fluid mimic the actual properties of the soil fluid as closely as possible. When an inappropriate test fluid is selected, the test sample can get clogged with entrapped air, bacterial growths, and fines. To avoid such problems, a standard test solution such as a deaerated 0.005-mol calcium sulfate ($CaSO_4$) solution, saturated with thymol (or sterilized with another substance such as formaldehyde), should be in the permeameter, unless there are specific reasons to choose another solution (Klute and Dirksen 1986).

2.4.2.2 Constant-Head Method

The constant-head test with the permeameter is one of the most commonly used methods for determining the saturated hydraulic conductivity of coarse-grained soils in the laboratory. The test operates in accordance with the direct application of Darcy's law to a soil liquid configuration representing a one-dimensional, steady flow of a percolating liquid through a saturated column of soil from a uniform cross-sectional area. In this method, a cylindrical soil sample of cross-sectional area *A* and length *L* is placed between two porous plates that do not provide any extra hydraulic resistance to the flow. A constant head difference, $H_2 - H_1$, is then applied across the test sample. By measuring the volume *V* of the test fluid that flows through the system during time *t*, the saturated hydraulic conductivity *K* of the soil can be determined directly from Darcy's equation:

$$K = \frac{VL}{\left[At\left(H_2 - H_1\right)\right]} \tag{2.4.4}$$

To improve the results, it is recommended that the test be performed several times under different head differences, $H_2 - H_1$. It is also recommended that the quantity of liquid collected be sufficient to provide at least three significant figures in the measured volume. In a simple version of the constant-head permeameter, the lower limit of the measurement of *K* is approximately 1×10^1 m/yr, which corresponds to the lower limit of the conductivity of sandy clay soils. For lower values of *K*, it is recommended that either an enhanced version of the constant-head permeameter (i.e., one that has a more sensitive method of measuring the volume flow rate) or the falling-head permeameter be used (Klute and Dirksen 1986). Table 2.4.6 presents variations of the constant-head method for measuring saturated hydraulic conductivity of soil materials in the laboratory.

2.4.2.3 Falling-Head Method

The falling-head test with the permeameter is primarily used for determining the ***K*** (or *k*) value of fine-grained soils in the laboratory. Like the constant-head method, the falling-head test also operates in accordance with direct application of Darcy's law to a one-dimensional, saturated column of soil with a uniform cross-sectional area. The falling-head method differs from the constant-head method in that the liquid that percolates through the saturated column is kept in an unsteady-state flow regimen in which both the head and the discharged volume vary during the test. In the falling-head test method, a cylindrical soil sample of cross-sectional area *A* and length *L* is placed between two highly conductive plates. The soil sample column is connected to a standpipe of cross-sectional area *a*, in which the percolating fluid is introduced into the system. Thus, by measuring the change in head in the standpipe from H_1 to H_2 during a specified interval of time *t*, the saturated hydraulic conductivity can be determined as follows (Klute and Dirksen 1986):

$$K = \left(\frac{aL}{At}\right)\ln\left(\frac{H_1}{H_2}\right) \tag{2.4.5}$$

The lower limit of ***K***, which can be measured in a falling-head permeameter, is about 1× 10^{-2} m/yr. This value corresponds approximately to the lower limit of conductivity of silts and coarse clays (Klute and Dirksen 1986).

A common problem encountered in using either the constant-head or falling-head test with the permeameter is related to the degree of saturation achieved within the soil samples during the test. Air bubbles are usually trapped within the pore space, and although they tend to disappear slowly by dissolving into the deaerated water, their presence in the system may alter the measured results. Therefore, after using these instruments to measure *K*, it is always recommended that the degree of saturation of the sample be verified by measuring the sample's volumetric water content and comparing the result with the total porosity calculated from the particle density.

For a more accurate laboratory measurement of *K* in soil samples in which the presence of air bubbles becomes critical, the conductivity test with back pressure is recommended. In this method, additional pressure (back pressure) is applied to the pore fluid of the soil sample, which reduces the size of the gas bubbles in the pores and, consequently, increases the degree of water saturation.

2.4.2.4 Field Methods

The several methods developed for in situ determination of saturated hydraulic conductivity of soils can be separated into two groups: (1) those that are applicable to sites near or below a shallow water table and (2) those that are applicable to sites well above a deep water table or in the absence of a water table. More specifically, these groups are applicable to sites located, respectively, in the saturated and unsaturated zones of the soil. In either group (similarly to the laboratory methods), the value of ***K*** is obtained from Darcy's law after measuring the gradient of the hydraulic head at the site and the resulting soil water flux. Table 2.4.6 lists several standard methods used for in situ determination of ***K*** in saturated and unsaturated regions of the soil.

Field Methods Used in Saturated Regions of the Soil. Many in situ methods have been developed for determining the saturated hydraulic conductivity of saturated soils within a groundwater formation under unconfined and confined conditions. These methods include (1) the auger-hole and piezometer methods, which are used in unconfined shallow water table conditions (Amoozegar and Warrick 1986), and (2) well-pumping tests, which were primarily developed for the determination of aquifer properties used in the development of confined and unconfined groundwater systems (EPA 1986).

Auger-Hole Method. The auger-hole method is the field procedure most commonly used for in situ determination of saturated hydraulic conductivity of soils. This method has many possible variations (Amoozegar and Warrick 1986). In its simplest form, it consists of the preparation of a cavity partially penetrating the aquifer, with minimal disturbance of the soil. After preparation of the cavity, the water in the hole is allowed to equilibrate with the groundwater; that is, the level in the hole becomes coincident with the water table level. The actual test is started by removing the entire amount of water from the hole and measuring the rate of the rise of the water level within the cavity.

Because of the three-dimensional aspect of the flow pattern of the water near the cavity, there is no simple equation for accurately determining the conductivity. Numerous available semiempirical expressions, however, can be used for approximating the saturated hydraulic conductivity for different soil configurations. These expressions are functions of the geometrical dimensions of the auger hole and the aquifer and the measured rate at which the water level in the hole changes with time (Amoozegar and Warrick 1986).

The auger-hole method is applicable to an unconfined aquifer with homogeneous soil properties and a shallow water table. In its simplest form, this method provides an estimate of the average horizontal component of the saturated hydraulic conductivity of the soil within the aquifer. Enhanced variations of the method have been developed to account for layered soils and for the determination of either horizontal or vertical components of saturated hydraulic conductivity. Results obtained by the auger-hole method are not reliable for cases in which (1) the water table is above the soil surface, (2) artesian conditions exist, (3) the soil structure is extensively layered, or (4) highly permeable small strata occur.

Piezometer Method. The piezometer method, like the auger-hole method, is applicable for determining the saturated hydraulic conductivity of soils in an unconfined aquifer with a shallow water table level. Unlike the auger-hole method, however, the piezometer method is appropriately designed for applications in layered soil aquifers and for determining either horizontal or vertical components of the saturated hydraulic conductivity.

This method consists of installing a piezometer tube or pipe into an auger hole drilled through the subsurface system without disturbing the soil. The piezometer tube should be long enough to partially penetrate the unconfined aquifer. The walls of the piezometer tube are totally closed except at its lower extremity, where the tube is screened open to form a cylindrical cavity of radius r and height hc within the aquifer. The water in the piezometer tube is first removed to clean the system and is then allowed to equilibrate with the groundwater level.

Similarly to the auger-hole method, the piezometer method consists of removing the water from the pipe and then measuring the rate of the rise of the water within the pipe. The saturated hydraulic conductivity is then evaluated as a function of the geometrical dimension of the cavity in the piezometer tube, the dimensions of the aquifer, and the measured rate of rise of the water table in the tube. The value for the conductivity is calculated with the help of a nomograph and tables (Amoozegar and Warrick 1986).

Depending on the relative length (hc) of the cavity as compared with its radius (r), the piezometer method can be used to determine the horizontal or vertical component of the saturated hydraulic conductivity. Thus, if hc is large compared to r, the results obtained reflect the horizontal component of $\boldsymbol{K}$. Otherwise, if hc is small compared to r, then the vertical component of $\boldsymbol{K}$ is estimated. The piezometer method is especially suitable for determining the conductivity of individual layers in stratified subsurface systems.

Well-Pumping (Slug) Methods. The well-pumping (slug) test is applicable for in situ determination of the saturated hydraulic conductivity in soil materials of unconfined and confined aquifers. This method consists of removing a slug of water instantaneously from a well and measuring the recovery of the water in the well. Variations of the well-pumping test, called single-well tests (EPA 1986), are listed in Table 2.4.6.

In contrast to the auger-hole and piezometer methods, the results of which reflect an in situ average of a relatively small region of soil around the created cavity in the soil, well-pumping tests also provide an in situ representation of the soil hydraulic conductivity, but averaged over a larger representative volume of the soil. The measured results of ***K*** primarily reflect the value in the horizontal direction. (Further references for these methods can be found in EPA (1986), Freeze and Cherry (1979), and Amoozegar and Warrick (1986).)

Field Methods Used in the Unsaturated Region of the Soil. Measuring the saturated hydraulic conductivity of unsaturated soils located above the water table (or in the absence of a water table) by in situ methods is more difficult than measuring ***K*** for saturated soils. The important difference is that the original unsaturated soil must be artificially saturated to perform the measurements. An extra-large quantity of water may be needed to saturate the medium, which results in a more elaborate and time-consuming measurement. The results of these in situ measurements of ***K*** are commonly called the field-saturated hydraulic conductivity.

Many in situ methods have been developed for determining the field-saturated hydraulic conductivity of soil materials within the unsaturated (vadose) zone of the soil. As listed in Table 2.4.6, the available standard methods for measuring field-saturated ***K*** include (1) the shallow-well pump-in or dry auger-hole, (2) the double-tube, (3) the ring infiltrometer, (4) the air-entry permeameter, and (5) the constant-head test in a single drill hole. A complete guide for comparing these standard methods is presented in ASTM D5126-90, Standard Guide for Comparison of Field Methods for Determining Hydraulic Conductivity in the Vadose Zone (ASTM 1992n). Further detailed discussion on these standard methods can also be found in the work of Amoozegar and Warrick (1986).

2.4.3 Data Input Requirements

In RESRAD (onsite) and RESRAD-OFFSITE, the user is requested to input a saturated hydraulic conductivity value in units of meters per year (m/yr) for three soil materials: contaminated, unsaturated, and saturated zones.

The vertical infiltration of water within the contaminated zone and through the unsaturated region of the soil, the subsequent vertical leaching, and the transport of contaminants into the underlying aquifer are the important aspects of the problem being modeled. Consequently, in RESRAD (onsite) and RESRAD-OFFSITE, the saturated hydraulic conductivity values related to the contaminated and unsaturated zones of the soil should represent the vertical component of ***K***. For isotropic soil materials, the vertical and horizontal components of ***K*** are the same; for anisotropic soils, however, the vertical component of ***K*** is typically one or two orders of magnitude lower than the horizontal component.

The major concern within the saturated zone is related to the horizontal transport of the contaminants that have infiltrated through the unsaturated zone and reached the aquifer. Therefore, the input value for the saturated hydraulic conductivity ***(K)*** of the soil material in the saturated zone should reflect the horizontal component of ***K***.

The estimation of the values of ***K*** to be used in RESRAD (onsite) and RESRAD-OFFSITE can be performed at different levels of site-specific accuracy, depending on the amount of information available. For generic use of the code, a set of default values of ***K*** is defined as 10 m/yr for the contaminated and unsaturated zones and 100 m/yr for the saturated zone. These values approximately represent the condition of an anisotropic sedimentary soil material, that is, silt, loess, or silty sand, in which the vertical component of ***K*** is one order of magnitude lower than the horizontal component. The code accepted value for hydraulic conductivity ranges between 10^{-3} and 10^{10} m/yr for all the three soil materials, as shown in Tables 1.1.1 and 1.1.2.

If the geological stratigraphy and the soil textures at the site are known, a better (i.e., more accurate and site-specific) estimation of ***K*** can be performed with the help of Table 2.4.1 through Table 2.4.4. However, if values in the literature are used in place of actual site data, no more than one significant digit is appropriate.

For an accurate site-specific estimation of the input data for RESRAD, the values of ***K*** should be measured either in the laboratory or in field experiments according to one of the standard methods listed in Table 2.4.6.

Because of the intrinsic difficulties of the methods available for in situ measurements of field-saturated ***K*** in unsaturated regions of the soil, it is recommended that laboratory methods be used for determining the vertical component of ***K*** in the contaminated and unsaturated zones. In these cases, either variation of the constant-head or falling-head method can be used, depending solely on the actual values of ***K*** being measured. As mentioned previously, the constant-head method is more applicable for large values of ***K*** (in the range of $10^0 - 10^6$ m/yr), and the falling-head method is more applicable for lower values of ***K*** (in the range of $10^{-2} - 10^2$ m/yr).

Determination of the horizontal component of ***K*** in the saturated zone of the soil can be accomplished either by laboratory (i.e., constant-head and falling-head) or field methods (i.e., auger-hole, piezometer, and well-pumping). In the laboratory, the value of the horizontal component of ***K*** in cohesionless soil materials can be approximated by the conductivity of a disturbed soil sample obtained by the permeameter method. For cohesive soil materials, the undisturbed cohesive soil sample can then be oriented in the horizontal direction to obtain the appropriate value of ***K***. In the field, most of the methods available for the determination of ***K*** in the saturated zone will reflect the value in the horizontal direction.

The saturation ratio can be estimated by using the following equation (Clapp and Hornberger 1978):

$$R_s = \left(\frac{I}{K_{sat}} \right)^{\frac{1}{2b+3}} \qquad (2.4.6)$$

where

I = infiltration rate (m/yr),

K_{sat} = saturated hydraulic conductivity (m/yr), and

b = soil-specific exponential parameter (dimensionless).

When the medium is saturated, R_s equals unity. Under unsaturated infiltration conditions, the saturation ratio is a function of the infiltration rate, the saturated hydraulic conductivity, and the texture of the soil, as shown in Equation (2.4.6). The volumetric water content of the unsaturated zone is calculated by

$$\theta = R_s \cdot \theta_{sat} \qquad (2.4.7)$$

The calculated volumetric water content is checked against the field capacity of the unsaturated soil and porosity. The field capacity sets the lower limit of the volumetric water content and is used to replace the calculated value when the calculated value is smaller. Once the volumetric water content is set to the field capacity, the saturation ratio is recalculated by using Equation (2.4.7). The porosity, on the other hand, sets the upper boundary of the volumetric water content and is used to replace the calculated value when the calculated value is larger.

2.5 SOIL-SPECIFIC EXPONENTIAL *b* PARAMETER

2.5.1 Definition

The soil-specific exponential *b* parameter is one of several hydrological parameters used to calculate the radionuclide leaching rate of the contaminated zone. (See also precipitation rate, irrigation rate, runoff coefficient, evapotranspiration coefficient, hydraulic conductivity, and soil porosity.) The soil-specific *b* parameter is an empirical and dimensionless parameter that is used to evaluate the saturation ratio (or the volumetric water saturation), R_s, of the soil, according to a soil characteristic function called the conductivity function (i.e., the relationship between the unsaturated hydraulic conductivity, $\boldsymbol{K}$, and the saturation ratio, R_s).

It has been suggested that a power function is an acceptable form of representing the conductivity function. As cited by Clapp and Hornberger (1978), Campbell (1974) derived a partly empirical and partly theoretical conductivity function on the basis of the power function model; this function proved to be reasonably accurate over a large number of cases. Campbell suggested the following power expression to represent the working relationship for the conductivity function:

$$k = R_s^{(2b+3)}, \tag{2.5.1}$$

where

k = relative conductivity (or relative permeability, dimensionless),

R_s = saturation ratio (dimensionless), and

b = fitting parameter, called the soil-specific exponential parameter, which must be determined experimentally.

The relative conductivity, k, at any location in the unsaturated zone is defined as a ratio of the unsaturated hydraulic conductivity, $\boldsymbol{K}$, at that point, to the saturated hydraulic conductivity, ***Ksat***. Thus, k can be expressed as follows:

$$k = \frac{K}{K_{sat}} \tag{2.5.2}$$

Substituting the definition of the relative permeability k into Equation (2.5.1) yields

$$\frac{K}{K_{sat}} = R_s^{(2b+3)} \tag{2.5.3}$$

or

$$R_s = \left(\frac{K}{K_{sat}}\right)^{\left(\frac{1}{2b+3}\right)} \tag{2.5.4}$$

In downward water infiltration into the unsaturated upper layer of the soil, the infiltration rate, I_r (see also precipitation rate), can be approximated by the unsaturated hydraulic conductivity, $\boldsymbol{K}$ (Hillel 1980a). Therefore, substituting I_r for $\boldsymbol{K}$ in Equation (2.5.4) yields

$$R_s = \left(\frac{I_r}{K_{sat}}\right)^{\left(\frac{1}{2b+3}\right)} \tag{2.5.5}$$

Equation (2.5.5) is used internally in the RESRAD (onsite) model to evaluate the volumetric water saturation, R_s, in all unsaturated regions of the soil system. According to Equation (2.5.5), under unsaturated infiltration conditions, the saturation ratio R_s is a function of the infiltration rate I_r, the saturated hydraulic conductivity K_{sat}, and the texture of the soil, as determined by the fitting parameter b. When the medium is fully saturated, I_r equals K_{sat}, and R_s equals unity.

2.5.2 Measurement Methodology

The soil-specific *b* parameter is an empirical fitting parameter and, therefore, must be determined experimentally. For each type of soil, the best estimate of *b* can be obtained by adjusting the best-fit values of the soil to an experimentally determined curve of relative permeability versus saturation, according to the power function model proposed above, Equation (2.5.1).

Determining the conductivity function of a soil sample experimentally by measuring the relative permeability and the saturation is not an easy laboratory task because of many technical and procedural difficulties. Yet some data have been reported in the literature that demonstrate reasonable agreement with the proposed model. For example, Clapp and Hornberger (1978) have reported that Campbell's model (Campbell 1974) for the conductivity function has proven to be acceptable under different conditions of soil saturation over a wide range of *b* values (0.17−13.6) and even for values of saturation, R_s, near unity (i.e., full saturation). Table 2.5.1 lists representative values of the soil-specific exponential *b* parameter for various soil textures. Section 2.1.2 provides a discussion on soil textures.

2.5.3 Data Input Requirements

In RESRAD (onsite) and RESRAD-OFFSITE, the user is requested to define an input value for the soil-specific *b* parameter for (1) the contaminated zone, (2) the unsaturated zone strata, and (3) the saturated zone. Input for the saturated-zone *b* parameter is required only if the water table drop rate (Section 2.10) is greater than zero.

TABLE 2.5.1 Representative Values of Soil-Specific Exponential b Parameter

Texture	Soil-Specific Exponential Parameter, *b*
Sand	4.05
Loamy sand	4.38
Sandy loam	4.9
Silty loam	5.3
Loam	5.39
Sandy clay loam	7.12
Silty clay loam	7.75
Clay loam	8.52
Sandy clay	10.4
Silty clay	10.4
Clay	11.4

Source: Clapp and Hornberger (1978).

Reported measured data indicate that values of *b* vary within the range of 0.17–13.6 (Clapp and Hornberger 1978). A default value of 5.3 was adopted in the RESRAD (onsite) and RESRAD-OFFSITE models. This value represents the condition of a silty loam soil material. Whenever possible, however, site-specific input data for *b* should be used in the RESRAD calculation. The range of *b* accepted by the code varies from 0 to 15 for cover and contaminated zone and unsaturated zone, and from 10^{-34} to 15 for saturated zone, as shown in Table 2.5.2.

A relatively more accurate value of parameter *b* for site-specific soil materials can be obtained from the data listed in Table 2.5.1. For most applications, this approach should suffice because of the difficulties in obtaining laboratory determinations of the soil conductivity function.

2.6 EROSION RATE

2.6.1 Definition

The erosion rate is the average volume of soil material that is removed from one place to another by running water, waves and currents, wind, or moving ice per unit of ground surface area and per unit of time. The erosion rate represents the average depth of soil that is removed from the ground surface per unit of time at the site and is expressed in units of length per time (LT^{-1}).

TABLE 2.5.2 Default Soil-specific Exponential *b* Parameters Used in RESREAD (onsite) and RESRAD-OFFSITE

Parameter Name	Unit	Default Value	Code-Accepted Values	References	Description
Cover and contaminated zone b parameter	-	5.3	0–15	Yu et al. 1993; EPA 1996; Clapp and Hornberger 1978	An empirical and dimensionless parameter that is used to evaluate the saturation ratio (or the volumetric water saturation) of the soil according to a soil characteristic function called the conductivity function.
Saturated zone b parameter	-	5.3	10^{-34}–15		
Unsaturated zone, soil-specific b parameter	-	5.3	0–15		

2.6.2 Measurement Methodology

Erosion rates can be estimated by means of the Universal Soil Loss Equation (USLE), an empirical model that has been developed for predicting the rate of soil loss by sheet and rill erosion, or its revised version RUSLE (Renard et al. 1997). However, orders-of-magnitude errors can result from using the USLE method without proper orientation. An appropriate guide for using the USLE method can be obtained from the U.S. Soil Conservation Service (SCS), which conducts county soil surveys on a regular basis. The SCS office near the site should be able to provide the USLE parameters mapped out for the site-specific soils and cover types for the area of interest. In addition to the USLE model, which is commonly used to predict the average annual soil loss from a watershed, some physics-based models such as SIBERIA (Willgoose 2005) and the CHILD (Channel-Hillslope Integrated Landscape Development) (Tucker 2011) model are also available for calculating the evolution of complex topography and landscape. Although these models provide more details about soil erosion, they usually require much more information and are computationally expensive.

If sufficient site-specific data are available, a site-specific erosion rate can be calculated by using the USLE method. Wischmeier and Smith (1978) and Foster (1979) discuss details of the calculation. Estimates based on the range of erosion rates for typical sites in humid areas east of the Mississippi River (based on model site calculations for locations in New York, New Jersey, Ohio, and Missouri) can also be used (Knight 1983). For example, for a site with a 2% slope, these model calculations predict a range of 8×10^{-7} to 3×10^{-6} m/yr for natural succession vegetation, 1×10^{-5} to 6×10^{-5} m/yr for permanent pasture, and 9×10^{-5} to 6×10^{-4} m/yr for row-crop agriculture. The rate increases by a factor of about 3 for a 5% slope, 7 for a 10% slope, and 15 for a 15% slope. If these generic values are used for a farm/garden scenario in which the dose contribution from food ingestion pathways is expected to be significant, an erosion rate of 6×10^{-4} m/yr should be assumed for a site with a 2% slope. This would lead to erosion of 0.6 m of soil in 1,000 yr. A proportionately higher erosion rate must be used if the slope exceeds 2%. An erosion rate of 6×10^{-5} m/yr, leading to erosion of 0.06 m of soil in 1,000 yr, can be used for a site with a 2% slope if it can be reasonably shown that the farm/garden scenario is unreasonable; for example, if the site is, and will likely continue to be, unsuitable for agricultural use.

Erosion rates are more difficult to estimate for arid than for humid sites. Although water erosion is generally more important than wind erosion, the latter can also be significant. Water erosion in the West is more difficult to estimate because it is likely to be due to infrequent heavy rainfalls for which the empirical constants used in the USLE may not be applicable. Long-term erosion rates are generally lower for sites in arid locations than for sites in humid locations. A more detailed discussion and data on soil erosion are presented in *Soil Physics* (Marshall and Holmes 1979), *Universal Soil Loss Equation: Past, Present, and Future* (Peterson and Swan 1979), and the *Nature and Properties of Soils* (Brady 1984).

2.6.3 Data Input Requirements

In RESRAD (onsite), the user is requested to input a value for the annual average erosion rate for the cover zone and the contaminated zone. These input values of the erosion rate are given in units of meters per year (m/yr).

For generic use of the code, a default value of the annual erosion rate equal to 0.001 m/yr (as shown in Table 2.6.1) was adopted in RESRAD (onsite). This value is about the same as the national average of erosion rate on cropland in 1982, according to the national resources inventory conducted by the USDA's NRCS and the Center for Survey Statistics and Methodology (CSSM) at Iowa State University in 2012, as shown in Figure 2.6.1 (USDA NRCS and CSSM 2015). For a particular site, however, a more accurate site-specific estimation of the erosion rates for both the cover and the contaminated zones should be attempted. Some reported studies on specific sites (e.g., the study on the Western New York Nuclear Service Center [DOE 2010] and the study by Ebbert et al. [1998]) may also provide useful information for users to determine their own site-specific erosion rate. The erosion rate of the contaminated zone becomes significant only if and when the cover zone is completely eroded, thus exposing the contaminated zone to the erosive effects of the environmental elements. If there is no initial cover, a greater erosion rate will remove the contaminated material faster. This may lead to lower doses than found for an initial cover case for an on-site receptor using RESRAD (onsite) code. However, because of the transport of contamination through erosion to off-site locations, a greater erosion rate may result in higher doses for an off-site receptor using RESRAD-OFFSITE code.

A site-specific estimation of the erosion rate for the cover and contaminated zones can be performed by means of the USLE or the revised USLE (RUSLE).

The USLE is directly used in RESRAD-OFFSITE to estimate soil erosion in contaminated and agricultural areas. The USLE parameters, the rainfall and runoff factor, soil erodibility factor, the slope length-steepness factor, the cover and management factor, and support practice factor are discussed by Yu et al. (2007).

TABLE 2.6.1 Default Erosion Rate Values Used in RESRAD

Parameter Name	Unit	Default Value	Code-Accepted Values	References	Description
Cover erosion rate	m/yr	0.001	0–5	Yu et al. 1993	The average volume of cover material that is removed per unit of ground surface area and per unit of time. Erosion rates can be estimated by means of the universal soil loss equation.
Contaminated-zone erosion rate	m/yr	0.001	0–5	Yu et al. 1993	The average volume of source material that is removed per unit of ground surface area and per unit of time.

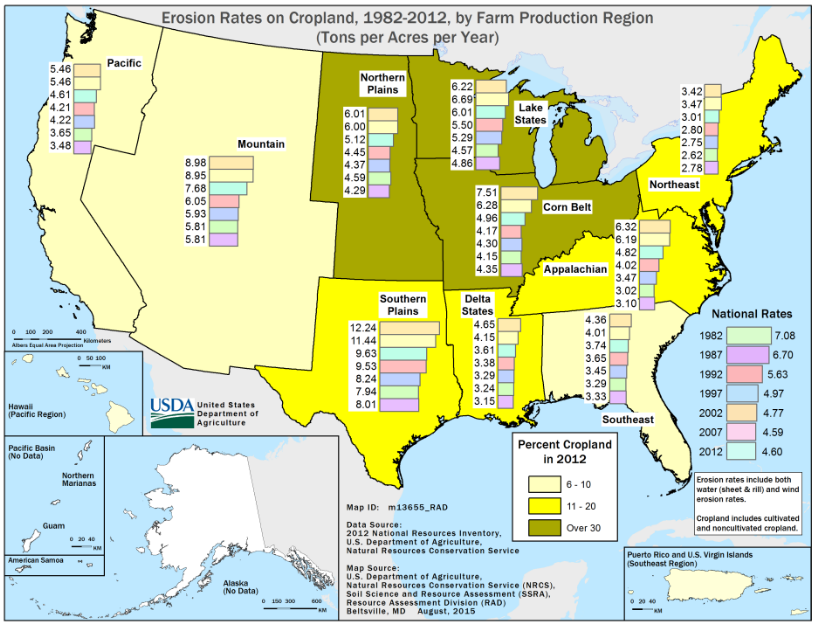


FIGURE 2.6.1 Erosion Rate on Cropland in the United States (Source: USDA NRCS and CSSM 2015)

2.7 HYDRAULIC GRADIENT

2.7.1 Definition

The hydraulic gradient is the change in hydraulic head per unit of distance of the groundwater flow in a given direction. The hydraulic gradient, J_x, in the flow direction x is expressed as follows:

$$J_x = \frac{h_1 - h_2}{\Delta x} \tag{2.7.1}$$

where h_1 and h_2 represent the hydraulic head at points 1 and 2, respectively, and Δx is the distance between these two points. Mathematically, the hydraulic gradient is a vector that can be expressed as grad h. The norm of the vector represents the maximum slope of the hydraulic gradient; its orientation represents the direction along the maximum slope. The hydraulic gradient is a dimensionless parameter, usually represented as a fraction rather than as a percentage.

In an unconfined (water table) aquifer, the horizontal hydraulic gradient of groundwater flow is approximately the slope of the water table. In a confined aquifer, it represents the difference in potentiometric surfaces over a unit distance. The potentiometric surface is the elevation to which water rises in a well that taps a confined aquifer. It is an imaginary surface analogous to a water table. In general, the hydraulic gradient of groundwater flow in a highly permeable geologic material, such as sand or gravel, is far less than that in a geologic material with a low permeability, such as silt or clay.

2.7.2 Measurement Methodology

The hydraulic head at a point in the saturated zone can be measured in the field by installing a piezometric nest at the site. A piezometer is basically a tube or pipe long enough to be introduced through the unsaturated zone down into the saturated zone. Its walls must be completely sealed along all its length, but it must be open to the atmosphere at the top and to the water flow at the bottom. The water level measured inside the piezometer, as compared with a defined reference level (such as mean sea level), gives the hydraulic head of the aquifer at the point of measurement.

The distribution of the hydraulic head in a groundwater system is actually three-dimensional. Thus, with the installation of three or more piezometers spatially distributed in an aquifer, it is possible to determine the spatial distribution of the hydraulic head at the site. If the distances between the piezometers are known, the hydraulic gradient of the dominant aquifer flow at the site can be evaluated. A detailed description of piezometer nests has been given by Freeze and Cherry (1979).

Based on the technical survey of 400 sites across the United States, Newell et al. (1990) developed a hydrogeologic database and provided the box plot of hydraulic gradient for 12 groups of hydrogeologic environments (see Figure 9 of Newell et al. [1990] for details). Their research indicates that the hydraulic gradient follows a lognormal distribution. The median of the national distribution of hydraulic gradient is 0.006 ft/ft[1].

2.7.3 Data Input Requirements

In RESRAD (onsite) and RESRAD-OFFSITE, the user is requested to input a value for the hydraulic gradient in the dominant groundwater flow direction in the underlying aquifer at the site. This parameter is dimensionless and should be entered as a decimal fraction rather than as a percentage. This parameter is needed for RESRAD (onsite) and RESRAD-OFFSITE to calculate the water flow rate per unit of cross-sectional area (i.e., Darcy velocity) in the saturated zone.

For generic use of the codes, a default value of 0.02 was adopted for the hydraulic gradient in the RESRAD (onsite) and RESRAD-OFFSITE models, as shown in Table 2.7.1. Because the hydraulic gradient varies significantly from one site to another, site-specific information should be applied for more accurate use of the code whenever possible.

---

[1] For a lognormal distribution (with median m and geometric standard deviation g), 68% of the distribution lies between m/g and $m \times g$.

Site-specific data on the hydraulic gradient and the general flow pattern of the groundwater system at the site can be obtained by installing a piezometric nest in the area, as suggested above. RESRAD users should also consider contacting a local or state hydrologist or geologist as a possible source of site-specific information.

TABLE 2.7.1 Default Hydraulic Gradient Values Used in RESRAD (onsite) and RESRAD-OFFSITE

Parameter Name	Unit	Default Value	Code-Accepted Values	Reference	Description
Saturated-zone hydraulic gradient	-	0.02	10^{-10}–10	Yu et al. 1993	The change in hydraulic head per unit of distance in the groundwater flow direction. In an unconfined (water table) aquifer, the horizontal hydraulic gradient of groundwater flow is approximately the slope of the water table. In a confined aquifer, it represents the difference in potentiometric surfaces over a unit distance.

2.8 LENGTH OF CONTAMINATED ZONE PARALLEL TO THE AQUIFER FLOW

2.8.1 Definition

The length, *l*, of the contaminated zone parallel to the aquifer flow is the maximum horizontal distance measured in the contaminated zone, from its upgradient edge to the downgradient edge, along the direction of the groundwater flow in the underlying aquifer.

The parameter *l* is used in the RESRAD (onsite) and RESRAD-OFFSITE codes to evaluate the dilution of the contaminated inflow water (which percolates the contaminated zone vertically and reaches the aquifer underneath) by the uncontaminated inflow groundwater in the Nondispersion Model for a well located near the contaminated zone.

2.8.2 Measurement Methodology

To evaluate the value of parameter *l* at a specific site, it is first necessary to determine the hydraulic gradient of groundwater flow at the site. As described in Section 2.7, the groundwater flow direction in the aquifer can be determined locally by installing a piezometric nest composed of three or more piezometers spatially distributed throughout the hydrogeological system. With a known groundwater flow direction and the horizontal extent of the contaminated zone, the parameter *l* can be determined by measuring the largest horizontal length of the contaminated zone parallel to the groundwater flow direction.

2.8.3 Data Input Requirements

In the RESRAD (onsite) and RESRAD-OFFSITE codes, the user is required to input a value of *l*, that is, the length of the contaminated zone parallel to the groundwater flow that represents the conditions at the site. The dimensions of *l* should be entered in units of meters (m).

A default value of 100 m was adopted in the RESRAD (onsite) and RESRAD-OFFSITE codes for parameter *l*. The default value of 100 m is the square root of the default contaminated zone area of 10,000 m^2. Whenever possible, however, site-specific information should be applied for more accurate use of the code.

2.9 WATERSHED AREA FOR NEARBY STREAM OR POND

2.9.1 Definition

A watershed is a region contoured by an imaginary line connecting ridges or summits of high land and drained by or draining into a river, river system, or a body of water such as a lake or pond. The watershed area is the surface area of the draining region above the discharge measuring points. This parameter is expressed in units of length squared (l^2). In the RESRAD (onsite) and RESRAD-OFFSITE codes, the watershed area parameter represents the area of the region draining into the nearby stream or pond located at the vicinity of the site.

The watershed area parameter is used in the RESRAD model to evaluate the dilution factor for the contamination of the water at the nearby stream or pond as it gets mixed with the inflow of water from the contaminated aquifer. Thus, the evaluation of the dilution factor for the ground/surface water pathway is based on the following assumptions (Gilbert et al. 1989):

1. The nearby body of water is a pond;
2. The inflow and outflow of water in the pond are in equilibrium;
3. The average annual inflow of radioactivity into the pond is equal to the average annual quantity of radioactivity that is leached from the contaminated zone into the groundwater system; and
4. The infiltrating water flow through the contaminated zone is vertically downward.

Under these conditions and assumptions, the dilution factor is then defined as the ratio of the average annual volume of water that percolates through the contaminated zone to the average annual total inflow of water into the pond. More specifically, the dilution factor is calculated internally in the code as the ratio of the contaminated zone area (AREA) to the watershed area (WAREA).

2.9.2 Measurement Methodology

The area of the watershed draining toward the pond located at the vicinity of the site can be evaluated by using a small-scale morphologic map of the region.

2.9.3 Data Input Requirements

In the RESRAD (onsite) and RESRAD-OFFSITE codes, the user is requested to input a value for the area of the watershed region draining into the stream or pond located in the vicinity of the site. The dimensions of the watershed area should be entered in units of square meters (m^2).

A default value of one million (1×10^6) m^2 for the watershed area was adopted in the RESRAD model. If found to significantly affect the results, site-specific information should be applied for more accurate use of the code.

Site-specific information on the watershed area can be obtained from small-scale hydrological and morphological maps covering the region under study. In the RESRAD codes, the watershed area must be larger than or equal to the area of the contaminated zone. The code will issue a warning if this condition is violated and will not proceed with the calculations until the violation is corrected.

2.10 WATER TABLE DROP RATE

2.10.1 Definition

The water table drop rate is the rate, in units of length per time (lT^{-1}), at which the depth of the water table is lowered. The level of the water table in a groundwater system fluctuates seasonally because of the erratically temporal variations of the processes involved in the hydrologic cycle (Section 3.1), as well as extra use of the water from the system. Under normal circumstances, the level of the water table is approximately stationary if averaged over long periods of time such as one year. For unusually high consumptive use of groundwater in the region, however, the water table may experience a significant drop during the annual period. In these cases, the average annual water table drop rate is not zero and results in the creation of an increase in the unsaturated-zone thickness over time. This process of increasing the unsaturated-zone thickness is modelled in the RESRAD (onsite) code. This parameter is not used in the RESRAD-OFFSITE code.

2.10.2 Measurement Methodology

The site-specific water table drop rate can be estimated by observing the change of the water level of a monitoring well appropriately installed at the site. It can also be estimated by consulting water table records of past decades.

2.10.3 Data Input Requirements

In RESRAD (onsite), the user is required to input a value for the average annual water table drop rate that represents conditions at the site. The water table drop rate should be expressed in units of meters per year (m/yr).

A default value of 0.001 m/yr was adopted in the RESRAD (onsite) code for the water table drop rate. This value is the same as the default value used for the erosion rate. Whenever possible, however, site-specific information should be applied for more accurate use of the code.

2.11 WELLPUMP INTAKE DEPTH

2.11.1 Definition

The parameter well-pump intake depth is the screened depth of a well within the aquifer (the saturated zone). The well-pump intake depth is measured in units of length (l). Based on the technical survey of 400 sites across the United States, Newell et al. (1990) developed a hydrogeologic database and provided the box plot of the saturated thickness of aquifer for 12 groups of hydrogeologic environments (see Figure 9 of Newell et al. [1990] for details). Their research indicates that the saturated thickness of aquifer follows a lognormal distribution. The median of the national distribution of the saturated thickness of aquifer is 30.0 ft (or 9.09 m).[1]

2.11.2 Data Input Requirements

In RESRAD (onsite), the user is required to input a value for the well-pump intake depth that represents conditions at the site. Its dimensions should be given in units of meters (m). This parameter is required for calculating the dilution factor for the nondispersion model and the well water concentration, as discussed in Appendixes E and K in Yu et al. (2001) and Chapter 3 of Yu et al, (2007). A default value of 10 m was adopted in the RESRAD model for the well-pump intake depth, as shown in Table 2.11.1. For more accurate use of the code, however, site-specific

---

[1] For a lognormal distribution (with median m and geometric standard deviation g), 68% of the distribution lies between m/g and $m \times g$.

TABLE 2.11.1 Default Well-pump Intake Depth Value Used in RESRAD

Parameter Name	Unit	Default Value	Code-Accepted Values	Reference	Description
Well-pump intake depth (below water table)	m	10	10^{-5}–1,000	Yu et al. 1993	The screened depth of a well within the aquifer (the saturated zone).

data should be applied whenever possible. In RESRAD-OFFSITE, the user is required to input a value for the depth of the aquifer contributing to the well or surface water body (Yu et al. 2007).

2.12 THICKNESS OF UNCONTAMINATED UNSATURATED ZONE

2.12.1 Definition

The uncontaminated unsaturated zone is the portion of the uncontaminated zone that lies below the bottom of the contaminated zone and above the water table. The RESRAD (onsite) and RESRAD-OFFSITE codes provide for up to five different horizontal strata within this zone. Each stratum is characterized by six radionuclide-independent parameters: (1) thickness of the layer, (2) soil density, (3) total porosity, (4) effective porosity, (5) soil-specific *b* parameter, and (6) hydraulic conductivity. Based on the technical survey of 400 sites across the United States, Newell et al. (1990) developed a hydrogeologic database and provided the box plot of the depth to top of aquifer for 12 groups of hydrogeologic environments (see Figure 9 of Newell et al. [1990] for details). Their research indicates that the depth to top of aquifer follows a lognormal distribution. The median of the national distribution of the depth to top of aquifer is 15.0 ft (or 4.55 m).[1]

2.12.2 Data Input Requirements

In RESRAD (onsite) and RESRAD-OFFSITE, the user is required to input a value for each stratum used in the calculation. Default values are supplied by the code for all parameters of an active stratum; however, the use of site-specific data is strongly recommended.

---

[1] For a lognormal distribution (with median m and geometric standard deviation g), 68% of the distribution lies between m/g and $m \times g$.

2.13 DISTRIBUTION COEFFICIENTS

2.13.1 Definition

The distribution coefficient, K_d, is the ratio of the mass of solute species adsorbed or precipitated on the solids per unit of dry mass of the soil, S, to the solute concentration in the liquids, C. The distribution coefficient represents the partition of the solute in the soil matrix and soil water, assuming that equilibrium conditions exist between the soil and solution phases. A linear Freundlich isotherm, which assumes complete reversibility of ion adsorption, has been extensively used to correlate the relationship between S and C, that is,

$$S = K_d C \, . \tag{2.13.1}$$

The transfer of radionuclides from the liquid to the solid phase or vice versa may be controlled by mechanisms such as adsorption and precipitation, depending on the radionuclides. The dimensions of the distribution coefficient are given in units of length (l) cubed per mass (M) (l^3/M).

In the literature, distribution coefficients measured from adsorption conditions abound, but it is well known that these experimental K_d values are not constant when used with soils. The time elapsed since the incorporation of the radionuclide in the soil affects the distribution coefficient because of the aging effect (IAEA 2010a); a fraction of the radionuclide may become fixed by the solid phase over time. The K_d values are dependent on the soil’s physical and chemical characteristics, which do not necessarily remain constant over the long term because soils are dynamic systems. Soil properties affecting the distribution coefficient include the texture of soils (sand, loam, clay, or organic soils) (Sheppard and Thibault 1991; Gil-Garcia et al. 2009a; Vandenhove et al. 2009; Gil-Garcia et al. 2009b); the organic matter content of the soils; pH values (Coughtrey et al. 1985; Gil-Garcia et al. 2009a; Vandenhove et al. 2009; Gil Garcia et al. 2009b); the soil solution ratio (Sheppard et al. 1983); the solution or pore water concentration (Nikula 1982; Hoeffner 1985; Sheppard et al. 1987; Sheppard and Thibault 1990); and the presence of competing cations and complexing agents (Gee et al. 1980, Nikula 1982; Gee et al. 1983; Rouston et al. 1984; Hoeffner 1985; Uchida and Kamada 1987; Bond and Smiles 1988). Because of its dependence on many soil properties, the value of the distribution coefficient for a specific radionuclide in soils can range over several orders of magnitude under different conditions. To reduce the variability, the K_d values can be grouped on the basis of fundamental soil properties, such as soil texture and organic matter content (IAEA 2010a; Sheppard 2011; Gil-Garcia et al. 2009a; Vandenhove et al. 2009; Gil Garcia et al. 2009b; Isherwood 1981).

2.13.2 Measurement Methodology

2.13.2.1 Experimental Methods

The two most common experimental techniques for the determination of K_d are the batch and column methods. Usually, the batch method is used to measure the distribution coefficient, K_d, under saturated equilibrium conditions. The column method is used to approach a more “natural” soil condition.

Batch Method. Measurement of the distribution coefficient can be performed quickly by the batch method with any radionuclide in any soil material or rock, independent of the porosity, brittleness, or other properties of the soil or rock. In most instances, the soil material or rock is continually agitated to facilitate mixing and contact. At specified times, to approach equilibrium conditions, the solid and solution are separated and the resultant distribution of the nuclide is determined. In the batch system, radionuclide desorption and adsorption are affected by the following: agitation effects (Barney and Brown 1980); solid-liquid separation techniques; and limitation of analytical determination, that is, multiple species of soil or rock cannot be differentiated if present (Serne and Relyea 1981).

The ASTM C1733 test method has been developed as a standard batch method (ASTM 2010d) to measure the distribution coefficient of inorganic species under steady-state conditions. This test method determines the K_d of chemical species by quantifying uptake onto solid materials by batch sorption techniques. This method can be applied directly to unconsolidated material samples or to disaggregated portion of samples. The sorption is strongly dependent on concentration of the species of interest in solution, pH, temperature, rock and soil properties including mineralogy (surface charge and energy), particle size distribution, and biological conditions. The method recommends considering all ionic species present in the migrating solution and using groundwater representative of the test zone (but containing added tracers) as contact liquid. The method also recommends running each concentration in duplicate, doing the analysis for five or more concentrations, and determining the time required for the tracer/solid system to achieve constant concentrations at the highest tracer concentration to be used in the experiment. The method further recommends keeping a 25:1 liquid-to-mass ratio, measuring the liquid in terms of mass, and collecting a small aliquot of liquid each time for analysis. The contact periods should differ by at least a one-day period. Before taking a sample for analysis, shake the mixture and allow it to settle for several minutes. Remove an appropriate quantity of liquid, filter it, and keep it for analysis. The soil solution mixture can also be separated by centrifugation at a minimum setting of 1,400 g for 20 minutes. The distribution ratio can then be calculated as

$$K_d = \frac{mass\,of\;\, solute\; on\, the\; solid\;\; phase\; per\; unit\; mass\, of\;\, solid\;\; phase}{mass\, of\;\, solute\; in\; solution\;\; per\; unit\; volume\, of\;\, the\; liquid\;\; phase}\,. \tag{2.13.2}$$

Column Method. Column experiments are used to simulate the migration of radionuclides through soils under saturated and/or unsaturated conditions. They allow observation of radionuclide migration rates without significant soil particle alteration caused by grinding, as in batch experiments, and produce more representative site-specific results. (Even removing a core sample to the laboratory results in alteration of the soil from its field condition.)

Typical equipment used in column experiments includes a reservoir to the column, a cylindrical holder to contain the crushed or intact soil being tested, and a sample collector for the column effluent. For experimentation on intact and fissured soil with low permeability, a high-pressure apparatus has to be used. The associated equipment costs, time constraints, experimental complications, and uncertainty in data reduction usually discourage potential users of the column system. Several operational problems with column experiments have been observed by numerous investigators: (1) homogeneity of column packing (Jackson et al. 1962; Hauth 1967; Ripple et al. 1974), (2) potential short-circuit effects (Danilk 1981; Klute and Dirksen 1986), and (3) residence time required for experimentation. To bridge the gap between batch and column experiments, a case study of cesium absorption on granite was done by Wang et al. (2009).

Theoretical models have been developed to describe solute transport in soil columns. Consider a situation in which water containing a dissolved tracer is introduced into a tracer-free soil column with a known dry density and volumetric water content. The hydrodynamic dispersion (i.e., the mechanical dispersion and molecular diffusion) of radionuclides throughout the column and the adsorption of radionuclides into the soil cause the initial sharp tracer front near the top end of the soil column to spread out downward. A mass balance equation for the radionuclide concentration in the liquid phase can be derived as follows:

$$R\frac{\partial C}{\partial t} = D\frac{\partial C}{\partial x^2} - \upsilon\frac{\partial C}{\partial x} , \qquad (2.13.3)$$

where

R = retardation factor,

D = coefficient of hydrodynamic dispersion,

v = average pore water velocity, and

C = radionuclide concentration in the water.

The retardation factor R is related to the distribution coefficient K_d of the radionuclide as follows:

$$R = 1 + \frac{\rho_b K_d}{\theta} \qquad (2.13.4)$$

where

ρ_b = dry soil density, and

θ = volumetric water content of the soil.

Therefore, K_d can be calculated if R is known. The solution to Equation (2.13.3) for a semi-infinite system is (Lapidus and Amundson 1952)

$$C(x,t) = \frac{C_o}{2}\left\{erfc\left[\frac{Rx - \upsilon t}{2(DRt)^{1/2}}\right] + \exp(\frac{\upsilon x}{D})erfc\left[\frac{Rx + \upsilon t}{2(DRt)^{1/2}}\right]\right\}, \qquad (2.13.5)$$

where C_o is the initial radionuclide concentration applied to the system. The relative effluent concentration, C', expressed in terms of two dimensionless parameters, the column Peclet number (P) and the number of pore volumes (T), is derived as follows:

$$C'(T) = \frac{1}{2}erfc\left[(\frac{P}{4RT})^{1/2}(R-T)\right] + \frac{1}{2}\exp(P)erfc\left[(\frac{P}{4RT})^{1/2}(R+T)\right], \qquad (2.13.6)$$

where

$$C' = C(L,t)/C_0 \ , \qquad (2.13.7)$$

$$T = \upsilon t/L \ , \qquad (2.13.8)$$

and

$$P = \upsilon L/D \ . \qquad (2.13.9)$$

The average interstitial or pore-water velocity is represented by v and is approximately equivalent to the ratio of the water flow rate to the volumetric water content. The length of the soil column is represented by L. The parameter L, in the case of field-measured concentration-time curves, simply refers to the soil depth at which the concentration was observed. The following expression is frequently used to describe displacement experiments (Danckwerts 1953; Rifai et al. 1956):

$$C(x,t) = \frac{C_0}{2}\,erfc\left[\frac{Rx - \upsilon t}{2(DRt)^{1/2}}\right] . \qquad (2.13.10)$$

This equation provides a close approximation to Equation (2.13.5) for relatively large values of P (>20). In terms of the Peclet number (P) and the number of pore volumes (T), when applied to the effluent concentration, Equation (2.13.10) can be written as follows:

$$C'(T) = \frac{1}{2}erfc\left[(\frac{P}{4RT})^{1/2}(R-T)\right] \qquad (2.13.11)$$

Many empirical methods based on the measured relative effluent concentration (C') versus the number of pore volumes (T) have been used for the analysis of P and R. These include the trial-and-error, slope, lognormal plot, and least-squares methods (Rifai et al. 1956; Van Genuchten and Wierenga 1986). The parameters P and R can also be calculated by using the method of moments (Aris 1958; Agneessens et al. 1978; Skopp 1985; Valocchi 1985; Jury and Sposito 1985) and methods for directly determining the coefficients K_d and D from the location and peak concentration of a short or instantaneous surface-applied tracer pulse (Kirkham and Powers 1972; Saxena et al. 1974; Yu et al. 1984). (Application of these methods is discussed in the original studies.)

2.13.2.2 Empirical Determination of the Distribution Coefficient

In addition to the experimental methods for determining the distribution coefficient (K_d), Baes and Sharp (1983), Baes et al. (1984), and Sheppard and Sheppard (1989) proposed an empirical approach to calculate K_d for radionuclide i from the soil-to-plant concentration ratio (B_{iv}), on the basis of the strong correlation between B_{iv} and K_d. Sheppard and Thibault (1990) proposed the following correlation equation:

$$\ln K_d = a + b(\ln B_{iv}) \ , \tag{2.13.12}$$

where a and b are constants. The value for the coefficient b is -0.5, on the basis of experimental data. The value of a depends on soil type: for sandy soil, $a = 2.11$; for loamy soil, $a = 3.36$; for clayey soil, $a = 3.78$; and for organic soil, $a = 4.62$. Equation (2.13.12) provides a method of estimating the distribution coefficient from the plant-soil concentration ratio, especially when experimental or literature data are not available. For actinides and transuranics, this approach may not be valid (IAEA 2010a).

2.13.3 Summary of Literature Review

In 2003, the International Atomic Energy Agency (IAEA) launched the program on Environmental Modeling for Radiation Safety (EMRAS), and one Working Group under this program worked on revising K_d values and transfer parameters for a large number of elements. Tables 2.13.1–2.13.5 list the K_d data developed for different soil types by the EMRAS Working Group (IAEA 2010a, Gil-Garcia et al. 2009a, Vandenhove et al. 2009, Gil-Garcia et al. 2009b). The data include geometric mean, geometric standard deviation, minimum and maximum values, and the mean and standard deviation of the underlying normal distribution. The data in these tables are from field and laboratory experiments with various contamination sources. The soils were grouped according to the sand and clay percentages and the organic matter content. For certain radionuclides, K_d values were also grouped by using the “cofactor” criterion. Table 2.13.6 lists the correlations between K_d and soil properties. Table 2.13.7 lists the K_d for some elements grouped on the basis of pH values. For some elements, the K_d values can also be grouped on the basis of other soil properties [water content (θ) and organic matter content (I) (Gil-Garcia et al. 2009b)]. Sheppard (2011) provided relationships for predicting the variations in

K_d values on the basis of soil properties (Table 2.13.8), such as soil pH, clay content, and organic carbon content. The data in the EMRAS compilation (IAEA 2010a; Gil-Garcia et al. 2009a, b; Vandenhove et al. 2009) was mainly based on short-term sorption studies, and the data of Sheppard (2011) represented desorption of indigenous elements. Table 2.13.9 lists the geometric mean K_d values for different soil types.

2.13.4 Data Input Requirements

The default distribution coefficients used in the RESRAD (onsite) and RESRAD-OFFSITE codes are listed in Table 2.13.10. From Tables 2.13.1–2.13.9, it can be seen that K_d is quite variant; that is, it assumes different values under different circumstances. Because K_d is one of the important input parameters that has a strong influence on the calculated results in the RESRAD (onsite) and RESRAD-OFFSITE codes, a site-specific value, if available, should always be used for risk assessment. In its decommissioning guidance (Schmidt et al. 2006), the U.S. Nuclear Regulatory Commission (NRC) encourages licensees to use sensitivity analysis to identify the importance of K_d on the resulting dose either (1) to demonstrate that a specific value used in the analysis is conservative or (2) to identify whether site-specific data should be obtained (if the licensee believes K_d is overly conservative).

In addition to the direct input of K_d values from the screen, the RESRAD (onsite) code provides four optional methods for deriving the distribution coefficient. The first method requires inputting a greater-than-zero value for the elapsed time since material placement (TI) and provision of the groundwater concentration of the radionuclide, which is measured at the same time as the radionuclide soil concentration. The second method uses the non-zero input leach rate (default is 0) to derive K_d. The third method is based on the correlation between the plant-soil concentration ratio and the water-soil distribution coefficient, which can be invoked by setting the K_d value to -1 on the input screen. The last method uses a solubility limit to derive an effective distribution coefficient. Only one of the four methods can be used in each RESRAD (onsite) execution. If more than one of the requirements is satisfied, RESRAD (onsite) will always choose according to the following order: the solubility limit method first, the groundwater concentration method second, the leach rate method third, and the plant/soil concentration ratio method last.

2.14 LEACH RATE

2.14.1 Definition

The leach rate is the fraction of the available radionuclide leached out from the contaminated zone per unit of time. It is assumed that the leaching process is driven by equilibrium distribution of the contaminant between the soil matrix and soil water. The leach rate is used in RESRAD (onsite) for calculating the source factor for adjusting radionuclide concentrations in the contaminated zone.

TABLE 2.13.1 Kd Data for Each Element for Sand Soil Type

Element	N	GM	GSD	Min	Max	μ	σ
Ac	1	450	NA[a]	NA	NA	NA	NA
Ag	3	130	5	36	695	4.87	1.61
Am	17	1,000	7	67	37,000	6.91	1.95
As	4	210	5	25	1,350	5.35	1.61
Be	1	240	NA	NA	NA	NA	NA
Bi	2	NA	NA	120	490	NA	NA
Br	1	15	NA	NA	NA	NA	NA
Ca[b]	7	3	4	0.7	28	1.1	1.39
Cd	30	110	8	2	1,770	4.7	2.08
Ce	3	400	1	316	490	5.99	0
Cl	3	0.5	4	0.1	1.1	−0.69	1.39
Cm	5	3,400	14	186	30,920	8.13	2.64
Co	18	260	18	5	36,756	5.56	2.89
Cr	9	8	8	1	100	2.08	2.08
Cs[b]	114	530	6	10	35,210	6.27	1.79
Cu	2	NA	NA	128	333	NA	NA
Dy	1	820	NA	NA	NA	NA	NA
Fe	4	320	1	220	424	5.77	0
Ga	1	310	NA	NA	NA	NA	NA
H	1	0.1	NA	NA	NA	NA	NA
Hf	2	NA	NA	450	3,270	NA	NA
Ho	1	240	NA	NA	NA	NA	NA
I	37	4	8	0.01	134	1.39	2.08
I-all	48	4	7	0.01	134	1.39	1.95
In	1	240	NA	NA	NA	NA	NA
IO_3	6	4	5	0.4	41	1.39	1.61
K[b]	60	3	3	0.7	179	1.1	1.1
La	1	5,300	NA	NA	NA	NA	NA
Lu	1	5,100	NA	NA	NA	NA	NA
Mg[b]	6	1	4	0.4	16	0	1.39
Mn	13	980	14	40	79,044	6.89	2.64
Mo	2	NA	NA	7	82	NA	NA
Na	6	2	4	0.4	23	0.69	1.39
Nb	2	NA	NA	160	187	NA	NA
Ni	26	130	10	3	7250	4.87	2.3
Np	8	14	4	3	108	2.64	1.39
P	2	NA	NA	9	760	NA	NA
Pa	1	540	NA	NA	NA	NA	NA
Pb[c]	9	220	4	25	1,349	5.39	1.39
Pd	2	NA	NA	55	127	NA	NA
Pm	1	450	NA	NA	NA	NA	NA
Po[c]	14	100	6	17	7,020	4.61	1.79
Pu	11	400	4	33	6,865	5.99	1.39
Ra[c]	20	3,100	8	49	40,000	8.04	2.08
Rb	1	55	NA	NA	NA	NA	NA
Ru	3	36	6	5	172	3.58	1.79
Sb	19	17	6	0.6	472	2.83	1.79

TABLE 2.13.1 (Cont.)

Element	N	GM	GSD	Min	Max	μ	σ
Sc	1	670	NA	NA	NA	NA	NA
Se	15	56	5	4	1616	4.03	1.61
Si	1	33	NA	NA	NA	NA	NA
Sm	1	240	NA	NA	NA	NA	NA
Sn	2	NA	NA	130	169	NA	NA
Sr[b]	65	22	6	0.4	2,424	3.09	1.79
Ta	2	NA	NA	240	379	NA	NA
Tb	1	5,400	NA	NA	NA	NA	NA
Tc	5	0.04	3	0.01	0.1	−3.22	1.1
Te	1	180	NA	NA	NA	NA	NA
Th[c]	12	700	11	35	100,000	6.55	2.4
Tm	1	330	NA	NA	NA	NA	NA
U[c]	50	110	12	0.7	66,667	4.7	2.48
V	1	180	NA	NA	NA	NA	NA
Y	5	22	2	10	47	3.09	0.69
Zn	17	110	23	0.9	27,815	4.7	3.14
Zr	4	32	16	2	600	3.47	2.77

Note: N = number of observations; GM = geometric mean; GSD = geometric standard deviation; Min = minimum; Max = maximum; μ = mean of the underlying normal distribution; and σ = standard deviation of the underlying normal distribution. For “sand” soil type, sand fraction ≥ 65% and clay fraction <18%.

[a] NA = not applicable.

[b] Source: Gil-Garcia et al. (2009a).

[c] Source: Vandenhove et al. (2009).

Source: Gil-Garcia et al. (2009b), except as noted.

TABLE 2.13.2 K_d Data for Each Element for Loam Soil Type

Element	N	GM	GSD	Min	Max	μ	σ
Ac	1	1,500	NA[a]	NA	NA	NA	NA
Ag	1	120	NA	NA	NA	NA	NA
Am	31	4,200	6	50	48,309	8.34	1.79
As	1	1,000	NA	NA	NA	NA	NA
Ba	1	0.4	NA	NA	NA	NA	NA
Be	1	810	NA	NA	NA	NA	NA
Bi	1	400	NA	NA	NA	NA	NA
Br	1	49	NA	NA	NA	NA	NA
Ca[b]	21	8	3	2	89	2.08	1.1
Cd	5	100	7	9	1,700	4.61	1.95
Ce	4	3,000	3	652	8,100	8.01	1.1
Cl	10	0.4	3	0.04	0.9	−0.92	1.1
Cm	9	19,000	2	6809	51,900	9.85	0.69
Co	71	810	15	2	103,595	6.7	2.71
Cr	9	45	23	1	1585	3.81	3.14
Cs[b]	191	3,500	4	39	55,100	8.16	1.39
Cu	1	490	NA	NA	NA	NA	NA
Fe	12	890	2	291	2,231	6.79	0.69
Hf	1	1,500	NA	NA	NA	NA	NA
Ho	1	810	NA	NA	NA	NA	NA
I	74	7	5	0.2	531	1.95	1.61
I-all	129	8	4	0.2	538	2.08	1.39
IO3	41	9	4	1	538	2.2	1.39
K[b]	81	20	4	2	911	3	1.39
Mg[b]	20	5	3	0.9	45	1.61	1.1
Mn	56	1,100	8	60	77,079	7	2.08
Mo	1	130	NA	NA	NA	NA	NA
Na	20	5	2	0.3	26	1.61	0.69
Nb	5	2,500	3	540	8370	7.82	1.1
Ni	14	180	5	8	1163	5.19	1.61
Np	12	23	4	1.3	117	3.14	1.39
P	2	NA	NA	30	380	NA	NA
Pa	1	1,800	NA	NA	NA	NA	NA
Pb[c]	5	10,000	3	3,600	43,000	9.21	1.1
Pd	1	180	NA	NA	NA	NA	NA
Po[c]	27	230	4	12	1,830	5.44	1.39
Pu	27	950	4	100	9,610	6.86	1.39
Ra[c]	19	1,100	17	12	120,000	7	2.83
Ra[c]	17	710	14	12	80,000	6.57	2.64
Rb	1	180	NA	NA	NA	NA	NA
Ru	3	300	3	82	990	5.7	1.1
Sb	92	61	3	4	2,065	4.11	1.1
Se	101	220	3	12	1,606	5.39	1.1
Si	1	110	NA	NA	NA	NA	NA
Sm	1	810	NA	NA	NA	NA	NA
Sn	1	450	NA	NA	NA	NA	NA
Sr[b]	120	57	5	2	2,549	4.04	1.61
Ta	1	810	NA	NA	NA	NA	NA

TABLE 2.13.2

Element	N	GM	GSD	Min	Max	μ	σ
Tc	14	0.07	3	0.01	0.1	−2.66	1.1
Th[c]	6	18,000	4	5,000	250,000	9.8	1.39
U[c]	84	310	12	0.9	38,710	5.74	2.48
Zn	48	2,400	4	211	153,070	7.78	1.39
Zr	2	NA	NA	2,200	8,100	NA	NA

Note: N = number of observation; GM = geometric mean; GSD = geometric standard deviation; Min = minimum; Max = maximum; μ = mean of the underlying normal distribution; and σ = standard deviation of the underlying normal distribution. When soil is not of sand, clay, or organic soil type, it is classified as loam soil type.

[a] NA = not applicable.

[b] Source: Gil-Garcia et al. (2009a).

[c] Source: Vandenhove et al. (2009).

Source: Gil-Garcia et al. (2009b), except as noted.

Reported leaching mechanisms have included diffusion, dissolution, ion exchange, corrosion and surface effects. Diffusion has traditionally been considered to be the most important leaching mechanism. However, it has been indicated that dissolution is also important for waste containing soluble salts and that ion exchange is important when sorbents such as zeolites or clay are included in the waste form.

A literature survey on leaching mechanisms by Colombo et al. (1985) indicates that factors that affect leaching have been divided into three categories: (1) system factors, (2) leachant factors, and (3) composition of contamination site form. System factors include time, temperature, pressure, radiation environment and ratio of waste form surface area to leachant volume. Leach rate is a function of time and the functional dependence typically changes over the long term. Therefore, changes in leach rate should be taken into account when long-term leach rate is used. Temperature is generally the first parameter to be varied in attempts to analyze rate processes. For example, the leachability of cement increases with temperature (Colombo et al. 1985).

Leachant factors include the effects of pH, oxidation potential (Eh), flow rate or replacement frequency, and composition. The solubility of most cations is strongly dependent on pH. The high pH limits the solubility of most radionuclides with the notable exception of cesium. Eh controls the oxidation state, and thus the solubility of elements such as cobalt with multiple oxidation states. Ames and Rai (1978) reported the Eh-pH diagrams predicting the presence of different solids for elements that exist in more than one oxidation state. Appendix J of User's

TABLE 2.13.3 K_d Data for Each Element for Clay Soil Type

Element	N	GM	GSD	Min	Max	μ	σ
Ac	1	2,400	NA[a]	NA	NA	NA	NA
Ag	1	180	NA	NA	NA	NA	NA
Am	1	8,100	NA	NA	NA	NA	NA
Be	1	1,300	NA	NA	NA	NA	NA
Bi	1	670	NA	NA	NA	NA	NA
Br	1	74	NA	NA	NA	NA	NA
Ca[b]	5	16	3	6	49	2.77	1.1
Cd	4	130	15	7	2721	4.87	2.71
Ce	3	910	15	122	20,000	6.81	2.71
Cl	5	0.2	3	0.06	0.9	−1.61	1.1
Cm	1	5,400	NA	NA	NA	NA	NA
Co	10	3,800	6	540	99,411	8.24	1.79
Cr	5	14	20	1	1,500	2.64	3
Cs[b]	36	5,500	4	566	375,000	8.61	1.39
Cu	2	NA	NA	101	2733	NA	NA
Fe	4	1,600	1	1,185	2,240	7.38	0
Hf	1	2,400	NA	NA	NA	NA	NA
Ho	1	1,300	NA	NA	NA	NA	NA
I	13	7	6	1	123	1.95	1.79
I-all	19	11	5	1	180	2.4	1.61
K[b]	12	43	3	9	294	3.76	1.1
Mg[b]	4	7	3	2	29	1.95	1.1
Mn	10	4,500	13	139	57,215	8.41	2.56
Mo	1	90	NA	NA	NA	NA	NA
Na	4	2	6	0.2	11	0.69	1.79
Nb	3	2,400	2	900	4,729	7.78	0.69
Ni	12	930	2	247	3,187	6.84	0.69
Np	2	NA	NA	20	55	NA	NA
P	1	49	NA	NA	NA	NA	NA
Pa	1	2,700	NA	NA	NA	NA	NA
Pb[c]	2	NA	NA	5,396	127,544	NA	NA
Pd	1	270	NA	NA	NA	NA	NA
Po[c]	1	732	NA	NA	NA	NA	NA
Pu	10	1,800	2	430	7,600	7.5	0.69
Ra[c]	6	38,000	12	696	950,000	10.55	2.48
Ra[c]	4	13,000	10	696	100,000	9.47	2.3
Rb	1	270	NA	NA	NA	NA	NA
Ru	4	500	2	203	989	6.21	0.69
Sb	18	140	2	38	614	4.94	0.69
Se	33	240	3	22	2,130	5.48	1.1
Si	1	180	NA	NA	NA	NA	NA
Sm	1	1,300	NA	NA	NA	NA	NA
Sn	1	670	NA	NA	NA	NA	NA
Sr[b]	19	95	4	9	747	4.55	1.39
Ta	1	1,300	NA	NA	NA	NA	NA
Tc	3	0.09	10	0.02	1	-2.41	2.3
Th[c]	7	4,500	3	800	24,000	8.41	1.1

TABLE 2.13.3 (Cont.)

Element	N	GM	GSD	Min	Max	μ	σ
U[c]	12	28	7	3	480	3.33	1.95
Zn	8	2,445	2	480	6,945	7.8	0.69
Zr	2	NA	NA	3300	10,300	NA	NA

Note: N = number of observation; GM = geometric mean; GSD = geometric standard deviation; Min = minimum; Max = maximum; μ = mean of the underlying normal distribution; and σ = standard deviation of the underlying normal distribution. For “clay” soil type, clay fraction ≥35%.

---

[a] NA = not applicable.

[b] Source: Gil-Garcia et al. (2009a).

[c] Source: Vandenhove et al. (2009).

Source: Gil-Garcia et al. (2009b), except as noted.

Manual for RESRAD Version 6 (Yu et al. 2001) describes estimating the effective distribution coefficient on the basis of radionuclide solubility when the stability-pH diagrams for radionuclides are provided. Leachant flow rate or replacement frequency affects the degree of saturation of the leachant with respect to leached material. Porosity in a solid is a major factor affecting diffusion within the solid. Changes in porosity due to dissolution of soluble material or other factors may affect long-term leachability.

TABLE 2.13.4 K_d Data for Each Element for Organic Soil Type

Element	n	GM	GSD	Min	Max	μ	σ
Ac	1	5,400	NA[a]	NA	NA	NA	NA
Ag	2	NA	NA	4,400	15,000	NA	NA
Am	13	2,500	5	210	110,000	7.82	1.61
Be	1	3,000	NA	NA	NA	NA	NA
Bi	1	1,500	NA	NA	NA	NA	NA
Br	1	180	NA	NA	NA	NA	NA
Ca[b]	1	110	NA	NA	NA	NA	NA
Cd	13	650	6	10	7,000	6.48	1.79
Ce	1	3,000	NA	NA	NA	NA	NA
Cl	2	NA	NA	0.1	1.2	NA	NA
Cm	3	7,400	2	5,056	12,000	8.91	0.69
Co	17	87	9	4	5,800	4.47	2.2
Cr	6	160	10	8	2,905	5.08	2.3
Cs[b]	108	270	7	4	95,000	5.6	1.95
Cu	4	320	3	76	883	5.77	1.1
Fe	3	1,400	3	521	4,900	7.24	1.1
Hf	1	5,400	NA	NA	NA	NA	NA

TABLE 2.13.4 (Cont.)

Element	n	GM	GSD	Min	Max	μ	σ
Ho	1	3,000	NA	NA	NA	NA	NA
I	9	36	4	8	581	3.58	1.39
I-all	11	32	3	8	581	3.47	1.1
IO3	1	13	NA	NA	NA	NA	NA
K[b]	76	19	3	2	134	2.94	1.1
Mn	3	160	4	36	490	5.08	1.39
Mo	2	NA	NA	18	27	NA	NA
Nb	1	2,000	NA	NA	NA	NA	NA
Ni	8	1,100	2	406	4,990	7	0.69
Np	4	810	1	500	1,200	6.7	0
P	1	110	NA	NA	NA	NA	NA
Pa	1	6,600	NA	NA	NA	NA	NA
Pb[c]	5	2500	3	880	10,266	7.82	1.1
Pd	1	670	NA	NA	NA	NA	NA
Pu	6	760	4	90	2951	6.63	1.39
Ra[c]	1	200	NA	NA	NA	NA	NA
Rb	1	670	NA	NA	NA	NA	NA
Ru	1	66,000	NA	NA	NA	NA	NA
Sb	3	75	8	8	540	4.32	2.08
Se	2	NA	NA	230	1,800	NA	NA
Si	1	400	NA	NA	NA	NA	NA
Sm	1	3,000	NA	NA	NA	NA	NA
Sn	1	1,600	NA	NA	NA	NA	NA
Sr[b]	37	110	6	3	6,500	4.7	1.79
Ta	1	3,000	NA	NA	NA	NA	NA
Tc	11	3	3	0.9	11	1.1	1.1
Th[c]	5	730	44	19	80,000	6.59	3.78
U[c]	9	1,200	6	33	7,600	7.09	1.79
Y	2	NA	NA	260	375	NA	NA
Zn	12	570	8	10	7,630	6.35	2.08
Zr	2	NA	NA	23	7,300	NA	NA

Note: N = number of observations; GM = geometric mean; GSD = geometric standard deviation; Min = minimum; Max = maximum; μ = mean of the underlying normal distribution; and σ = standard deviation of the underlying normal distribution. For "organic" soil type, organic fraction >20%.

[a] NA = not applicable.

[b] Source: Gil-Garcia et al. (2009a).

[c] Source: Vandenhove et al. (2009).

Source: Gil-Garcia et al. (2009b), except as noted.

TABLE 2.13.5 K_d Data for Each Element for Generic Soil Type

Element	N	GM	GSD	Min	Max	μ	σ
Ac	4	1,700	3	450	5,400	7.44	1.10
Ag	9	380	7	36	15,000	5.94	1.95
Am	62	2,600	6	50	110,000	7.86	1.79
As	7	550	5	25	2,991	6.31	1.61
Be	5	990	3	240	3,000	6.9	1.10
Bi	6	480	2	120	1,500	6.17	0.69
Br	4	56	3	15	180	4.03	1.10
Ca[b]	34	8	3	0.7	110	2.08	1.10
Cd	61	150	9	2	7,000	5.01	2.20
Ce	11	1,200	5	122	20,000	7.09	1.61
Cl	22	0.3	3	0.04	1.2	-1.2	1.10
Cm	18	9,300	4	186	51,900	9.14	1.39
Co	118	480	16	2	103,595	6.17	2.77
Cr	31	40	20	1	7,943	3.69	3.00
Cs[b]	469	1,200	7	4	375,000	7.09	1.95
Cu	11	530	3	76	2,733	6.27	1.10
Dy	2	NA[a]	NA	820	2,100	NA	NA
Fe	23	880	2	220	4,900	6.78	0.69
Ga	2	NA	NA	280	310	NA	NA
Hf	6	2,500	3	450	8,500	7.82	1.10
Ho	4	930	3	240	3,000	6.84	1.10
I	157	5	6	0.01	581	1.61	1.79
I-all	250	7	5	0.01	581	1.95	1.61
In	2	NA	NA	240	730	NA	NA
IO_3	67	8	4	0.4	538	2.08	1.39
K[b]	237	13	4	0.7	911	2.56	1.39
Mg[b]	30	4	3	0.4	45	1.39	1.10
Mn	83	1,200	9	36	79,044	7.09	2.20
Mo	9	38	3	7	130	3.64	1.10
Na	30	3	3	0.2	26	1.1	1.10
Nb	11	1,500	4	160	8,370	7.31	1.39
Ni	64	280	7	3	7,250	5.63	1.95
Np	26	36	6	1.3	1200	3.58	1.79
P	6	87	5	9	760	4.47	1.61
Pa	4	2,000	3	540	6,600	7.6	1.10
Pb[c]	23	2,100	10	25	127,544	7.65	2.30
Pd	6	180	2	55	670	5.19	0.69
Pm	2	NA	NA	450	450	NA	NA
Po[c]	42	180	5	12	7,020	5.19	1.61
Pu	62	740	4	32	9,610	6.61	1.39
Ra[c]	51	2,500	13	12	950,000	7.82	2.56
Ra[c,d]	47	1,800	10	12	100,000	7.5	2.30
Rb	4	210	3	55	670	5.35	1.10
Ru	15	270	8	5	66,000	5.6	2.08
Sb	152	62	4	0.6	2065	4.13	1.39
Sc	2	NA	NA	670	3,500	NA	NA
Se	172	200	3	4	2,130	5.3	1.10

TABLE 2.13.5 (Cont.)

Element	N	GM	GSD	Min	Max	μ	σ
Si	4	130	3	33	400	4.87	1.10
Sm	4	930	3	240	3,000	6.84	1.10
Sn	12	1,600	6	130	31,000	7.38	1.79
Sr[b]	255	52	6	0.4	6500	3.95	1.79
Ta	5	780	3	240	3000	6.66	1.10
Tb	2	NA	NA	5,400	6600	NA	NA
Tc	33	0.2	9	0.01	11	-1.61	2.20
Te	2	NA	NA	180	790	NA	NA
Th[c]	46	1,900	10	19	250,000	7.55	2.30
U[c]	178	200	12	0.7	66,667	5.3	2.48
V	2	NA	NA	180	410	NA	NA
Y	7	47	4	10	375	3.85	1.39
Zn	92	950	11	0.9	153,070	6.86	2.40
Zr	11	410	21	2	10,300	6.02	3.04

Note: N = number of observations; GM = geometric mean; GSD = geometric standard deviation; Min = minimum; Max = maximum; μ = mean of the underlying normal distribution; and σ = standard deviation of the underlying normal distribution. For generic soil type, the data includes all soil types combined including sand, loam, clay, organic, and the “unspecified” soil type reported in the original reference sources.

[a] NA = not applicable.

[b] Source: Gil-Garcia et al. (2009a).

[c] Source: Vandenhove et al. (2009).

[d] Values excluding one dataset.

Source: Gil-Garcia et al. (2009b), except as noted.

TABLE 2.13.6 Correlations between K_d and Soil Main Properties for Selected Elements

Element – Soil Type	Regression Equations	Number of Observations	Correlation Coefficient	% Variance Explained
Cd – All soils	Log $K_d = 0.8(0.4) + 0.21(0.07) \times \mathrm{pH}$	55	0.38	13
	Log $K_d = -0.1(0.5) + 0.34(0.08) \times \mathrm{pH} + 0.4(0.1) \times \log(\mathrm{OM})$	54	0.49	24
Cd – Mineral soils	Log $K_d = -0.7(0.4) + 0.41(0.06) \times \mathrm{pH}$	43	0.71	49
Co – All soils	Log $K_d = -0.7(0.3) + 0.63(0.05) \times \mathrm{pH}$	113	0.75	56
	Log $K_d = -1.5(0.4) + 0.74(0.06) \times \mathrm{pH} + 0.5(0.2) \times \log(\mathrm{OM})$	110	0.77	59

TABLE 2.13.6 (Cont.)

Element – Soil Type	Regression Equations	Number of Observations	Correlation Coefficient	% Variance Explained
Co – Mineral soils	Log K_d = −1.2(0.4) + 0.71(0.06) × pH	97	0.76	58
Cr(VI) – All soils	Log K_d = 4.7(0.6)–0.52(0.08) × pH	12	-0.89	78
Cs – All soils	Log K_d = 0.94(0.04) × log(RIP/K_{ss})	257	0.78	65
Cu – Mineral soils	Log K_d = −3(1) + 0.8(0.1) × pH	5	0.95	88
I – All soils	Log K_d = 0.63(0.04) + 0.6(0.1) × log(OM)	227	0.55	30
	Log K_d = −1.4(0.4) + 0.6(0.1) × log(Fe)	124	0.44	18
	Log K_d = −0.6(0.4) + 0.7(0.1) × log(OM) + 0.3(0.1) × log(Fe)	124	0.63	39
Ni – All soils	Log K_d = 0.1(0.3) + 0.34(0.05) × pH	58	0.68	46
	Log K_d = −1.6(0.5) + 0.55(0.06) × pH + 0.27(0.09) × log(clay)	38	0.82	67
	Log K_d = −0.7(0.3) + 0.41(0.04) × pH + 0.7(0.1) × log(OM)	58	0.84	70
Ni – Mineral soils	Log K_d = −0.6(0.3) + 0.43(0.04) × pH	51	0.82	66
	Log K_d = −1.6(0.5) + 0.55(0.06) × pH + 0.27(0.09) × log(clay)	38	0.82	67
	Log K_d = -0.9(0.3) + 0.45(0.04) × pH + 0.6(0.1) × log(OM)	51	0.86	74
Pb – All soils[a]	Log K_d = 1.25(0.45) + 0.37(0.08) × pH	21	0.52	68
Sr – All soils	Log K_d = −0.05(0.09) + 0.86(0.03) × log(CEC/(Ca + Mg)$_{ss}$)	96	0.95	90
U – Soil adequate for agriculture[a]	Log K_d = −0.77(0.11) × pH + 7.7 (0.7)	110	0.3	20
Zn – All soils	Log K_d = −0.1(0.5) + 0.52(0.08) × pH	88	0.55	30
	Log K_d = −1.0(0.6) + 0.6(0.1) × pH + 0.5(0.2) × log(OM)	86	0.59	35
Zn – Mineral soils	Log K_d = −1.2(0.5) + 0.71(0.09) × pH	75	0.69	47
	Log K_d = −1.8(0.6) + 0.8(0.9) × pH + 0.5(0.2) × log(OM)	73	0.71	50

Note: Values in brackets show the uncertainty in the number; OM = organic matter content, RIP = radiocesium interception potential (mmol/kg), K_{ss} = concentration of K in soil solution (cmol$_c$/L), CEC = cation exchange capacity (cmol$_c$/kg), (Ca + Mg)$_{ss}$ = concentration of Ca and Mg in soil solution (cmol$_c$/L).

[a] Source: Vandenhove et al. (2009).

Source: Gil-Garcia et al. (2009b), except as noted.

TABLE 2.13.7 K_d Values Grouped According to pH Values

Element	Soil Group	Number of Observations	Geometric Mean	Geometric Standard Deviation	Minimum	Maximum
Cd	pH<5	8	11	3	2	64
	5≤pH<6.5	11	18	4	6	250
	pH≥6.5	24	380	6	4	4,360
Co	pH<5	21	12	5	2	153
	5≤pH<6.5	50	1,100	5	29	99,941
	pH≥6.5	26	4,600	4	547	103,595
Ni	pH<5	10	15	2	3	48
	5≤pH<6.5	11	58	4	7	1,100
	pH≥6.5	30	820	4	40	7,250
Pb[a]	3≤pH≤6.4	13	570	6	25	6,200
	6.4<pH≤8.3	8	7,900	7	301	127,544
Th[a]	pH<5	11	1,275	15	19	10,200
	5≤pH<8	26	3,261	8	100	100,000
	pH≥8	6	310	7	35	3,200
U[a]	pH<5	36	71	11	0.7	6,700
	5≤pH<7	77	740	8	2.6	66,667
	pH≥7	61	68	8	0.9	6,160
Zn	pH<5	9	8	8	1	301
	5≤pH<6.5	49	1,600	6	6	30,157
	pH≥6.5	17	4,300	4	437	153,070

[a] Source: Vandenhove et al. (2009).

Source: Gil-Garcia et al. (2009b), except as noted.

TABLE 2.13.8 Regression Equations for K_d Values for Some Nuclides

Element	Regression Equation	Number of Soils	Geometric Standard Deviation	Soil Organic Carbon (%), 5th/95th Percentiles
As	$\text{Log}(K_d) = 2.39 + 0.085 \times \text{pH}$	178	1.8	1.1/4.7
Cd	$\text{Log}(K_d) = 2.35 + 0.114 \times \text{pH}$	150	1.4	0/5.0
Ce	$\text{Log}(K_d) = 1.84 + 0.469 \times \text{pH} - 0.00162 \times \text{clay} \times \text{pH}$	209	2.0	0/4.7
Cl	1.4 L/kg for mineral soils and 150 L/kg for organic soils	11/3	NA[a]	NA
Co	$\text{Log}(K_d) = 1.46 + 0.247 \times \text{pH} + 0.00709 \times \text{clay}$	342	7.5	0.27/30

TABLE 2.13.8 (Cont.)

Element	Regression Equation	Number of Soils	Geometric Standard Deviation	Soil Organic Carbon (%), 5th/95th Percentiles
Cr	$\text{Log}(K_d) = 1.61 + 0.29 \times \text{pH} + 0.381 \times \log(\text{organic carbon})$ – CrIII; 9.4 L/kg – CrVI	83; 51	3.0; 2.0	0.29/30; 0.06/8.0
Cs[b]	$\text{Log}(K_d) = 3.03 + 0.101 \times \text{pH} + 0.0117 \times \text{clay}$	470	5.6	0.09/40
Cu	$\text{Log}(K_d) = 2.47 + 0.0656 \times \text{pH} + 0.00726 \times \text{clay}$	205	1.4	0/4.7
Fe	$\text{Log}(K_d) = 2.01 + 0.00442 \times \text{clay} \times \text{pH}$	44	3.2	0.81/31
Ho	$\text{Log}(K_d) = 2.15 + 0.338 \times \text{pH} - 0.00094 \times \text{clay} \times \text{pH}$	161	1.7	0.0/4.8
I	$\text{Log}(K_d) = 0.953 + 0.701 \times \log(\text{organic carbon})$	114	8.1	0.19/49
La	$\text{Log}(K_d) = 3.26 + 0.234 \times \text{pH} - 0.0448 \times \text{clay} + 0.00517 \times \text{clay} \times \text{pH}$	227	1.8	0.0/4.7
Mn[c]	$\text{Log}(K_d) = -0.330 + 0.457 \times \text{pH}$	402	15	0.4/7.7
Mo	$\text{Log}(K_d) = 3.22 - 0.212 \times \text{pH} + 0.0125 \times \text{clay}$	215	1.9	0.44/4.8
Nb	$\text{Log}(K_d) = 2.45 + 0.348 \times \text{pH} + 0.0960 \times \text{clay} - 0.0159 \times \text{clay} \times \text{pH}$	92	2.8	0.0/8.6
Nd	$\text{Log}(K_d) = 2.98 + 0.271 \times \text{pH} - 0.0112 \times \text{clay} + 0.204 \times \log(\text{organic carbon})$	228	1.9	3.5/5.5
Ni[d]	$\text{Log}(K_d) = 0.816 + 0.229 \times \text{pH}$	410	2.5	0.3/14
Np	$\text{Log}(K_d) = -1.71 + 0.332 \times \text{pH} + 0.960 \times \log(\text{organic carbon}) + 0.00740 \times \text{clay} \times \text{pH}$	159	5.7	0.42/40
Pa	1380 L/kg for mineral soils and 6600 L/kg for organic soils	4	NA	NA
Pb	$\text{Log}(K_d) = 1.96 + 0.276 \times \text{pH} + 0.294 \times \log(\text{organic carbon})$	362	6.4	0.3/9.8
Pu	$\text{Log}(K_d) = 1.77 + 0.193 \times \text{pH} + 0.637 \times \log(\text{organic carbon})$	175	3.9	0.1/4.9
Ra	$\text{Log}(K_d) = -2.64 + 0.676 \times \text{pH}$	38	30	0.01/9.9
Sb	$\text{Log}(K_d) = 3.24 - 0.107 \times \text{pH} + 0.00614 \times \text{clay}$	197	1.4	0.0/4.7
Se	$\text{Log}(K_d) = 2.02 + 0.0929 \times \text{pH} - 0.00964 \times \text{clay}$	123	2.0	0.6/30

TABLE 2.13.8 (Cont.)

Element	Regression Equation	Number of Soils	Geometric Standard Deviation	Soil Organic Carbon (%), 5th/95th Percentiles
Sm	Log(K_d) = 2.77 + 0.273 × pH + 0.00852 × clay + 0.253 × log(organic carbon)	218	1.8	0.0/4.8
Sn	2100 L/kg	32	1.7	0.18/8.6
Sr	Log(K_d) = 2.93 – 0.224 × pH + 0.0217 × clay	481	4.9	0.09/14
Tc	2.1 L/kg if aerobic Log(K_d) = −0.0243 + 0.253 × pH + 0.531 × log(organic carbon) if anaerobic	118, 33	3.8, 1.9	0.06/59, 0.18/54
Th	Log(K_d) = 1.90 + 0.346 × pH	39	12	0.8/35
Tl	Log(K_d) = 4.08 – 0.0842 × pH + 0.0181 × clay	170	1.5	0/4.7
Tm	Log(K_d) = 1.94 + 0.369 × pH =0.00150 × clay × pH	87	2.1	0.0/8.6
U	Log(K_d) = 9.05–0.989 × pH + 0.00290 × clay × pH where pH ≥5.5 and Log(K_d) = 1.75 + 0.0145 × clay × pH where pH <5.5	318, 28	4.7, 11	0.9/50, 1.8/50
W	6020 L/kg	19	2.2	0.9/10
Yb	Log(K_d) = 2.71 + 0.244 × pH – 0.000962 × clay × pH	203	1.8	0.0/4.7

[a] Not applicable

[b] For desorption of indigenous Cs, the K_d values will be 4.2-fold higher than predicted by the equation, and for sorption of new Cs-137, they will be 4.2-fold lower.

[c] For desorption of indigenous Mn, the K_d values will be 4.0-fold higher than predicted by the equation, and for sorption of new Mn-54, they will be 4.0-fold lower.

[d] For desorption of indigenous Ni, the K_d values will be 6.2-fold higher than predicted by the equation, and for sorption of new Ni, they will be 6.2 fold lower.

TABLE 2.13.9 Summary of Geometric Mean K_d Values (cm^3/g) for Each Element by Soil Type

	Soil Type				
Element	Sand	Loam	Clay	Organic	Generic
Ac	450	1,500	2,400	5,400	1,700
Ag	130	120	180	NA[a]	380

TABLE 2.13.9 (Cont.)

	Soil Type				
Element	Sand	Loam	Clay	Organic	Generic
Am	1,000	4,200	8,100	2,500	2,600
As	210	1000	NA	NA	550
Ba	NA	0.4	NA	NA	NA
Be	240	810	1,300	3,000	990
Bi	NA	400	670	1,500	480
Br	15	49	74	180	56
Ca[b]	3	8	16	110	8
Cd	110	100	130	650	150
Ce	400	3,000	910	3,000	1,200
Cl	0.5	0.4	0.2	NA	0.3
Cm	3,400	19,000	5,400	7,400	9,300
Co	260	810	3,800	87	480
Cr	8	45	14	160	40
Cs[b]	530	3,500	5,500	270	1,200
Cu	NA	490	NA	320	530
Dy	820	NA	NA	NA	NA
Fe	320	890	1,600	1,400	880
Ga	310	NA	NA	NA	NA
H	0.1	NA	NA	NA	NA
Hf	NA	1,500	2,400	5,400	2,500
Hg	NA	NA	NA	NA	6,300
Ho	240	810	1,300	3,000	930
I-all	4	8	11	32	7
IO_3	4	9	NA	13	8
I^-	4	7	7	36	5
In	240	NA	NA	NA	730
Ir	NA	NA	NA	NA	3
K[b]	3	20	43	19	13
La	5,300	NA	NA	NA	NA
Lu	5,100	NA	NA	NA	NA
Mg[b]	1	5	7	NA	4
Mn	980	1,100	4,500	160	1,200
Mo	NA	130	90	NA	38
Na	2	5	2	NA	3
Nb	NA	2,500	2,400	2,000	1,500
Ni	130	180	930	1,100	280
Np	14	23	NA	810	36
P	NA	NA	49	110	87
Pa	540	1,800	2,700	6,600	2,000
Pb[c]	220	10,000		2,500	2,100
Pd	NA	180	270	670	180
Pm	450	NA	NA	NA	NA
Po[c]	100	230	732	NA	180
Pt	NA	NA	NA	NA	24
Pu	400	950	1800	760	740

TABLE 2.13.9 (Cont.)

	Soil Type				
Element	Sand	Loam	Clay	Organic	Generic
Ra[c]	3,100	1,100	38,000	200	2,500
Ra[c, d]	NA	710	13,000	NA	1,800
Rb	55	180	270	670	210
Rh	NA	NA	NA	NA	4
Ru	36	300	500	66,000	270
Sb	17	61	140	75	62
Sc	670	NA	NA	NA	NA
Se	56	220	240	NA	200
Si	33	110	180	400	130
Sm	240	810	1,300	3,000	930
Sn	NA	450	670	1,600	1,600
Sr[b]	22	57	95	110	52
Ta	NA	810	1,300	3,000	780
Tb	5,400	NA	NA	NA	NA
Tc	0.04	0.07	0.09	3	0.2
Te	180	NA	NA	NA	790
Th[c]	700	18,000	4,500	730	1,900
Tm	330	NA	NA	NA	NA
U[c]	110	310	28	1,200	200
V	180	NA	NA	NA	NA
Y	22	NA	NA	NA	47
Zn	110	2,400	2445	570	950
Zr	32	NA	NA	NA	410

[a] NA = not applicable.

[b] Source: Gil-Garcia et al. (2009a).

[c] Source: Vandenhove et al. (2009).

[d] Estimates exclude data with very low Ca^{2+} concentration in external solution.

Source: Gil-Garcia et al. (2009b), except as noted.

TABLE 2.13.10 RESRAD Default Value and Distribution for the K_d Parameter for Different Elements

Element	Kd (cm^3/g or L/kg)	Lognormal Distribution[a] μ	σ
Ac	20	6.72	3.22
Ag	0	5.38	2.1
Al	0	6.45	3.22
Am	20	7.28	3.15
As	114	NA[b]	NA
At	0	NA	NA
Au	0	4.65	3.22
Ba	50	6.33	3.22
Be	810	NA	NA
Bi	0	4.65	3.22
Bk	70	NA	NA
Br	49	NA	NA
C	0	2.4	3.22
Ca	50	1.4	0.78
Cd	0	3.52	2.99
Ce	1,000	7.6	2.08
Cf[c]	1,380	7.23	3.22
Cl	0.1	1.68	3.22
Cm[c]	1,380	8.82	1.82
Co	1,000	5.46	2.53
Cr	30	4.63	2.76
Cs	4,600	6.1	2.33
Cu	333	NA	NA
Dy[c]	935	NA	NA
Er[c]	935	NA	NA
Es[c]	1,380	NA	NA
Eu[c]	825	6.72	3.22
F[c]	257	NA	NA
Fe	1,000	5.34	2.67
Fm[c]	935	NA	NA
Fr	200	NA	NA
Ga[c]	745	NA	NA
Gd[c]	825	6.72	3.22
Ge	0	3.87	3.22
H	0	−2.81	0.5
Hf	1,500	NA	NA
Hg	52	NA	NA
Ho	800	NA	NA
I	0.1	1.52	2.19
In	158	NA	NA
Ir	0	5.32	3.22
K	5.5	1.7	0.49

TABLE 2.13.10 (Cont.)

Element	Kd (cm^3/g or L/kg)	Lognormal Distribution[a] μ	Lognormal Distribution[a] σ
La	4.98	NA	NA
Lu[c]	935	NA	NA
Md[c]	935	NA	NA
Mg	63	NA	NA
Mn	200	5.06	2.29
Mo	125	NA	NA
Na	10	5.04	3.22
Nb	0	5.94	3.22
Nd	158	NA	NA
Ni	1,000	6.05	1.46
Np[c]	257	2.84	2.25
Os	157	NA	NA
P	30	NA	NA
Pa	50	5.94	3.22
Pb	100	7.78	2.76
Pd	180	NA	NA
Pm[c]	825	6.72	3.22
Po	10	5.2	1.68
Pr	157	NA	NA
Pt	24	NA	NA
Pu	2,000	6.86	1.89
Ra	70	8.17	1.7
Rb	125	NA	NA
Re	43.5	NA	NA
Rh	4	NA	NA
Rn	0	NA	NA
Ru	0	7.37	3.13
S	0	3.65	3.22
Sb	0	5.94	3.22
Sc	0	6.84	3.22
Se	0	4.73	0.57
Si	110	NA	NA
Sm[c]	825	6.72	3.22
Sn	0	6.72	3.22
Sr	30	3.45	2.12
Ta	0	5.55	3.22
Tb	157	NA	NA
Tc	0	-0.67	3.16
Te	0	3.64	3.22
Th	60,000	8.68	3.62
Ti[c]	1,380	NA	NA
Tl	0	4.26	3.22
Tm[c]	935	NA	NA
U	50	4.84	3.13
V[c]	935	NA	NA

TABLE 2.13.10 (Cont.)

Element	Kd (cm^3/g or L/kg)	Lognormal Distribution[a] μ	Lognormal Distribution[a] σ
W	157	NA	NA
Y	720	NA	NA
Yb[c]	935	NA	NA
Zn	0	6.98	4.44
Zr	2,200	7.23	3.22

Note: μ = mean of the underlying normal distribution and σ = standard deviation of the underlying normal distribution.

[a] Source: Yu et al. (2000).

[b] NA = not applicable.

[c] RESRAD (onsite) uses plant/soil ratio to estimate K_d value.

2.14.2 Measurement Methodology

The leaching rate and concentration of radionuclides in the environment can be investigated by means of laboratory or field tests. Laboratory leaching tests involve mixing radionuclide samples with a liquid to determine which constituents will be leached by (or dissolved into) the liquid and potentially released to the environment in a liquid phase (e.g., to ground water or surface water). Laboratory experiments may include single extraction/batch tests or multiple extraction/flow-around/flow-through (“dynamic”) leaching tests. Field experiments generally involve the use of controlled pilot landfill cells or field lysimeters. Field lysimeter tests can be used to address different aspects of leaching, such as the physical mechanisms involved, chemical interactions between the waste and the leaching fluid, the kinetics of leaching, and leaching as a function of pH, time, and liquid-to-solid (L/S) ratio (Kim 2002; Washington State Department of Ecology 2003). A relative complete information on the commonly used leaching tests can be found from the a survey by the Washington State Department of Ecology (2003).

Laboratory tests fall into two general categories: (1) single extraction/batch tests (sometimes referred to as “static” extraction tests), and (2) multiple extraction/flow-around and flow-through leaching tests (sometimes referred to as “dynamic” tests). Single extraction tests include all tests in which a specific amount of leaching fluid is put into contact with a specific amount of waste for a specified length of time, without renewal of the leaching fluid. The resulting leachate is removed from the test, either at various times to derive kinetic information (changing concentrations over time) or, more commonly, at the end of the test and then analyzed. The assumption made in a single extraction test is that a steady-state condition (equilibrium) is achieved by the end of the testing period, although this may not necessarily be the case in practice. Reaching equilibrium in single extraction leach testing is critical to predicting leaching behavior over long periods of time. If tests are conducted at non-equilibrium conditions, leaching behavior does not reach capacity, and predictions of long-term leaching behavior will be based on leachate concentrations that are too low or too high.

Multiple extraction tests continuously or intermittently renew the leaching fluid to maintain a driving force for leaching. Multiple extraction tests provide information about the kinetics of contaminant mobilization. There are three primary types of multiple extraction tests: serial batch tests, flow-around tests, and flow-through tests. In a serial batch test, a portion of a granular sample is mixed with the leaching fluid and agitated at a set L/S ratio for a specified period of time. The leachate is then separated from the solids and replaced with a fresh leaching fluid until the desired number of leaching periods has been completed. Data from these tests can be used to infer temporal release of leachable constituents. In a flow-through test, the leaching fluid is passed, either intermittently or continuously, through an open container packed with a porous solid sample. The leachate is periodically sampled and analyzed for the parameters of interest. The results are used to examine contaminant release over time and as a function of L/S ratio. There are two typical types of flow-through tests, lysimeter tests and column tests, which differ primarily in size and duration. The commonly used leaching tests are summarized in Table 2.14.1 (Washington State Department of Ecology 2003)

TABLE 2.14.1 Commonly Used Leaching Tests (Source: Washington State Department of Ecology 2003)

Single Extraction/Batch Leaching Tests	• ASTM D 3987, Standard Test Method for Shake Extraction of Solid Waste with Water • ASTM D 6234, Standard Method for Shake Extraction of Mining Waste by the SPLP • SPLP, Synthetic Precipitation Leaching Procedure • TCLP, Toxicity Characteristic Leaching Procedure • DRET, Dredge Elutriate Test • SET, Standard Elutriate Test • NEN 7341, Availability Test • EN 12457/1-4, Compliance Test for Granular Waste Materials and Sludges
Multiple Extraction/Flow-around and Flow-through Leaching Tests	Sequential Batch Tests • ASTM D 4793, Standard Test Method for Sequential Batch Extraction of Waste with Water • ASTM D 5744, Standard Test Method for Accelerated Weathering of Solid Materials Using a Modified Humidity Cell • SBLT, Sequential Batch Leachate Test Flow-around Test • NEN 7345, Tank Leach Test Flow-through Tests

- ASTM D 4874, Standard Test Method for Leaching Solid Waste in a Column Apparatus
- PCLT, Pancake Column Leach Test
- NEN 7343, Column Test
- prEN 14405, Upflow percolation test

2.14.3 Data Input Requirements

In RESRAD (onsite), the leach rate is entered in units of one over time (T^{-1}). An input value of 0 (default value) for the leach rate will invoke the calculation of this parameter via a first-order leaching model that uses the value of the soil/water distribution coefficient in the contaminated zone. If the input value of this parameter is greater than 0, however, it will be used to derive the soil/water distribution coefficient of the contaminated zone on the basis of the same first-order leaching model. The input soil/water distribution coefficients are then replaced by the derived value.

Because the leach rate constant and the soil-water distribution coefficients are two of the most critical parameters affecting the calculated results of water-related pathways, site-specific values should always be used when available. The default leach rate constant in RESRAD (onsite) is zero. Note that, if a non-zero leach rate is input by the user, RESRAD (onsite) can use the non-zero input leach rate from the user to back-calculate the corresponding distribution coefficient K_d (Yu et al. 2001). In RESRAD-OFFSITE code, users must enter distribution coefficients for the transport and accumulation zones even if they enter a non-zero value for the leach rate. RESRAD-OFFSITE does not perform transport in the contaminated zone for the Version 2 release option. Therefore, it does not need to back-calculate the K_d; it simply uses the user input leach rate.

The first-order ion-exchange leaching model used in RESRAD (onsite) that estimates the leach rate from the distribution coefficient and other site-specific parameters is in general a conservative approach for estimating the leaching of radionuclides (Base and Sharp 1983, Yu 1987). When no leach rate data are available, the input of a site-specific K_d value is sufficient.

In the RESRAD-OFFSITE code Version 3, a new source term model allows users to input both initial and final leach rates (or to use a distribution coefficient with the instantaneous equilibrium desorption release model). The source release model implemented in RESRAD-OFFSITE Version 2 (i.e., the first-order, rate-controlled leaching model, also referred to as the exponential leaching model, which is basically the same as the one used in RESRAD [onsite] code), still remains in the RESRAD-OFFSITE Version 3. This new source term model includes (1) the “first order release with transport” option, in which the release of the radionuclide is proportional to the inventory in the primary contamination and the user-specified leach rate is the proportionality constant, and (2) the "“equilibrium desorption release” option, in which the user specifies the distribution coefficient, which quantifies the partitioning of the radionuclide between the solid and aqueous phases. Compared with the source term model in RESRAD-OFFSITE Version 2, this new source term model allows the RESRAD-OFFSITE code to simulate waste leaching from containers or disposal cells with a linear or stepwise change in release rates over time. This new source term model is described in detail in Appendix D of NUREG/CR-7127 (Yu et al. 2013). It considers three releases: (1) the atmospheric release of particulates due to resuspension and of volatiles due to diffusion and evapotranspiration, (2) the surface water release due to erosion by runoff water, and (3) the groundwater release due to leaching by infiltrating water and distribution in the primary contaminated zone. The new source term model considers not only the parent radionuclide but also all radionuclides in the decay chain. In addition, the new source term model allows users to perform both sensitivity and probabilistic analyses on the input parameters of the model.

2.15 VOLUMETRIC WATER CONTENT

2.15.1 Definition

The water content in soils is usually expressed as a dimensionless ratio of either two masses or two volumes, or is given as a ratio of mass per unit volume. These dimensionless ratios can be reported either as decimal fractions or percentages, if multiplied by 100. To avoid confusion between the two dimensionless water content ratios, their basis (i.e., mass or volume) should always be stated. However, in cases in which no indication is given, the figure is assumed to be based on mass because in the determination of the soil water content, the mass-basis figure is usually obtained first and then converted to a volume-basis figure (Gardner 1986). In the RESRAD (onsite) and RESRAD-OFFSITE codes, the input data related to the water content in soil materials are entered on a volume basis (volumetric water content).

The water content in soils on a mass basis, w, is defined as the ratio of the mass of the liquid phase (water), M_l, in the given soil sample to the mass of the solid material, M_s, according to the following expression:

$$w = \frac{M_l}{M_s} \quad . \tag{2.15.1}$$

The volumetric water content, θ, in the soil (also called the volume wetness or volume fraction of soil water) represents the fraction of the total volume of soil that is occupied by the water contained in the soil. Assuming that V_l is the volume of the liquid phase (water) in the soil sample and that V_t is the total volume of the sample, the volumetric water content, θ, can then be defined as follows:

$$\theta = \frac{V_l}{V_t} = \frac{V_l}{V_s + V_p} \quad , \tag{2.15.2}$$

where V_s and V_p represent the volumes of the solid phase and the pore space, respectively.

From the definition presented in Equations (2.15.1) and (2.15.2), the volumetric water content, θ, can be expressed in terms of the mass-basis water content, w, according to the following formula:

$$\theta = \frac{V_l}{V_t} = \frac{M_l/\rho_w}{M_s/\rho_b} = \left[\frac{M_l \rho_b}{M_s \rho_w}\right] = w\left[\frac{\rho_b}{\rho_w}\right] \ , \tag{2.15.3}$$

where

ρ_b = bulk density of the soil (see Section 2.1), and

ρ_w = water density.

The volumetric water content is also expressed in terms of the total porosity, p_t, and the water saturation (or saturation ratio), R_s, according to the following expression:

$$\theta = p_t R_s \quad , \tag{2.15.4}$$

where p_t is the total porosity (see Section 2.2 of this handbook), and R_s, the saturation ratio, is defined as the ratio of the volume of water, V_l, to the volume of the pore space, V_p. Therefore, considering the definitions of p_t and R_s, the expression for the volumetric water content can be rewritten as follows:

$$\theta = p_t \, \frac{Vl}{Vp} \ . \tag{2.15.5}$$

The possible values for θ range from near zero, for dry soils approaching zero saturation, up to the value of the total porosity for fully saturated soils. The lower limit of zero for the volumetric water content is hardly achievable, because it is difficult to completely eliminate the water from the soil. In sandy soils, the upper limit, which is equal to the total porosity p_t, is also hardly achievable because of the difficulty of eliminating all the air bubbles from the soil in order to saturate it completely. Yet, because clayey soils swell upon wetting, the values of θ for these soils can exceed their total porosity.

2.15.2 Measurement Methodology

Direct and indirect methods can be used to determine the volumetric water content of soils. The direct methods consist essentially of drying and weighing a known volume of a soil sample. The indirect methods are based on the correlation of certain physical and physicochemical properties of the soil with its water content.

An extensive discussion on both direct and indirect methodologies for measuring water content in soils is presented by Gardner (1986). A range of comparative assessments of soil water sensing methods under laboratory and field conditions were conducted by soil water instrumentation experts from different countries throughout the world. and the results of these studies is presented in IAEA Training Course Series No. 30 (IAEA 2008). Klysz and Balayssac (2007) used ground-penetrating radar to determine the volumetric water content of concrete.

On FUSRAP sites, the standard method used for determining the (mass-basis) water content in soil materials is ASTM D 2216-10 (ASTM 2010e). This method is related to the determination of the mass-basis water content, *w*, rather than to the volumetric water content, as required in the RESRAD (onsite) and RESRAD-OFFSITE codes. However, the volumetric water content can be determined from Equation (2.15.3) when the mass-basis water content and the bulk density of the soil material (see Section 2.1 of this handbook) are known.

In general, in a direct measurement method, the volumetric water content of a soil sample is evaluated on the basis of three measured quantities: (1) W_w, the wet weight of the soil sample; (2) W_d, the oven-dried weight of the sample; and (3) V_t, the field volume or the total volume of the sample. With these measured quantities available, the volume of the liquid phase (water), V_l, in the sample can then be calculated as

$$V_l = \frac{W_w - W_d}{\rho_w} \quad , \tag{2.15.6}$$

and the volumetric water content (θ) can finally be determined from Equations (2.15.2) and (2.15.6) as

$$\theta = \frac{V_l}{V_t} = \frac{W_w - W_d}{V_t \rho_w} \quad , \tag{2.15.7}$$

where ρ_w is the density of water.

Variations in the direct methods for determining the volumetric water content are related to different ways of collecting the soil samples, measuring the field volume (V_t), and drying the samples. Possible direct methods of collecting the soil samples and measuring V_t have been discussed in Section 2.1 of this handbook.

The definition of a dry state for the soil sample (and the establishment of a method to achieve this state) constitutes the key problem in determining the volumetric water content in soils. As a common practice, the oven-dried weight of the soil sample is measured after drying the sample at 105°C until a near-constant weight is reached (Hillel 1980b). As discussed by Gardner (1986), however, this oven-drying procedure is not precise enough and could create uncertainties and inaccuracies in the measured result. Therefore, if the determinations of water content for a particular site are considered critical, procedures other than the oven-drying method should be adopted (Gardner 1986).

The indirect methods of measuring the water content in soils rely on certain physical and physicochemical properties of the soil and their relation to the volumetric water content (θ). Usually these relationships are complicated and require a sophisticated methodology and equipment to exploit them. The indirect methods of measuring volumetric water content are applicable for in situ rather than laboratory determinations and involve measuring some property of the soil that is affected by the soil water content, such as (1) electrical conductivity, (2) neutron scattering, (3) neutron and gamma-ray absorption, (4) heat conductivity, (5) travel time or frequency of an electromagnetic pulse, or (6) frequency of an oscillating circuit (Gardner 1986, IAEA 2008). Table 2.15.1 lists the different indirect methods used in measuring the water content in soil. Table 2.15.2 lists the interferences observed in some types of soil water sensors.

As seen in the discussion of the determination of soil densities, the indirect methods used for measuring volumetric water content present some advantages over the other related laboratory techniques. The main advantages are (1) in situ evaluation of the water content;

(2) minimum disturbance of the soil; (3) relatively short measurement time, (4) applicability to deeper subsoil determinations because of minimum excavation requirements; and (5) nondestructiveness, with the possibility of continuous or repeated measurements at the same spot. The disadvantages of such indirect methods are that they are more sophisticated and require

TABLE 2.15.1 Indirect Methods Used by Different Sensors for Measuring Water Content in Soil

Method	Measurement	Explanation
Neutron moisture meter (NMM)	Count of slow neutrons around a source of fast neutrons	A radioactive source emits fast neutrons (5 MeV), which lose energy as they collide with other atoms, in particular hydrogen. The concentration of slow neutrons is measured. Since the only rapidly changing source of hydrogen in the soil is water, water content in soil can be calibrated vs. the count of slow neutrons.
Thermal sensors	Heat conductivity or heat capacity of the soil	A pulse of heat is generated and the subsequent rise or fall in temperature of adjacent soil is measured over time. Soil is a poor conductor of heat and water a good one, so the amount of heat or rate of heat transmission is closely related to water content in soil.
Time domain reflectometer (TDR)	Travel time of an electromagnetic pulse	A fast rise time electromagnetic pulse is injected into a waveguide inserted into or buried in the soil. The time required for the pulse to travel along the metal rods of the waveguide is determined by the bulk electrical permittivity of the soil. The water content in soil is a major factor influencing the bulk permittivity. True TDR involves capture of a waveform and analysis to find the travel time of the highest frequency part of the pulse.
Campbell frequency domain reflectormeter (FDR)	Repetition time for a fast-rise-time electromagnetic pulse	Same as TDR sensors, except reliance on reflected pulse reaching a set voltage rather than waveform analysis causes the method to be more influenced by bulk permittivity and temperature.
Capacitive sensors	Frequency of an oscillating circuit	An oscillating current is induced in a circuit, part of which is a capacitor that is arranged so that the soil becomes part of the dielectric medium affected by the electromagnetic field between the capacitor's electrodes. The water content of soil influences the electrical permittivity of the soil, which in turn affects the capacitance, causing the frequency of oscillation to shift.
Conductivity sensors (e.g., granular matrix sensors and gypsum blocks)	Electrical conductivity of a porous medium in contact with the soil	An alternating current voltage is placed on two electrodes in a porous material in contact with the soil, and the amount of current is a measure of the conductivity and amount of water in the porous material between the electrodes. These sensors are used for estimation of soil water tension (suction).

Tensiometers	Matric and gravitational soil water potential components	Capillary forces retaining water in the soil pores are connected through the soil water to water in a porous cup connected to a tube filled with water. This generates a negative pressure within the tube, which can be measured with a vacuum gauge. This method is used for estimation of soil water tension.

Source: IAEA (2008).

TABLE 2.15.2 Characteristics of Some Types of Soil Water Sensors

Technology	Sensed Volume	Interferences
NMM	3×10^4 cm^3 (wet soil) 28×10^4 cm^3 (dry soil)	Cl, B, Fe, C
TDR	Soil volume along length of probe rods, and ~10 mm above and below the plane of the rods, and 10 mm to the side of the plane of the rods (e.g., ~320 cm^3 for a 20-cm probe with three rods and 3-cm rod-to-rod spacing)	Salt, electrical conductivity of soil and temperature, magnetic minerals
Capacitive and FDR	Highly variable; usually 90% of the reading comes from within 20 mm of the sensitive face of the sensor, but sometimes the sensed volume is smaller than the height of the sensor. Typically ~200–400 cm^3.	Salt, electrical conductivity of soil (including clay type, content, and water content) and temperature
Thermal sensors	Highly variable; 20-mm zone around the sensor, which is small.	Metallic soil components
Conductivity sensors	Will equilibrate with a volume of soil that is determined by the soil hydraulic conductivity. Typically 500 cm^3 in wet soil, but much smaller in dry soil.	Temperature, salts other than $CaSO_4$ used in the sensor

Source: IAEA (2008).

expensive equipment and highly trained operators who must be able to handle the frequent calibration procedures, the electronics, and the sampling equipment. In the case of a system that uses radioactive elements, the operator must be particularly well trained in the radiation aspects and radiological protection procedures of the whole operation.

2.15.3 Data Input Requirements

To use RESRAD (onsite) and RESRAD-OFFSITE, it is necessary to define an input value for the volumetric water content (θ) of the soil of the cover zone and the building foundation material (i.e., concrete). To use RESRAD-OFFSITE, it is necessary to define an input value for the volumetric water content in agricultural areas, livestock feed-growing areas, and off-site dwelling areas. In both codes, the dimensionless values of the volumetric water content are entered as decimal fractions rather than as percentages.

Volumetric moisture content, absorbable moisture, and water exchangeable porosity of the concrete sampled from three demolished buildings (slabs, columns, and walls) were measured at the Savannah River Site. The in-field volumetric moisture content of the concrete samples ranged from 0.070 to 0.132, with an average of 0.096. The volumetric moisture that the samples absorbed ranged from 0.005 to 0.079, with an average of 0.036, and the final saturated moisture content (porosity) ranged from 0.083 to 0.178 with an average of 0.132 (Sappington and Phifer 2005).

For generic use of the model, a set of default values for the volumetric water content is defined internally in the code. The default values are $\theta = 0.05$ for the cover material and $\theta = 0.01$ for the building foundation material (i.e., concrete). Considering the default values for total porosity, 0.4 and 0.1, the volumetric water content values correspond to saturations of 0.125 and 0.1 for the cover material and concrete, respectively.

For more accurate use of the codes, site-specific values of the volumetric water content should be experimentally determined according to the methods presented in Section 2.15.2.

2.16 FIELD CAPACITY

2.16.1 Definition

The field capacity, θ_r, also called specific retention, irreducible volumetric water content, or residual water content, is defined as the ratio of the volume of water retained in the soil sample, after all downward gravity drainage has ceased, to the total volume of the sample. The field capacity is related to the total porosity and the effective porosity, as discussed in Sections 2.2 and 2.3, respectively, of this handbook.

The field capacity is the upper limit of water storage and the plant wilting point is the lower limit of water storage. At the wilting point, the plants are no longer able to absorb water from the soil. After irrigation or rainfall that saturates the soil, because of gravitational force there will be a rapid downward movement (drainage) of some soil water. The velocity of the drainage depends on the hydraulic conductivity of the soil. When the soil is at field capacity, the spaces between the soil particles contain both water and air. When the soil is saturated, all the spaces between soil particles are filled with water. The field capacity depends on the structure and texture of the soil. For example, owing to its large grain sizes, sand does not have the large surface area needed to hold a lot of water and therefore has a low field capacity. In contrast, owing to large surface area, clay soil has a high field capacity. Table 2.16.1 lists field capacity and plant wilting point for different soil types.

2.16.2 Measurement Methodology

To determine the field capacity of a soil, the soil sample is first saturated with water and covered to prevent evaporation, and is then allowed to drain completely under the action of gravity until it reaches its irreducible saturation. The value of θ_r can then be obtained according to the methods used for measuring volumetric water content (Section 2.15.2).

TABLE 2.16.1 Field Capacity and Plant Wilting Point for Different Soil Types

Soil Type	Field Capacity	Plant Wilting Point
Sand	0.1	0.05
Loamy sand	0.12	0.05
Sandy loam	0.18	0.08
Sandy clay loam	0.27	0.17
Loam	0.28	0.14
Sandy clay	0.36	0.25
Silty loam	0.31	0.11
Silt	0.30	0.06
Clay loam	0.36	0.22
Silty clay loam	0.38	0.22
Silty clay	0.41	0.27
Clay	0.42	0.30

Source: Saxton and Rawls (2006).

2.16.3 Data Input Requirements

To use RESRAD (onsite) and RESRAD-OFFSITE, the user is required to define (or to use the default values of) the field capacity of two distinct materials: (1) contaminated zone, and (2) unsaturated zone. For the RESRAD (onsite) code, field capacity of the saturated zone is also required. In both codes, the field capacity values are entered as decimal fractions rather than as percentages. As a default value, the RESRAD (onsite) code adopts the value of 0.2 for all three materials and the RESRAD-OFFSITE adopts the value of 0.3 for the contaminated and unsaturated zones. These default values are provided for generic use of the codes. For more accurate use of the code, site-specific data should be applied.

If site-specific data are not available and the soil type is known, Table 2.16.1 can be used for estimating the field capacity. However, if no information is available on soil type, then the values of field capacity should be experimentally determined according to the method presented in Section 2.16.2.

This page intentionally left blank.

3 METEOROLOGICAL PARAMETERS

3.1 PRECIPITATION RATE

3.1.1 Definition

The precipitation rate, *Pr*, is the average volume of water in the form of rain, snow, hail, or sleet that falls per unit of area and per unit of time at the site. It is measured in units of volume per area per time (LT^{-1}).

Precipitation is one of the primary processes of the hydrologic cycle, that is, the endless movement of water through the various elements of the environment (oceans, atmosphere, land surface water bodies, and subsurface soil systems). Other processes of the hydrologic cycle include evapotranspiration, infiltration, overland flow (runoff), streamflow, deep percolation, and groundwater flow. Thorough descriptions of these processes have been presented in numerous texts in the hydrology literature (Chow 1964; Linsley et al. 1982; Bedient, Huber 1988, and Smith 1992).

A simplified description of the hydrologic cycle could start with considering the water vapor contained in the atmosphere, which under appropriate conditions condenses and precipitates over the oceans and the continental land. The portion of the water that falls over the surface land, that is, precipitation, is subsequently dispersed by following different pathways. Thus, from the precipitation, a parcel of water is retained in the vicinity of the place where the precipitation falls and is then transferred back to the atmosphere through evaporation (i.e., the conversion of the water from a liquid at the soil surface to a vapor) and transpiration (i.e., the indirect loss of water vapor from the soil to the atmosphere through plant tissue). The combined effect of evaporation and transpiration is commonly called evapotranspiration. Another parcel of the precipitation water penetrates the subsurface soil system, in a process known as deep percolation, and is added to the groundwater flow system. Finally, the last parcel of precipitation water (the one that is not transferred back to the atmosphere and does not percolate deep into the soil) becomes overland flow, also called surface runoff, and feeds local streams, rivers, or lakes. Both the surface and the subsurface flows of water move toward low elevations and eventually reach the oceans. Evaporation, primarily from the oceans and inland surface waters, transfers water vapor back to the atmosphere, thus completing the hydrologic cycle.

The concept of the hydrologic cycle is applicable to a large-scale hydrologic system on earth and can be represented mathematically by a water balance (or budget) equation based on the law of the conservation of matter. The same principle can be applied to any hydrologic system of any scale, whether it is a small basin or a large watershed, to generate a water balance equation that, in its simplest form, can be expressed as follows:

$$q_{in} - q_{out} = \frac{ds}{dt} , \tag{3.1.1}$$

where

q_{in} = water inflow rate into the system,

q_{out} =outflow rate, and

ds/dt = change with time of the water stored within the system.

To illustrate the application of the water balance concept, consider a hydrologic system represented by irrigated agricultural land and the movement of water through it. According to the law of the conservation of matter, ΔS (i.e., the change in the volume of water stored in the soil per unit of land surface area) during a given time period ΔT must be equal to the difference between the average inflow rate in time and space (i.e., precipitation, P_r, plus irrigation, IR_r, rates) and the outflow rate (i.e., deep percolation, I_r, plus runoff, R_r, and evapotranspiration, ET_r, rates). The water balance equation for this system could then be represented as follows:

$$\frac{\Delta S}{\Delta T} = \left(P_r + IR_r\right) - \left(I_r + R_r + ET_r\right), \qquad (3.1.2)$$

where all the inflow and outflow rates are expressed in units of LT^{-1}.

The precipitation over a specific hydrologic system is an erratic process with large fluctuations in the time domain. Consequently, because all the inflow and outflow processes mentioned previously are related to the precipitation, they also present large and erratic variations over time. As a result, the change in the soil-water storage (ΔS) is highly dependent on the period of time (ΔT) being considered. For short periods, ΔS is also an erratic process and can present relatively large values. However, for a long period, such as an entire season or a whole seasonal cycle of one year, ΔS, particularly in the upper part of the soils, is likely to be small in relation to the total water balance of the system (Hillel 1980a).

Thus, considering annual averages of the inflow and outflow water rates in this hypothetical hydrologic system of a generic piece of irrigated agricultural land, the respective water budget equation can be reduced to the following:

$$P_r + IR_r = I_r + R_r + ET_r \qquad (3.1.3)$$

Except for the deep percolation rate, I_r, all the terms of Equation (3.1.3) can be determined either by performing direct field measurements or by using specific coefficients derived from soil and other environmental characteristics. The experimental methodologies for field measurement of the precipitation, runoff, irrigation, and evapotranspiration rates are described in this handbook (Sections 3.1.2, 3.2.2, 3.3.2, and 3.4.2, respectively). Direct field measurement of the deep percolation (infiltration) component of the field water balance has not yet proven to be practical (Hillel 1980a) and, therefore, the deep percolation rate is often determined from the other measured components of the equation as follows:

$$I_r = \left(P_r + IR_r\right) - \left(R_r + ET_r\right) \tag{3.1.4}$$

The parameter I_r, or the water deep percolation rate, represents the amount of water that percolates through the upper layers of the soil and eventually ends up being added to the groundwater flow underneath the hydrologic system. In the RESRAD (onsite) model, the parameter I_r is used to calculate the radionuclide leaching from the contaminated zone and the final contamination of the groundwater. The deep percolation rate is not user input but is calculated internally in the code as a function of the precipitation (P_r) and irrigation (IR_r) rates and the runoff (C_r) and evapotranspiration (C_e) coefficients. The latter two parameters are defined, respectively, as follows:

$$C_r = \frac{R_r}{P_r} \tag{3.1.5}$$

and

$$C_e = \frac{ET_r}{\left(1 - C_r\right)P_r + IR_r} \; . \tag{3.1.6}$$

Detailed discussion of the runoff (C_r) and evapotranspiration (C_e) coefficients and the irrigation rate (IR_r) are presented in Sections 3.2.1, 3.3.1, and 3.4.1, respectively.

Thus, from Equations (3.1.4), (3.1.5), and (3.1.6), the deep percolation rate, I_r, can be expressed as follows:

$$I_r = \left(1 - C_e\right)\left[\left(1 - C_r\right)P_r + IR_r\right] \tag{3.1.7}$$

The mass balance equation, Equation (3.1.7), is the one used in RESRAD (onsite) to calculate the deep percolation rate of water into the soil.

3.1.2 Measurement Methodology

The precipitation rate at a site-specific location can be measured with a precipitation gauge, which basically consists of a receptacle with vertical walls and an opening at the top with a specified area. The ratio of the volume collected in the receptacle during a specified period of time to the area of the opening at the top of the receptacle gives the point estimate of the precipitation rate at a specific location and time.

In principle, any receptacle with an open collector area of known dimensions, plus a volume-measuring device, can be used as a precipitation gauge. However, because of some operational features of these devices, unless they are of the same shape and dimensions and similarly exposed, precipitation rate measurements are usually not comparable (Linsley et al. 1982).

The standard precipitation gauge adopted by the U.S. National Weather Service has a collector (receiver) with an 8-in. (20.3-cm) diameter and can measure the precipitation to the nearest 0.25 mm. Two types of precipitation gauges can be used, recording and nonrecording. The recording gauge, the most commonly used, records on a strip of paper, paper punch, or data logger every 0.01 in. (0.0254 cm) of precipitation along the time scale. The recorded data are then reported as an average precipitation rate, total volume, or intensity variation.

According to Bedient and Huber (1988), a network of five to ten gauges per 260 km^2 (100 mi^2) is usually required in urban areas to define precipitation variability. The maintenance costs of such networks are high and, therefore, for a particular application, it is usually more convenient to rely on data collected locally from existing networks with gauges already installed near the site of interest. Local rain gauge networks that are usually maintained by cities and sewage treatment plants, for example, could serve as a first source of information on the precipitation rate at the site. On a larger scale, information on the precipitation rate could be obtained from national networks. Precipitation gauge networks designed to provide point estimates of precipitation rates in the United States and its territories are maintained by the U.S. National Weather Service and the U.S. Geological Survey.

Data on the point estimates of precipitation rates obtained from either local or national networks can be used to estimate the average areal precipitation rate over a specific area. The areally averaged values of the precipitation rate can be derived by three methods (Bedient and Huber 1988): arithmetic mean, the Thiessen polygon method, and the isohyetal method.

The arithmetic mean of the point precipitation rates provides the simplest and most straightforward way to obtain an estimate of the areal precipitation rate at a particular site. For cases in which the gauges are uniformly distributed and the point values have minimal variations, this method provides satisfactory results.

The Thiessen polygon method consists of areally weighing the point precipitation from each gauge. This is the most commonly used method, although not the most accurate.

The isohyetal method consists of drawing contour lines of equal precipitation (isohyets) and areally weighing the average precipitation between pairs of contour lines crossing over the area of the site being considered. It is the most accurate among the methods for determining areally averaged values of the precipitation rate but requires an extensive gauge network to draw the isohyets accurately.

A distribution of values of average annual precipitation rates over the U.S. continental territory, transcribed from the most current PRISM climate dataset of 30 years, is shown in Figure 3.1.1.

If measurements are taken for a site-specific precipitation rate, users are referred to DOE’s environmental regulatory guide (DOE 1991) on radiological effluent monitoring.

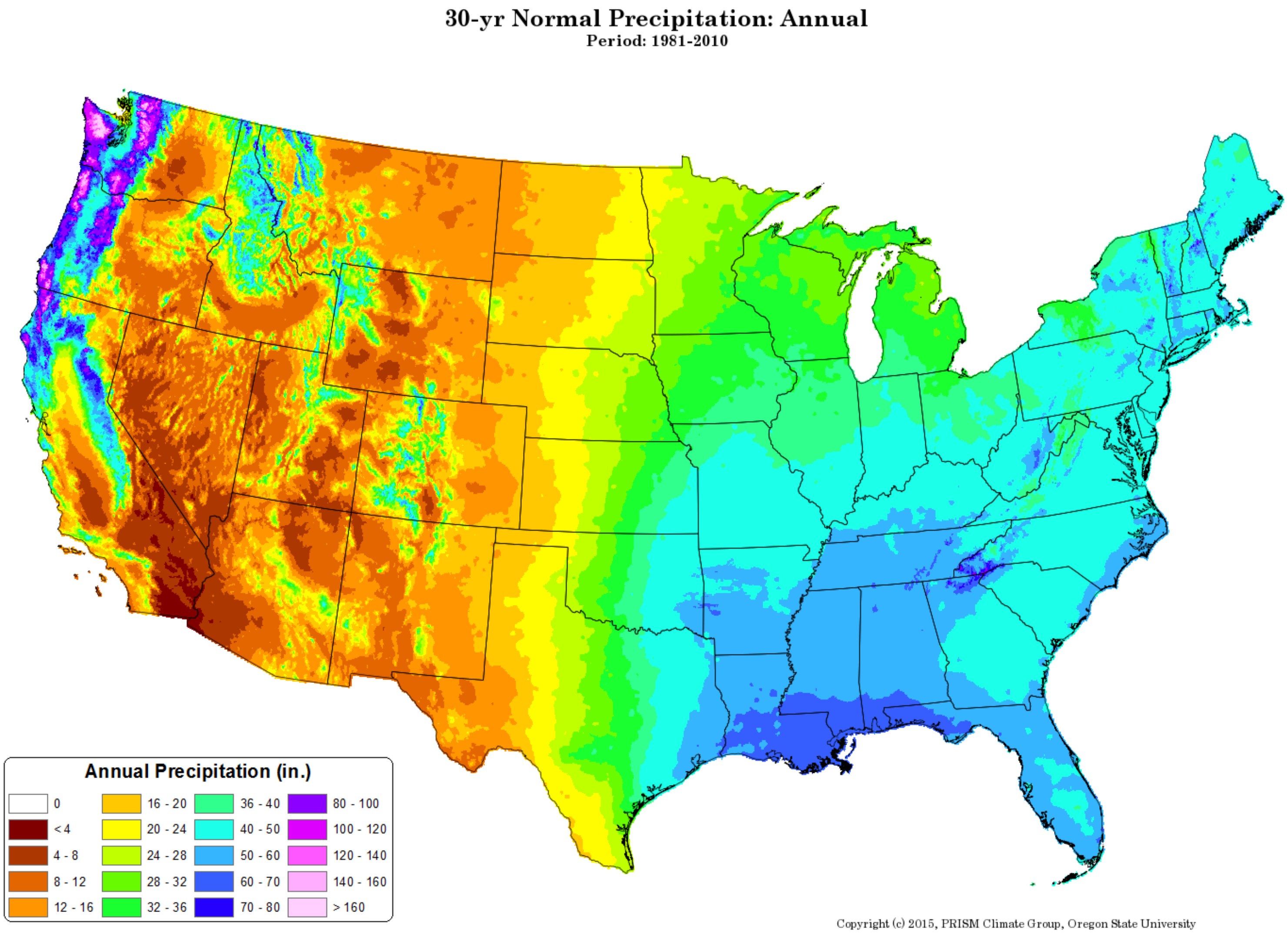


FIGURE 3.1.1 Distribution of Average Annual Precipitation Rates (in./yr) over the U.S. Continental Territory (Source: http://www.prism.oregonstate.edu/normals/ [Copyright © 2015, PRISM Climate Group, Oregon State University, http://prism.oregonstate.edu, Map created on June 22, 2015])

3.1.3 Data Input Requirements

In RESRAD (onsite) and RESRAD-OFFSITE, the user is requested to input a value for the annual average areal precipitation rate (P_r) that is representative of the conditions at the site. The precipitation rate is expressed as an annual average rate in units of meters per year (m/yr).

The precipitation rate and other input parameters, such as the irrigation rate and the runoff and evapotranspiration coefficients (Sections 3.4.1, 3.2.1, and 3.3.1, respectively), are used in RESRAD (onsite) and RESRAD-OFFSITE to determine the water deep percolation rate, according to Equation (3.1.7). The deep percolation rate is ultimately used to calculate the radionuclide leaching rate of the contaminated zone and the subsequent contamination of the underlying groundwater system.

For generic use of the codes, a default value of the precipitation rate (P_r) equal to 1 m/yr (about 40 in./yr) was adopted in the RESRAD (onsite) and RESRAD-OFFSITE models, as shown in Table 3.1.1. This value approximately represents the condition of a relatively humid region. Whenever possible, however, and especially for sites located in a dry region of the country, such as in the western United States, site-specific input data for P_r should be used in the RESRAD (onsite) and RESRAD-OFFSITE calculations.

Annual average values of P_r in units of in/yr for the U.S. continental territory, based on 30 years (1981–2010) of recording, are presented in the US Department of Agriculture PRISM project (http://www.prism.oregonstate.edu/normals/). In the absence of site-specific data, the information provided in this atlas can be used as a provisional and gross estimate of the site-specific value of P_r at any particular location in the United States.

Site-specific data on the precipitation rate at a site can be obtained from a rain gauge network installed around the site or from already installed networks, such as those maintained by cities. In cases in which data are available on the annual average point precipitation rates at specific locations in the vicinity of a site, the user can estimate the site-specific areal precipitation rate by using one of the three averaging methods described in Section 3.1.2.

TABLE 3.1.1 Default Precipitation Rates Used in RESRAD (onsite) and RESRAD-OFFSITE

Parameter Name	Unit	Default Value	Code-Accepted Values	Reference	Description
Precipitation rate	m/yr	1.0	0–10	Yu et al. 1993	The average volume of water in the form of rain, snow, hail, or sleet that falls per unit of area per unit of time at the site.

If data on the precipitation rate (P_r) are not being collected at a site or its vicinity, a site-specific estimation of P_r can be obtained from the U.S. National Weather Service or the U.S. Geological Survey network database. The user may also refer to *Climatological Data, National Summary* and *Climatic Atlas of the United States*, published by the U.S. Environmental Data Service, for a site-specific estimate of P_r, if no local data are available.

3.2 RUNOFF COEFFICIENT

3.2.1 Definition

The average annual runoff coefficient, C_r, is the fraction of the average annual precipitation that does not infiltrate into the soil and is not transferred back to the atmosphere through evapotranspiration. The runoff coefficient represents the fraction of the precipitation, in excess of deep percolation and evapotranspiration, that becomes surface flow and ends up in either perennial or intermittent surface water bodies. The runoff coefficient is a dimensionless parameter.

In a well-designed and well-operated irrigation system, the flow and the quantity of the irrigation water are controlled by an appropriate drainage system (ditching) and the duration of each application. Consequently, under normal circumstances, the irrigation water does not contribute significantly to the overall average annual runoff. On the basis of these assumptions, the average annual runoff coefficient (C_r) can be defined mathematically by the following expression:

$$C_r = \frac{R_r}{P_r} , \qquad (3.2.1)$$

where

R_r = average annual runoff rate and

P_r = average annual precipitation rate.

Because R_r is always smaller than (or at the most equal to) P_r, the values of C_r vary within the range of zero to one.

The runoff rate at a specific location is influenced by the morphology of the region, the degree of the slopes, the type of soil material, and the type of soil utilization. Table 3.2.1 lists values for the runoff coefficient, C_r, under various conditions of soils and soil uses.

3.2.2 Estimation Methodology

A methodology for estimating the runoff coefficient (C_r) is presented in Table 3.2.1. The value of Cr can be evaluated on the basis of the type of soil and the land utilization at the specific site.

3.2.3 Data Input Requirements

In RESRAD (onsite) and RESRAD-OFSITE, the user is requested to input a value for the average annual runoff coefficient (C_r) that represents conditions at the site. The runoff coefficient is a dimensionless parameter and its input value should be entered in the form of a decimal fraction rather than as a percentage.

For generic use of the codes, a default value of 0.2 was adopted in the RESRAD (onsite) and RESRAD-OFFSITE models for the runoff coefficient. According to the methodology presented in Table 3.2.1, this default value of C_r represents an agricultural environment of cultivated (c3 = 0.1), flat (c1 = 0.3) land with a sandy loam type of soil (c2 = 0.4). Whenever possible, however, site-specific information should be applied for more accurate use of the code. If site-specific data are not available, Table 3.2.1 may be used to estimate the average annual runoff coefficient (C_r).

The runoff coefficient and other input parameters, such as the precipitation and irrigation rates and the evapotranspiration coefficient (Sections 3.1.1, 3.4.1, and 3.3.1, respectively), are used in RESRAD (onsite) and RESRAD-OFFSITE to determine the water deep percolation rate according to a mass balance equation, Equation (3.1.7), presented in Section 3.1. The water deep percolation rate is ultimately used to calculate the radionuclide leaching rate of the contaminated zone and the subsequent contamination of the underlying groundwater system.

TABLE 3.2.1 Runoff Coefficient Values

Type of Area	Coefficient	Value
Agricultural environment[a]		
Flat land with average slopes of 0.3–0.9 m/mi	c1	0.3
Rolling land with average slopes of 4.6–6.1 m/mi	c1	0.2
Hilly land with average slopes of 46–76 m/mi	c1	0.1
Open sandy loam	c2	0.4
Intermediate combinations of clay and loam	c2	0.2
Tight, impervious clay	c2	0.1
Woodlands	c3	0.2
Cultivated lands	c3	0.1

Table 3.2.1 (Cont.)

Type of Area	Coefficient	Value
Urban environment		
Flat, residential area, about 30% impervious	C_r	0.4
Moderately steep, residential area, about 50% impervious	C_r	0.65
Moderately steep, built-up area, about 70% impervious	C_r	0.8
Lawns		
Sandy soil, flat, 2%	C_r	0.1
Sandy soil, average, 2–7%	C_r	0.15
Sandy soil, steep, >7%	C_r	0.2
Clay soil, flat, 2%	C_r	0.17
Clay soil, average, 2–7%	C_r	0.22
Clay soil, steep, >7%	C_r	0.35
Unimproved areas (forest)	C_r	0.15
Business		
Downtown areas	C_r	0.95
Neighborhood areas	C_r	0.7
Residential		
Single-family areas	C_r	0.5
Multifamily units, detached	C_r	0.6
Multifamily units, attached	C_r	0.7
Suburban	C_r	0.4
Apartment dwelling areas	C_r	0.7
Industrial		
Light areas	C_r	0.7
Heavy areas	C_r	0.8
Parks, cemeteries	C_r	0.25
Playgrounds	C_r	0.35
Railroad yard areas	C_r	0.4
Streets		
Asphaltic and concrete	C_r	0.95
Brick	C_r	0.85
Drives, walks, and roofs	C_r	0.95
Gravel areas	C_r	0.5
Graded or no plant cover		
Sandy soil, flat, 0–5%	C_r	0.3
Sandy soil, flat, 5–10%	C_r	0.4
Clayey soil, flat, 0–5%	C_r	0.5
Clayey soil, average, 5–10%	C_r	0.6

3.3 EVAPOTRANSPIRATION COEFFICIENT

3.3.1 Definition

Evapotranspiration is one of the processes of the hydrologic cycle and represents the total volume of water that changes phase, that is, from the liquid or solid state to the gaseous state, near the ground surface and is transferred to the atmosphere during a fixed period of time. Consequently, it represents the combination of two separate processes: (1) evaporation (i.e., the change of phase of water near the ground surface and the direct transfer of water vapor from the ground to the atmosphere) and (2) transpiration (i.e., the transfer of water from the ground to the atmosphere through plants and their foliage).

Evapotranspiration is also called “consumptive use” in the hydrology literature and is defined as the quantity of water used by either cropped or natural vegetation in transpiration or in the building of plant tissue, together with water evaporated from the adjacent soil or from intercepted precipitation, during a fixed period of time (Veihmeyer 1964).

Two parameters need to be defined in relation to the concept of evapotranspiration: (1) the evapotranspiration rate, Et_r, and (2) the evapotranspiration coefficient, C_e.

The evapotranspiration rate, ET_r, is the total volume of water vapor that is transferred to the atmosphere because of the combined effect of evaporation and transpiration, per unit of the ground surface area and per unit of time at the site. It is measured in units of volume per area per time (LT^{-1}). The evapotranspiration rate is neither required as input data to the RESRAD code, nor is it used implicitly within the model. However, the measured or estimated site-specific value of Et_r is used to estimate the input value of the evapotranspiration coefficient, which is used in the code. For consistency with other correlated parameters handled in the RESRAD code, the evapotranspiration rate is expressed as an annual average rate in units of meters per year (m/yr).

The evapotranspiration coefficient, C_e, is the ratio of the total volume of water leaving the ground as the result of evapotranspiration, Et_r, to the total volume of water available within the root zone of the soil $[(1 - C_r)P_r + IR_r]$ during a fixed period of time. It can then be expressed as follows:

$$C_e = \frac{ET_r}{\left(1 - C_r\right) P_r + IR_r} , \tag{3.3.1}$$

where

P_r = precipitation rate (m/yr),

IR_r = irrigation rate (m/yr), and

C_r = runoff coefficient (dimensionless).

(All these parameters are defined in this handbook; see Sections 3.1.1, 3.4.1, and 3.2.1, respectively.)

In well-irrigated agricultural land, transpiration predominates over evaporation in composing the total evaporation. Under these circumstances, the evapotranspiration coefficient represents the efficiency by which the water available in the root zone of the soil is actually transferred through the plant system and into the atmosphere. Thus, for cultivated land, the evapotranspiration coefficient (C_e) is also called the “irrigation efficiency.” Most irrigation projects are inherently inefficient; the average irrigation efficiency is less than 50% (Hillel 1980a).

The evapotranspiration process is fundamentally governed by the meteorological conditions at the site, as well as by the properties of the soil/plant system. Meteorological parameters such as air temperature, wind speed, atmospheric pressure, air humidity, and exposure to the sun all have an important role in determining the evapotranspirational demand at a specific location and time of year. However, it is the amount of water available in the root zone of the soil that limits the occurrence of the evapotranspiration process. Thus, the power of the atmosphere to extract water from the ground surface because of evaporation decreases as the moisture content of the soil decreases. The smaller the moisture content, the more strongly the water is bound to the porous matrix of the soil because of capillarity, and thus more energy is needed to extract it. Transpiration is also limited by the availability of water at the root zone, the ability of the soil to supply and transmit water toward the root zone, and the ability of the root system to absorb water from the soil in its vicinity. Below a certain value of soil moisture called the plant wilting point, the roots of the plants are not able to extract water from the soil, and the transpiration process is broken, resulting in dehydration and wilting. Therefore, as a combination of evaporation and transpiration, the actual evapotranspiration at a specific site depends on external climatic conditions and on the type and density of vegetation covering the ground surface as well as on soil moisture, root distribution, and other soil properties.

The concept of the “potential evapotranspiration rate,” ET_{pr}, has been introduced into the hydrology literature to represent the so-called “climatic demand” for water, independently of the transient properties of the soil (Hillel 1980a). As such, the potential evapotranspiration rate, ET_{pr} (or the evaporating power of the atmosphere), is defined as the evapotranspiration rate that occurs on the ground of a land area totally covered with vegetation and where sufficient water is continuously available for the needs of the plants. The actual evapotranspiration rate, ET_r, is then a function of the potential evapotranspiration rate, ET_{pr}, and the quantity of water available in the root zone of the soil. Where there is an excess of water in the root zone, the value of ET_r is at its maximum, equal to ET_{pr}, and the excess water percolates the soil toward the groundwater system. During a water shortage period, however, the value of ET_r becomes lower than ET_{pr}, with no resulting percolation.

The actual evapotranspiration rate, ET_r, at any location in the contiguous U.S. territory can be estimated from 30 years (1971–2000) of actual evapotranspiration data (Sanford and Selnick 2012). A distribution of average actual evapotranspiration over the U.S. continental territory is shown in Figure 3.3.1.

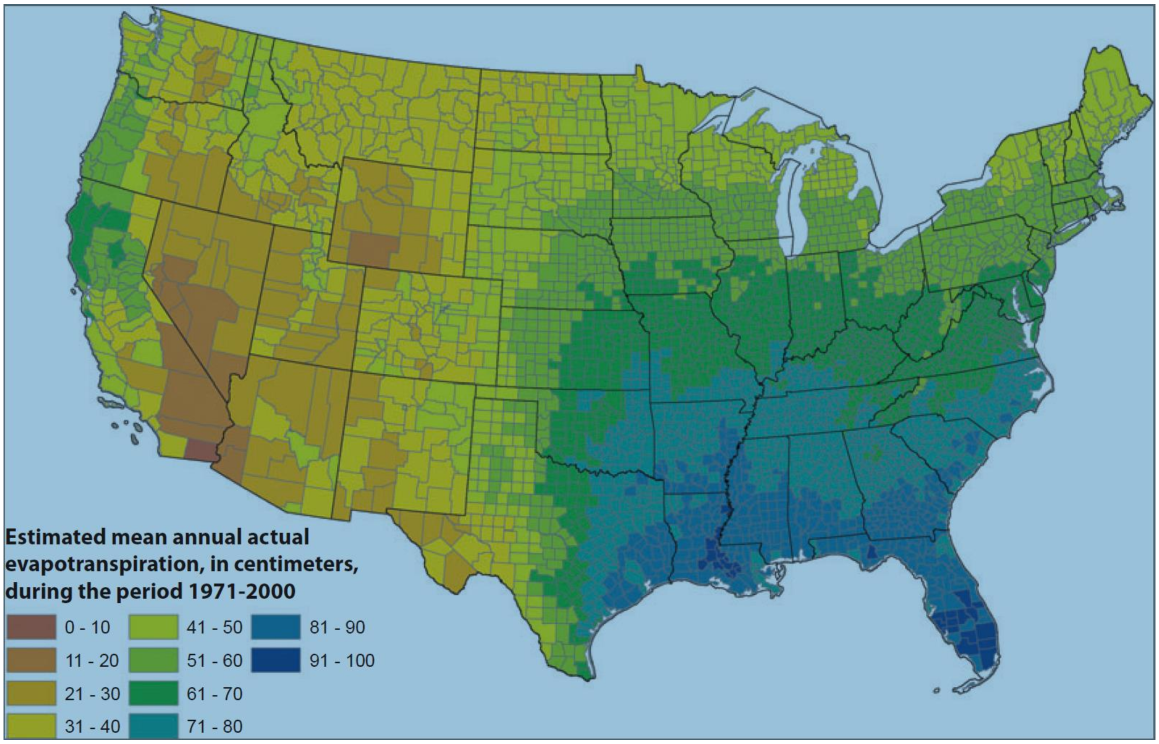


FIGURE 3.3.1 Distribution of Average Annual Actual Evapotranspiration Rate (cm/yr) over the U.S. Continental Territory (Source: Sanford and Selnick (2012); Permission granted by John Wiley and Sons with License Number 3722641205262)

The evapotranspiration coefficient and other input parameters such as the precipitation rate, the irrigation rate, and the runoff coefficient are used in RESRAD to determine the water percolation rate, according to Equation (3.1.7) in Section 3.1.1. The water percolation rate is ultimately used to calculate the radionuclide leaching rate of the contaminated zone and the subsequent contamination of the underlying groundwater system.

3.3.2 Measurement Methodology

Estimation of the evapotranspiration coefficient, C_e (to be used as input data to the RESRAD (onsite) and RESRAD-OFFSITE codes), should be obtained from measured (or otherwise estimated) values of the evapotranspiration rate, ET_r, the precipitation rate, P_r, the irrigation rate, IR_r, and the runoff coefficient, C_r, according to Equation (3.3.1).

There are many methods of measuring or estimating the actual (ET_r) and the potential (ET_{pr}) evapotranspiration rate. However, no one method can be used for all purposes (Veihmeyer 1964). Most of the methods used for estimating ET_r can also be used for estimating ET_{pr}, provided that the available water supply is sufficient for the area under observation during the duration of the test. These methods can be classified into three broad categories: (1) the theoretical approach, based on physical principles governing the process; (2) the analytical approach, based on conservation principles, either as a mass or as an energy balance; and (3) the empirical approach, based on experimental results expressing the correlation between measured evapotranspiration and local climatic conditions.

A generic description of various methods used for measuring evapotranspiration is presented by Veihmeyer (1964). The methods available are (1) soil-moisture sampling, (2) lysimeter measurement, (3) inflow-outflow measurements, (4) integration method, (5) energy balance, (6) vapor transfer, and (7) groundwater fluctuations. For example, the lysimeter method consists of using a large barrel (also called a tank or evapotranspirometer) with about a 1-m diameter and a 2-m depth that is filled with soil and buried in the ground so that its top is flush with the ground surface. Individual crops and/or natural vegetation are grown on and around the lysimeter. The evapotranspiration rate can then be determined on the basis of the mass balance by measuring the infiltration flux seeping out of the bottom of the lysimeter and the rainfall rate. The loss of water necessary to maintain satisfactory plant growth represents the evapotranspiration. When operated properly, the lysimeter can provide reasonably reliable values of potential evapotranspiration. However, reliable measurements of actual evapotranspiration (particularly when it is much lower than the potential) are rarely attainable because of the difficulty in maintaining comparable soil moisture and vegetation cover conditions on and around the lysimeter (Linsley et al. 1982).

Because of the inherent difficulties of field methods for measuring evapotranspiration, several empirical formulas have been developed to relate the potential evapotranspiration to some readily available climatic data, such as temperature, sunshine, wind velocity, and so forth. A list of typical evapotranspiration equations is presented in Table 11.2 of the *Handbook of Applied Hydrology* (Veihmeyer 1964, pp. 11–27). Two publications from the NOAA (1982a,b) have been used in estimating the potential evapotranspiration on FUSRAP sites and the mean annual actual evapotranspiration over the period 1971–2000 (Sanford and Selnick 2012) for cases in which no site-specific data are available.

3.3.3 Data Input Requirements

In RESRAD (onsite) and RESRAD-OFFSITE, the user is requested to input a value for the annual average evapotranspiration coefficient, C_e, that is representative of conditions at the site. The input value of C_e is given in dimensionless units.

In the process of estimating the value of C_e as an input value for RESRAD (onsite) and RESRAD-OFFSITE, it is assumed that the cultivated land at the site under consideration is maintained with the necessary level of moisture in the soil for the growth and development of the crop. This condition is achieved either by natural precipitation or by a combination of precipitation and irrigation. In other words, it is assumed that the required moisture content for potential evapotranspiration based on the annual average is maintained in the soil.

Therefore, the estimation of the input value of C_e for some site-specific conditions is based on a previously measured (or otherwise determined) value of the potential evapotranspiration, ET_{pr}, the precipitation rate, P_r, the irrigation rate, IR_r, and the runoff coefficient, C_r, according to the definition of C_e presented in Equation (3.3.1).

A default value of C_e equal to 0.5 (dimensionless) was adopted in the RESRAD (onsite) and RESRAD-OFFSITE models. This value represents the condition of 50% efficiency in the irrigation process at a generic site. Under this condition, 50% of the water available in the root zone of the soil is transferred to the atmosphere, and 50% of the water infiltrates the soil and percolates toward the aquifer system. Whenever possible, however, site-specific input data for C_e should be used in the calculations.

Field measurements of the average annual evapotranspiration rate, ET_r, usually are expensive and time-consuming. Therefore, if data on ET_r have not been collected at the site or its vicinity, a site-specific estimation of ET_r (and ultimately of C_e) should be obtained from information in the literature. For a gross estimation of ET_{pr}, the user can consult the annual average values of potential evapotranspiration for the U.S. continental territory as shown in Figure 3.3.1. Two NOAA publications (NOAA 1982a,b) provide useful information that can be used to estimate the value of ET_r (and ultimately of C_e) at any particular location in the United States. For most applications, in the absence of site-specific data, this approach should suffice because of the intrinsic uncertainties associated with the model itself and the natural variability of the potential evapotranspiration at any site.

3.4 IRRIGATION RATE

3.4.1 Definition

The irrigation rate, IR_r, is the average volume of water that is added to the soil at the site per unit of surface area and per unit of time. It is measured in units of volume per area per time, or LT^{-1}. In the RESRAD (onsite) and RESRAD-OFFSITE codes, the irrigation rate is expressed as an annual average rate in units of meters per year (m/yr). It is not the actual rate of irrigation applied during the growing season but is the amount of irrigation water applied over a period of one year.

Irrigation is the practice of supplying water artificially to the soil in order to permit agricultural use of the land in an arid region or to compensate for occasional droughts in semi-dry or semi-humid regions. Irrigation is closely dependent on the precipitation rate at the site, in the sense that a well-designed and well-operated irrigation system should optimize the spatial and temporal availability of water in the soil.

As discussed earlier (Section 3.1), irrigation, in conjunction with precipitation, provides the inflow water into a hydrologic system formed by the soil in a section of agricultural land and the water that circulates through it. The outflow of water in this system is the result of processes such as surface runoff, evapotranspiration, and deep percolation rates.

The irrigation rate and other input parameters such as the precipitation rate and the runoff and evapotranspiration coefficients are used in the RESRAD (onsite) and RESRAD-OFFSITE codes to determine the following:

- The rate of infiltration into the primary contamination
  - The transport in the unsaturated zone
- The rate of infiltration in the off-site areas (not applicable for the RESRAD [onsite] code)
- The accumulation at off-site agricultural areas, pastures and dwelling sites (not applicable for the RESRAD [onsite] code)
- The foliar uptake by vegetation
- The computation of the release and distribution of tritium in the environment
- The estimation of the release rate from the primary contamination

3.4.2 Measurement Methodology

The average annual irrigation rate at a site is determined as a ratio of the total volume of irrigation water added to the field during the year to the surface area of the irrigated land. This quantity is not measured in the field per se but is obtained from the operational activities of the irrigation system.

A well designed and operated irrigation system should be able to supply water to plants at a rate sufficient to balance their transpiration rate requirements. The objective is to provide water to the soil in a well-distributed manner during the crop season so that the plants can maintain their own hydration without loss of continuity. As long as the water uptake rate from the plants' roots matches the water loss due to the plants' transpiration from their foliage, they can maintain their hydration. As soon as the water intake from the roots becomes lower than the transpiration, however, the plants start losing moisture, resulting in a stressful situation for the development of the crop (Hillel 1980a).

For irrigation purposes, a maximum allowable depletion or fraction of available water-holding capacity representing the plant's readily available water is the ideal operating range of soil water content for irrigation management (Zotarelli et al. 2010). Ideally, the irrigation should start at low soil water content (much before the soil water content reaches the permanent plant wilting point) and end when the soil water content reaches field capacity.

Therefore, the required rate of irrigation at a specific agricultural site is governed by the properties of the soil and the plants, and, fundamentally, by the meteorological conditions at the site. The soil/plant system properties determine the ability of the soil to supply and transmit water to the roots, as well as the ability of the roots to extract water from the soil at a rate needed to overcome transpiration. The meteorological conditions, however, dictate the rate at which the plants are required to transpire and, therefore, the amount of water needed for their survival.

Estimation of the annual irrigation rate at a specific site can be obtained in different ways, depending on the degree of knowledge about site agricultural activities. When information on irrigation systems in operation at the site or at its vicinity is available, the annual irrigation rate can be obtained from operational records. When little information is available on the irrigation procedures at a site, an estimation of the irrigation rate can be obtained on the basis of the measured (or assessed) values of the potential evapotranspiration and precipitation rates and on the basis of an estimated “irrigation efficiency.”

Irrigation efficiency, C_e, is the ratio of the volume of water used consumptively (such as in evapotranspiration) to the total volume of water applied to the field (Hillel 1980a). This definition is similar to the one for the evapotranspiration coefficient (Section 3.3.1), and can be expressed as follows:

$$Irrigation\ Efficiency = C_e = \frac{ET_r}{(1-C_r)P_r + IR_r} \tag{3.4.1}$$

According to Hillel (1980a), most irrigation projects are inherently inefficient and although irrigation efficiencies of 80 to 90% can be achieved in actual practice with proper water management, the average irrigation efficiency is less than 50%. Thus, by assuming a value for the irrigation efficiency (e.g., about 50%) at a specific site with little available data on agricultural activities; and by determining the potential evapotranspiration rate, ET_r, the precipitation rate, P_r, and the runoff coefficient, C_r; the predicted necessary average annual irrigation rate, IR_r, at the site can be estimated as follows:

$$IR_r = \frac{ET_r}{C_e} - (1 - C_r)P_r \tag{3.4.2}$$

3.4.3 Data Input Requirements

In the RESRAD (onsite) and RESRAD-OFFSITE codes, the user is required to input a value for the annual average irrigation rate, IR_r, that represents conditions at the site. The IR_r should be entered in units of meters per year (m/yr).

A default value of IR_r equal to 0.2 m/yr was adopted in the RESRAD (onsite) and RESRAD-OFFSITE codes. This value approximately represents the conditions of a relatively humid region where only a small amount of irrigation is needed per year. For an arid region, 1 m/yr is considered to be an appropriate generic value for IR_r.

When there is no site-specific information on the annual average irrigation rate, the input value of IR_r at the site can be estimated on the basis of the irrigation efficiency (usually below 50%) and the measurement (or estimation) of the potential evapotranspiration rate, ET_r, the precipitation rate, P_r, the runoff coefficient, C_r, and the evapotranspiration coefficient, C_e (i.e., irrigation efficiency), according to the following expression:

$$IR_r = \frac{ET_r}{C_e} - (1 - C_r)P_r \,. \tag{3.4.3}$$

3.5 AVERAGE ANNUAL WIND SPEED

3.5.1 Definition

The average annual wind speed is the overall average of the wind speed, measured near the ground, in a one-year period. This parameter is measured in units of length per time (LT^{-1}).

3.5.2 Data Input Requirements

In RESRAD (onsite) and RESRAD-OFFSITE, the user is required to input a value for the average annual wind speed that represents conditions at the site. The dimensions of the wind speed input should be given in units of meters per second (m/s).

A default value of 2 m/s with a typical value range of 0.1–20 m/s was adopted in the RESRAD model for the average annual wind speed, as shown in Table 3.5.1. Observations of near-surface wind speeds from two National Climate Data Center (NCDC) datasets containing land-based sites across the contiguous United States are shown in Figure 3.5.1 (Pryor et al. 2009). For more accurate use of the code, however, site-specific data should be used whenever possible. If measurements are performed to get site-specific data, these measurements should be consistent with guidance in DOE's guide for radiological effluent monitoring (DOE 1991).

Site-specific information on the time distribution of the wind speed and direction at the site can be obtained with the installation of a simple meteorological station instrumented with an anemometer (for measuring the wind speed) and wind vanes (for measuring wind direction). Although simple, the installation, operation, and maintenance of such systems are time-consuming and require the attention of trained personnel. A more general estimation of the average annual wind speed at a site can be obtained from other meteorological information systems in the area (such as at a commercial airport). For most applications, in the absence of site-specific data, this approach should suffice because of the intrinsic uncertainties associated with the natural variability of the wind speed and direction at the site.

TABLE 3.5.1 Default Average Annual Wind Speed Used in RESRAD (onsite) and RESRAD-OFFSITE

Parameter Name	Unit	Default Value	Code-Accepted Values	Reference	Description
Average annual wind speed	m/s	2	0.1–20	Yu et al. 1993	The overall average of the wind speed, measured near the ground.

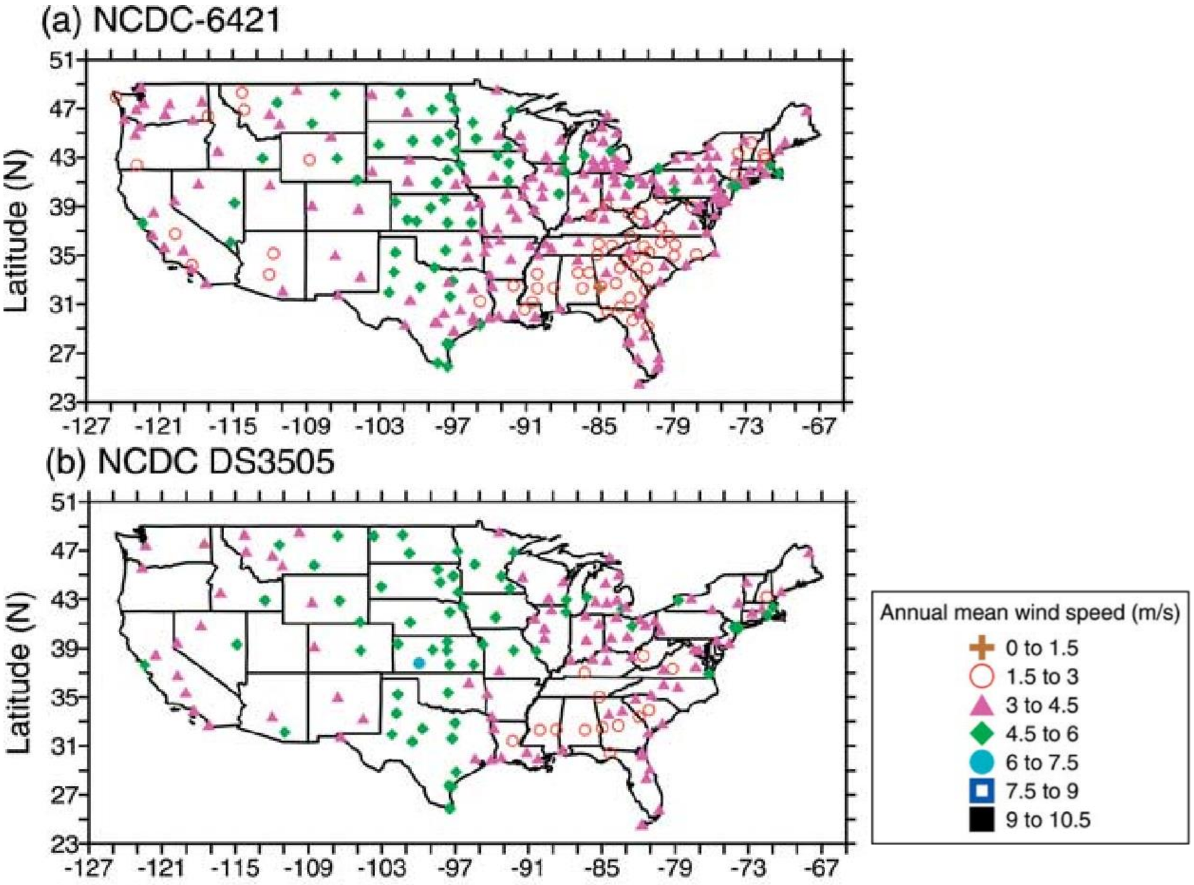


FIGURE 3.5.1 Average of Annual Mean Wind Speeds 1979–2000 from NCDC Data Sets (Source: Pryor et al. 2009; Permission granted by John Wiley and Sons with License Number 3722641022414).

3.6 MASS LOADING FOR INHALATION

3.6.1 Definition

The mass loading parameter is the concentration of soil particles in the air and is obtained directly from empirical data for locations and conditions similar to those applicable for the scenario used. This parameter is measured in grams per cubic meter (g/m^3).

3.6.2 Summary of Literature Review

Three models are commonly used for the process by which dust becomes airborne. One is a resuspension factor model in which the airborne dust concentration (C_{dust}) is given as a function of the resuspension factor (R_f), the effective depth of the layer of dust from which resuspension occurs (d_r), and the bulk soil density (ρ_b). The formula relating these variables is

$$C_{dust} = R_f \, d_r \rho_b \, . \tag{3.6.1}$$

The second is a resuspension rate model in which the airborne dust concentration is given as a function of the resuspension rate (R_r), surface dust concentration ($\sigma_s = \rho_b \, d_r$), and the average deposition velocity (V_d). The formula is

$$C_{dust} = R_r \sigma_s / V_d \tag{3.6.2}$$

The third model used by the RESRAD code is a mass loading model in which an average value of the airborne dust concentration is specified on the basis of empirical data. Use of a mass loading factor from empirical data eliminates consideration of details of the resuspension mechanisms; in particular, the effective depth of the disturbed layer can be ignored.

The mass loading for inhalation input to RESRAD (onsite) and RESRAD-OFFSITE provides the time-averaged respirable concentration of contaminated soil and dust. The respirable portion of resuspended material can be represented by the PM-2.5 fraction of airborne particulate matter (i.e., particulates ≤2.5 µm in diameter). The PM-2.5 particles represent the fine particle fraction that poses the highest respiratory hazard (EPA 2004). The cumulative distribution function for mass loading using the ambient PM-2.5 air concentrations and total suspended particulates concentration from the EPA’s Air Data web site (EPA 2014) were developed for RESRAD (onsite) and RESRAD-OFFSITE (Yu et al. 2007). The size distribution of suspended particulates generally includes a greater fraction of smaller particles than that of originating soil. The suspended particles have larger surface area and consequently may have higher activity per unit mass compared to the activity concentration in the soil (Shinn 1998, Bechtel SAIC 2006).

The 24-hr weighted air concentrations of PM-2.5 particles for FY 2011–2014 from the EPA’s Air Data web site were reviewed for more than 1000 counties in the U.S. The ambient average PM-2.5 air concentration ranged from 1×10^{-6} to 4.5×10^{-5} g/m^3, and the 98th percentile PM-2.5 air concentrations ranged from 5×10^{-6} to 1.7×10^{-4} g/m^3 (EPA 2014). Healy and Rodgers (1979) used 1×10^{-4} g/m^3 for predictive purposes and found that the predicted results and the real cases were comparable. The value of mass loading, 1×10^{-4} g/m^3, used for predictive purposes is more conservative compared to the average ambient concentrations observed and takes into account the enhanced activity concentration in resuspended particulates compared to the activity concentration in soil. The EPA (EPA 1977) has used the same value to screen calculations.

The mass loading value will fluctuate above its ambient level depending on human activities such as plowing and cultivating dry soil or driving on an unpaved road. The estimated mass loading for construction activities is about 6.0×10^{-4} g/m^3; for exposure to construction traffic on unpaved roads, it is 4.0×10^{-4} g/m^3; and for agriculture-generated dust, it is about 3.0×10^{-4} g/m^3 (Oztunali et al. 1981). The maximum respirable dust loading inside the cab of heavy construction equipment during a surface coal mining operation was found to be 1.8×10^{-3} g/m^3 (Oztunali et al. 1981). Estimates of mass loadings have been as high as 1.3 g/m^3 for instantaneous mass loadings during tilling.

3.6.3 Data Input Requirements

The mass loading for inhalation input to RESRAD (onsite) and RESRAD-OFFSITE provides the time-averaged respirable concentration of contaminated soil and dust. A mass loading value of 1.0×10^{-4} g/m^3 is the default value used in RESRAD (onsite) and RESRAD-OFFSITE. This value is a conservative estimate compared to the U.S. annual average ambient PM-10 air concentrations (Yu et al. 2000) and PM-2.5 air concentrations (Yu et al. 2007, EPA 2014); it takes into account short periods of high mass loading and sustained period of normal activity on a typical farm (Healy and Rodgers 1979).

4 RADON EMANATION PARAMETERS

4.1 EFFECTIVE RADON DIFFUSION COEFFICIENT

4.1.1 Definition

The random movement of the radon gas atoms mixed in the air results in a net migration of the radon gas toward the direction of its decreasing concentration in the air. This phenomenon is called molecular or atom diffusion. The diffusion of radon in open air can be described by Fick's law, which states that the flux density of the diffusing substance is linearly proportional to its concentration gradient. Fick's law can be expressed as follows:

$$J = - D_o \vec{\nabla} C \ , \tag{4.1.1}$$

where

J = a vector representing the flux density of radon activity in units of activity·$l^{-2}T^{-1}$,

$\vec{\nabla} C$ = a vector representing the gradient of radon activity concentration in the air in units of activity·l^{-4}, and

D_o = molecular (or atom) diffusivity or the diffusion coefficient of radon in open air in units of $l^2 \cdot T^{-1}$.

Therefore, the diffusion coefficient D_o can be defined from Fick's equation and expressed as the ratio of the magnitudes of the vectors J to $\vec{\nabla} C$:

$$D_o = \frac{|J|}{|\vec{\nabla} C|} . \tag{4.1.2}$$

For radon diffusion in open air, Fick's law is uniquely expressed and, consequently, the diffusion coefficient of radon in open air, D_o, is also uniquely defined. However, when applied to the conditions of radon diffusion in porous media, such as in soils, Fick's equation can be written in different ways, depending on how the variables flux density J and concentration C are defined. Fick's equation can be written in four distinct ways when applied to the molecular diffusion phenomenon in porous media, depending on whether the bulk or pore volume is used to define the concentration and whether the bulk or pore area is used to define the flux density. These different ways of defining the radon diffusion coefficient in soil lead to some confusion in selecting and using these parameters because the symbols and nomenclature used have not been standardized (Nazaroff et al. 1988).

Two distinct ways of defining the diffusion coefficient of radon in porous media have been adopted in the literature: (1) D_e is the effective radon diffusion coefficient and (2) D is the bulk radon diffusion coefficient. However, Culot (1976) and Nazaroff et al. (1988) have noted discrepancies regarding the way these two coefficients are defined and used in modeling the diffusion of radon through porous media. Therefore, the definitions of D_e and D adopted in this handbook are those suggested by Nazaroff et al. (1988).

Thus, the effective (or interstitial) radon diffusion coefficient, D_e, is defined from Fick's equation as the ratio of the diffusive flux density of radon activity across the pore area, J_e, to the gradient of the radon activity concentration in the pore or interstitial space, ∇C. This definition is equivalent to that relating the bulk flux density to the gradient of the bulk concentration of radon activity in the soil and can be expressed as follows:

$$D_e = \frac{|J_e|}{|\vec{\nabla} C|} \tag{4.1.3}$$

The bulk radon diffusion coefficient, D, is defined as the ratio of the diffusive flux density of radon activity across a geometric or superficial area of the medium, J_b, to the gradient of the radon activity concentration in the pore space, ∇C, and can be expressed as follows:

$$D = \frac{|J_b|}{|\vec{\nabla} C|} \tag{4.1.4}$$

The bulk and the effective radon diffusion coefficients in soil, D and D_e, respectively, are related by the total soil porosity, p_t, according to the following expression:

$$D = p_t D_e \,. \tag{4.1.5}$$

In general, the diffusion coefficient in porous media is a property of the diffusing species, the pore structure, the type of fluids present in the pores, the adsorption properties of the solid matrix, the fluid saturations, and temperature. For radon diffusion in porous media, the diffusivity for the other isotopes of radon (e.g., radon-220) has been observed to be comparable to that for the isotope radon-222 (Nazaroff et al. 1988).

Several attempts have been made to correlate the radon diffusion coefficients in porous media (D and D_e) to the radon diffusion coefficient in open air (D_o) and the physical properties of the medium such as the total porosity (p_t). These attempts have not been conclusive. According to experimental work performed by Currie (1960a,b) and quoted by Rolston (1986) and Nazaroff et al. (1988), the coefficients D and D_o can be correlated by an expression of the following form:

$$\frac{D}{D_o} = \gamma \, {p_t}^{\mu} \,, \tag{4.1.6}$$

where γ and μ represent measures of pore shape of the soil materials. This empirical relationship can fit data from a wide range of dry porous materials in which the values of γ generally lie between 0.8 and 1.0 and the values of μ lie around 1.0. This empirical relationship is not applicable, however, for very wet soil and strongly aggregated soil (Rolston 1986).

The influence of soil moisture content on the effective diffusion coefficient of radon in soil has been investigated by Rogers and Nielson (1991), who proposed the following empirical expression:

$$D_e = D_o\, p_t \exp(-6\, p_t\, R_s - 6R_s^{14\, p_t}) \tag{4.1.7}$$

where

D_o = radon diffusivity in open air (= 1.1×10^{-5} m^2/s),

p_t = total soil porosity, and

R_s = water saturation in the soil (or the fraction of the pore space filled with water, also called the saturation ratio).

With a specific p_t, Equation (4.1.7) gives a single value of D_e for a given water saturation ratio. However, the actual D_e could vary widely for different types of material, as seen in the original data (Rogers et al. 1984) used for fitting the equation.

4.1.2 Measurement Methodology

The diffusivity (or the diffusion coefficient) of radon in porous materials can be measured by both field and laboratory experiments. In either case, the experimental evaluation of the diffusivity consists in determining the numerical value of the coefficient appearing in Fick's equation. Because of the difficulties in implementing field methods, laboratory methods are generally used to determine the radon diffusivity in porous media and particularly in soil materials.

4.1.2.1 Laboratory Methods

Variations of the laboratory methods for measuring radon diffusivity in porous media have been developed, and as yet no standard (or recommended) method exists. All the various laboratory methods are based on the solution of the mass balance equation that represents the diffusion process in a one-dimensional configuration. Depending on the approximation taken on the time domain for the solution of the diffusion equation, these methods can be separated into two distinct groups: (1) the steady-state diffusion method and (2) the transient diffusion method (Nielson et al. 1982).

The steady-state method used in the laboratory for the determination of the radon diffusivity in a porous material without a source of radon within it is based on the solution of a one-dimensional diffusion equation in the x-direction, expressed as follows:

$$\frac{d^2C}{dx^2} - \frac{\lambda}{D_e} C = 0 \ . \tag{4.1.8}$$

This steady-state equation is obtained by coupling the one-dimensional Fick’s equation,

$$J_e = - D_e \frac{dC}{dx} \ , \tag{4.1.9}$$

with the one-dimensional, steady-state continuity equation,

$$\frac{d\,J_e}{dx} = - \lambda C \ , \tag{4.1.10}$$

where

J_e = effective flux density of radon activity (pCi)/(m^2 × s),

C = concentration of radon activity in the pore space (pCi/m^3), and

λ = radon decay constant (1/s).

A steady-state diffusion method for determining the effective radon diffusion coefficient (D_e) in uncontaminated (no radon source) porous materials was implemented by Silker and Kalkwarf (Silker 1981, Silker and Kalkwarf 1983) on the basis of theoretical developments by Cohen and Cohen (1979).The apparatus used in this method consists of a column of test material of known thickness, d, which is sealed at one end to an air chamber of known volume containing a radon source with a known and constant strength. The other end of the test column is kept open. As a boundary condition for this system, it is assumed that in a steady-state situation, the effective flux density of radon activity at the bottom of the column, J_{eo}, is constant and uniquely dependent on the strength of the radon source and the geometry of the system. Also, the radon activity concentration at the open end of the column is assumed to be negligible (i.e., zero).

On the basis of these assumptions and conditions, the effective radon diffusivity, D_e, can then be evaluated by the following equation (Silker and Kalkwarf 1983):

$$\lambda\, d \left(\frac{C_o}{J_{eo}} \right) = \left(\frac{d}{l} \right) \left[\frac{1 - e^{-2\frac{d}{l}}}{1 + e^{-2\frac{d}{l}}} \right] \ , \tag{4.1.11}$$

where

d = thickness of the porous material in the column,

C_o = radon activity concentration within the air chamber, and

l = radon diffusion length (or relaxation length) parameter within the porous material, which is defined as follows:

$$l = \sqrt{\frac{D_e}{\lambda}} \quad . \tag{4.1.12}$$

The right side of Equation (4.1.11) is a well-defined function of the parameter ratio d/l and is independent of the measured values of C_o and J_{eo}. The left side of the equation is dependent on the measured results. Therefore, by selecting the size (i.e., thickness) of the test sample, d; determining the effective flux density J_{eo} on the basis of the strength of the radon source and the column diameter; and making several measurements of C_o; Equation (4.1.11) can be graphically or numerically solved for the ratio d/l and subsequently for D_e.

Typically, for soils, the samples used in the determination of D_e have a cylindrical shape with a thickness of 10 cm and a diameter of 14 cm. After equilibration, the steady-state radon concentration in the bottom chamber, C_o, is determined by several measurements taken over a 7- to 14-day period. Each measurement consists of withdrawing about 5 cm^3 of gas from a typical 800-cm^3 bottom chamber and determining the radon concentration by using either a scintillation flask technique (such as a Lucas cell) or charcoal absorption and gamma-ray spectrometry (Silker and Kalkwarf 1983).

4.1.2.2 Field Methods

The most popular field (i.e., in situ) method for measuring radon diffusion coefficient is the radon depth profiling method (IAEA 2013, Sakoda et al. 2010a, Sahoo et al. 2010, Nazaroff et al. 1988). In this method, radon concentrations at different depths from the surface of a ground source are measured by inserting a probe equipped with a radon monitor to the source. The data on radon concentrations in pore-air at different depths, as measured by the probe, are least-square fitted to the following equation derived from diffusion theory (Nazaroff 1992, Nazaroff et al. 1988):

$$C(z) = C_\infty \left(1 - e^{-\frac{z}{l}}\right) \quad , \tag{4.1.13}$$

where

$C(z)$ = radon concentration in pore-air at depth z (Bq/m^3 or pCi/m^3),

C_∞ = radon concentration at a large depth of the ground source, typically >2 m for soil (Bq/m^3 or pCi/m^3),

z = depth (m), and

l = radon diffusion length (m).

The parameters C_∞ and l are obtained as fitting parameters. From the value of l, the effective diffusion coefficient, D_e, can be calculated as $D_e = \lambda l^2$.

Another field method that can be applied to obtain the radon effective diffusion coefficient involves the in situ accumulator technique. A radon accumulator (see IAEA 2013 for more information) consisting of a cylindrical or rectangular cup is placed on the surface of a ground source to be investigated. According to the prediction based on diffusion theory (Aldenkamp et al. 1992), the radon concentration in the headspace of the accumulator would increase approximately as a saturating exponential with time. By measuring the radon concentration in the headspace over time until a steady state is obtained and fitting the measured data to the theoretical equation, the value of a time constant, τ_e which is the effective time for radon buildup in the cup, can be determined (Aldenkamp et al. 1992). The time constant is related to the radon effective diffusion coefficient in the source matrix and the dimension of the accumulator. A look-up table (see Table 4 of IAEA 2013) can be used to obtain the effective diffusion coefficient, D_e, corresponding to the time constant, τ_e, determined from data fitting and information on the volume of the headspace and chamber radius of the accumulator.

4.1.3 Data Input Requirements

In RESRAD (onsite) and RESRAD-OFFSITE, the user is requested to input an effective diffusion coefficient value of radon for three materials: (1) the soil of the cover zone, (2) the soil of the contaminated zone, and (3) the building foundation material (i.e., concrete). These input values of D_e are in units of square meters per second (m^2/s). For each porous material considered, the value of D_e is assumed to be the same for both radon isotopes addressed in RESRAD, that is, radon-222 and radon-220.

The effective radon diffusivity values in porous media (soils and concrete included) vary over a range of several orders of magnitude, depending on the porous material and particularly on its degree of water saturation. Table 4.1.1 lists values of effective diffusion coefficients of radon obtained by different researchers for a range of unconsolidated soil materials, concrete, and other building materials. Because of the differences in the experimental methodologies adopted by the various researchers, these experimental data are not easily comparable. Nevertheless, they provide observed values of D_e in the field.

Typically, the effective diffusion coefficient of radon in unconsolidated soil material with a low moisture content is about 10^{-6} m^2/s. The upper limit is represented by the radon diffusion coefficient in open air, D_o, which is about 1.1×10^{-5} m^2/s. At the lower extreme, in a fully saturated soil material the radon diffusion coefficient may be as low as 10^{-10} m^2/s. In RESRAD (onsite) and RESRAD-OFFSITE, a default value of D_e equal to 2.0×10^{-6} m^{-2}/s was adopted for both the cover and contaminated zones. According to the data presented in Table 4.1.1, this default value of D_e would represent the average effective radon diffusion coefficient in soils with a low moisture content. The observed range of variation of D_e in uncracked concrete, as presented in Table 4.1.1, is from 8.0×10^{-9} to 5.2×10^{-7} m^2/s. A default value of D_e equal to 3.0×10^{-7} m^2/s was adopted in the RESRAD (onsite) and RESRAD-OFFSITE models to represent the effective radon diffusion coefficient in concrete. The presence of cracks can increase the effective diffusion coefficients outside of the range for intact concrete.

The values of the effective radon diffusion coefficient (D_e) to be used in RESRAD (onsite) and RESRAD-OFFSITE can be estimated at different levels of site-specific accuracy, depending on the amount of information available. For generic use of the code, a set of default values of D_e was defined as 2.0×10^{-6} m^2/s for the cover and contaminated zones and 3.0×10^{-7} m^2/s for the building foundation (i.e., concrete). If the type of soil materials at the site is known, a slightly more accurate estimation of D_e can be performed with the help of Table 4.1.1. For most applications, this approach will suffice because of the natural variability of D_e within the soil and building materials of any specific site.

In cases in which there are reasons to suspect that the default values of the effective radon diffusion coefficient (D_e) do not reflect the conditions at a specific site and there is no possibility of measuring D_e, the RESRAD (onsite) and RESRAD-OFFSITE codes are able to estimate it internally on the basis of the values of the water saturation (calculated from the volumetric water content) and total porosity, according to Equation (4.1.7). To implement this option, the user should enter any negative number as an input value of D_e to RESRAD (onsite) or RESRAD-OFFSITE.

For an accurate site-specific estimate of the input data to RESRAD (onsite) or RESRAD-OFFSITE, however, the values of D_e should be measured with either the laboratory or field methods. Whenever necessary and possible, the measurement of D_e in the soil cover zone (it is assumed that it is not contaminated with radon sources) should be performed in the laboratory by using a method such as the Silker and Kalkwarf (1983) technique.

4.2 RADON EMANATION COEFFICIENT

4.2.1 Definition

The radon emanation coefficient, ε, is the fraction of the total amount of radon produced by radium decay that escapes from the source particles and gets into the pores of the source matrix. It is also called the emanating power, emanating fraction, release ratio, and escape-to-production ratio. The radon emanation coefficient is a dimensionless parameter and is represented as either a fraction or a percentage.

TABLE 4.1.1 Observed Effective Diffusion Coefficients for Radon in Unconsolidated Soil Materials and Concrete[a]

Porous Material	Effective Radon Diffusion Coefficient, D_e (m^2s^{-1})	Comment	Reference
Unconsolidated soil material			
Compacted silty sands	$(3.0 \pm 1.3) \times 10^{-6}$	$p_t = 0.29–0.36$ $R_s = 0.05–0.34$	Silker and Kalkwarf (1983)
Compacted clayey sands	$(3.2 \pm 1.5) \times 10^{-6}$	$p_t = 0.32–0.39$ $R_s = 0.09–0.55$	Silker and Kalkwarf (1983)
Compacted inorganic clays	$(2.5 \pm 1.0) \times 10^{-6}$	$p_t = 0.32–0.43$ $R_s = 0.06–0.34$	Silker and Kalkwarf (1983)
Silty sandy clay	2.7×10^{-6}	$w = 1.5\%$ dry weight	Strong et al. (1981)
	2.5×10^{-7}	$w = 10.5\%$ dry weight	
	6.0×10^{-8}	$w = 17.3\%$ dry weight	
Uranium mill tailings	$(5.4 – 7.2) \times 10^{-6}$	$w = (0.7–1.5)\%$ dry weight	Strong et al. (1981)
Loams	8×10^{-7}	Dry	Tanner (1964)
Mud	5.7×10^{-10}	$\theta = 37\%$	Tanner (1964)
Concrete	$(1.1 – 4.0) \times 10^{-7}$	$p_t = 0.11–0.13$	Poffijn et al. (1988)
	$2.1 \times 10^{-8}–5.2 \times 10^{-7}$	$p_t = 0.13–0.27$, air-dry	Rogers et al. (1994)
	1.2×10^{-8}	$p_t = 0.25$	Culot (1976)
	3.4×10^{-8}	$p_t = 0.05$	Culot (1976)
	3.3×10^{-8}	$p_t = 0.068$	Zapalac (1983), Stranden et al. (1984)
	$(0.8 – 9.3) \times 10^{-8}$	-	Stranden (1988), Cosma et al. (2001)
Other materials			
Brick	$(0.8 – 5.0) \times 10^{-7}$	-	Stranden (1988), Keller et al. (2001)
Gypsum	$(1.0 – 4.0) \times 10^{-6}$	-	Stranden (1988), Keller et al. (2001)

[a] p_t = total porosity, R_s = volumetric water saturation, w = percent water content by weight, and θ = percent volumetric water content.

Source: Adapted from Nazaroff et al. (1988), with the addition of data reported after 1988.

The two most common radioisotopes of radon gas, radon-222 and radon-220, are generated by a radioactive process of alpha decay from two radium isotopes, radium-226 and radium-224, respectively. Because of the conservation of linear momentum in the alpha-decay process, the newly created radon-222 and radon-220 atoms are left with a kinetic (usually called "recoil") energy of about 86 and 103 keV, respectively (Nazaroff et al. 1988). Thus, after being generated, the radon atoms tend to move away from their original location until their recoil energy is totally transferred to the surrounding medium. Consequently, depending on their original location within the solid phase of the matrix, the pore space distribution within the matrix, and the moisture content of the matrix, the newly created radon atoms may end up within the same solid particle in which they were created, within the adjacent solid particle because of posterior penetration after escaping from the host solid grain, or within the pore of the matrix.

Experimental data reported by several investigators indicate that the radon emanation coefficient for a specific type of source material is strongly influenced by the moisture content of the source matrix, particularly within the range of low water saturation (Strong and Levins 1982, Rogers et al. 1983, Barton and Ziemer 1986, Nazaroff et al. 1988, Breitner et al. 2010). On the basis of results of this kind, it has been hypothesized that the amount of water present in the pore increases the absorption of the recoil energy of the radon atom passing through it, thus enhancing the chance that the atom will terminate its recoil within the water. Partition equilibrium of radon in the water and air phases in the pore will follow afterwards on the basis of Henry's law.

Although temperature may influence the magnitude of the radon emanation coefficient, it has been demonstrated that within the normal range of temperature variability in surface soils, this effect is of minor importance (Nazaroff et al. 1988). The other factors that influence the radon emanation coefficients include porosity, particle size and shape, and elemental composition of the source material. Weathering may also influence the characteristics of the source materials, thereby affecting the value of the emanation coefficients (IAEA 2013). However, overall, the influence of these parameters is less than that of moisture content. Therefore, in Section 4.2.2, which summarizes measurement data reported in the literature, the variation with respect to moisture content is documented whenever the information is available.

The radon emanation coefficient, ε, is one of the characteristic source parameters that determine the rate of radon emanation into the pores of the source matrix. The other source characteristic parameter in relation to radon production is the concentration of radium (radium-226 and/or radium-224) in the source particles, S_{Ra}. In RESRAD (onsite) and RESRAD-OFFSITE, the source of radon generation in the pore air or the rate of radon generation and emanation into the matrix gas phase (pore air), $\dot{S}$, in units of $\text{pCi/m}^3\cdot\text{s}$, is calculated as follows:

$$\dot{S} = \varepsilon \, \rho_s \, S_{Ra} \, \lambda \left(\frac{1 - p_t}{p_t} \right) , \tag{4.2.1}$$

where

ε = radon emanation coefficient (dimensionless),

ρ_s = source particle density (kg/m^3) in the matrix,

S_{Ra} = mass concentration of radium (radium-226 or radium-224*) in the source particles (pCi/kg),

λ = radon (radon-222 or radon-220) decay constant (1/s), and

p_t = the total porosity of the matrix.

4.2.2 Summary of Literature Data

The values of the radon emanation coefficient for source materials depend on the radon isotope being considered, the source material, and the moisture content of the source matrix. Experimental measurements of ε in different soils, rocks, and other materials have been reported by many investigators. Because of the differences in the experimental methodologies adopted by the various investigators, these data are not easily comparable. Most of the reported data are for radon-222.

Although the data for radon-220 are incomplete in that they do not cover a broad range of source materials under different degrees of water saturation, the summary by Ishimori et al. (IAEA 2013) as listed in Table 4.2.1 may give an indication of the expected values of $\in$ for radon-220 in the field.

Table 4.2.2 summarizes the mean values and ranges of ε for radon-222 in minerals, rocks, soils, mill tailings, and fly ashes. They were determined by Sakoda et al. (2011) after reviewing and analyzing a large number of literature data. Detailed listings of the source data along with information on the moisture content for each type of source material are provided in the subsequent tables. These detailed listings are thought useful for narrowing the range of ε to obtain a refined value if more source-specific information is available. Table 4.2.3 lists the values for various types of minerals. Table 4.2.4 lists the values for various types of rock. Table 4.2.5 lists the values for various types of soil from different countries. The values listed in Table 4.2.5 across different countries were further sorted and analyzed to obtain the means and ranges for 6 commonly seen soil types—sand, clay, silt, sandy loam, silty loam, and gravel (listed in Table 4.2.6). These means and ranges were compared with those summarized in the previous data collection effort (see Table 8.1 of Yu et al. 1993), and they were found to be comparable to each other. Because data for heavy loam soils are not explicitly included in Table 4.2.5, those obtained from the previous effort were retained and included in Table 4.2.6. Table 4.2.7 lists the values for various uranium mill tailings, while Table 4.2.8 lists the values for fly ashes.

---

* In the RESRAD (onsite) codes, the radium-224 (half-life less than one-half year) may be considered as an associated radionuclide, depending on the cutoff time selected by the user. In that case, its principal parent radionuclide, thorium-228, is used as the source for radon-220.

TABLE 4.2.1 Summary of Radon-220 Emanation Coefficients for Various Source Materials

Material		Number of Samples	Mean Radon-220 Emanation Coefficient	Reference
Mineral	Reported data	2	0.0005 ± 0.0001	Rama and Moore 1984
		11	0.011 ± 0.016	Rama and Moore 1990
	Average	13	0.01 ± 0.02	
Rock	Reported data	5	0.157 ± 0.086	Rama and Moore 1984
		32	0.105 ± 0.145	Howard et al. 1995
	Average	37	0.11 ± 0.14	
Soil	Reported data	12	0.11 ± 0.02	Megumi and Mamuro 1974
		16	0.16 ± 0.06	Greeman and Rose 1996
	Average	28	0.14 ± 0.05	

Note: Moisture content may differ among the samples, although test temperature was around room temperature in all cases.

Source: Taken from Table 3 of IAEA 2013.

TABLE 4.2.2 Summary of Radon-222 Emanation Coefficients for Various Source Materials

Type of Material	Number of Samples	Mean and Range of Radon-222 Emanation Coefficient
Mineral	75	0.03 (<0.005–0.25)
Rock	113	0.13 (<0.005–0.40)
Soil	1,025	0.20 (<0.005–0.83)
Mill tailing[a]	77	0.17 (0.04–0.35)
Fly ash	46	0.03 (<0.005–0.35)

[a] Mostly uranium mill tailings.

Note: Moisture content differs among the samples. When the radon emanation coefficients of the same sample were measured under conditions of changing moisture content, the maximum value was used to calculate the mean radon emanation coefficient.

Source: Sakoda et al. 2011.

4.2.3 Measurement Methodology

One method of measuring the radon emanation coefficient (ε) of a porous material contaminated with radium consists basically of measuring the radon concentration in the air within a sealed accumulation chamber in which a sample of the source material has been left for an extended period of time (4 days to 4 weeks, depending on the source material) to ensure the establishment of radioactive equilibrium between radon and radium. After that, the total activity of radon released into the air from the sample material is evaluated by a measurement of radon concentration and the effective volume of the measurement system. The total activity of radium in the sample material is determined by various methods such as alpha spectrometry, gamma spectrometry, liquid scintillation spectrometry or mass spectrometry (IAEA 2010b). Either a closed-loop or a flow-through approach can be used to monitor the radon concentration in the air, as shown in Figures 4.2.1 and 4.2.2. Detailed descriptions of the measurement setup are given by Ishmori et al. (IAEA 2013).

TABLE 4.2.3 Radon-222 Emanation Coefficients for Various Types of Minerals

Type of Mineral	Moisture Content	Radon-222 Emanation Coefficient	References
Allanite	–	0.003	Barretto 1973
Apatite	Dry, –	0.005, 0.008	Barretto 1973, Rama and Moore 1990
Biotite	–	0.028–0.066	Barretto 1973
Cerite	Saturated	0.0168	Garver and Baskaran 2004
Cleavelandite	–	0.0406	Krishnaswami and Seidemann 1988
Glauconite	–	0.030	Barretto 1973
Hornblrnde	–	0.0342	Krishnaswami and Seidemann 1988
Magnetite	–	0.040, 0.18	Barretto 1973, Rama and Moore 1984
Monazite	Dry, –	0.0002–0.15	Barretto 1973, Rama and Moore 1984, 1990, Garver and Baskaran 2004
Muscovite	0	< 0.015	Sakoda et al. 2010a,b
Orthoclase	–	0.105	Krishnaswami and Seidemann 1988
Plagioclase	–	0.0185	Krishnaswami and Seidemann 1988
Quartz	0	0.046	Sakoda et al. 2010a,b
Sphene	–	0.002–0.047	Barretto 1973
Thorite	Dry, saturated	0.0538, 0.149	Garver and Baskaran 2004
Uraninite	Dry	0.0053, 0.019	Garver and Baskaran 2004, Rama and Moore 1990
Xenotime	–	0.0009	Barretto 1973
Zircon	Dry, –	0.0001–0.121	Barretto 1973, Rama and Moore 1990, Garver and Baskaran 2004

Note: a "–" means moisture content not specified.

Source: Obtained by grouping data presented in Table A1 of Sakoda et al. (2011) by mineral type.

TABLE 4.2.4 Radon-222 Emanation Coefficients for Various Types of Rocks

Type of Rock	Moisture Content	Radon-222 Emanation Coefficient	References
Alaskite	–	0.154	Barretto 1973
Alum shale	0–0.21	0.011–0.35	Schery et al. 1984
Argillite	Saturated	0.04–0.15	Ferry et al. 2002
Asphalt limestone	–	0.019	Barretto 1973
Basalt	–	0.00143–0.043	Abumurad and Al-Tamimi 2001, Barretto 1973
Biotite granite	Dry	0.31	Funtua et al. 1997
Conglomerate	–	0.023–0.104	Barretto 1973
Dacite	–	0.069	Barretto 1973
Diabase	–	0.094	Barretto 1973
Dol. subarkose	–	0.056	Barretto 1973
Gabbro	–	0.036	Barretto 1973
Gneiss	–	0.010–0.140	Barretto 1973
Granite	–	0.0145–0.404	Barretto 1973, Holub and Brady 1981, Krishnaswami and Seidemann 1988
Granite gneiss	0	0.021–0.026	Sakoda et al. 2008, Sakoda et al. 2010a,b
Granite pebble	–	0.11–0.15	Rama and Moore 1984
Granite saprolite	Dry, saturated	0.025–0.18	Rama and Moore 1984
Granodiorite	–	0.183 (0.039–0.400)	Barretto 1973
Laterite	–	0.0252–0.0286	Barretto 1973
Limestone	–	0.018 (0.016–0.022)	Barretto 1973
Lugaurite	–	0.154	Barretto 1973
Metaconglomerate	–	0.26	Barretto 1973
Limestone	–	0.006	Barretto 1973
Pegmatite	–	0.030–0.043	Barretto 1973
Phosphate	–	0.13	Lysandrou et al. 2007
Phosphorite	–	0.14	López-Coto et al. 2009
Qtz. diorite	–	0.047–0.079	Barretto 1973
Qtz. monzonite	–	0.092	Barretto 1973
Quartzite	–	0.059 (0.0109–0.105)	Barretto 1973
Red silt	–	0.081	Barretto 1973
Rhyolite	Dry	0.22	Funtua et al. 1997
Sandstone	–	0.07 (0.032–0.119)	Barretto 1973
Serpentinite	–	0.010	Barretto 1973
Shale	–	0.0267	Barretto 1973
Silt shale	–	0.077	Barretto 1973
Syenite	–	0.093	Barretto 1973
Volcanic glass	–	0.0051	Barretto 1973
Volcanic tuff	Dry,saturated	0.014–0.26	Barretto 1973, Schery et al. 1982, Strong and Levins 1982

Note: a "–" means moisture content not specified.

Source: Obtained by grouping data presented in Table A2 of Sakoda et al. (2011) by rock type.

TABLE 4.2.5 Radon-222 Emanation Coefficients for Various Types of Soil

Country	Type of Soil	Number of Samples	Moisture Content (volume fraction)	Radon-222 Emanation Coefficient	References
Austria	Sand (quartz)	16	–	0.074 (0.016–0.276)	Bossew 2003
Denmark	Glacial deposit – moraine clay	12	0.03–0.22 wt	0.11–0.39	Damkjær and Korsbech 1985
	Glacial deposit – moraine sand	3	0.04–0.13 wt	0.09–0.25	Damkjær and Korsbech 1985
	Glacial deposit – diluvial clay	8	0.15–0.27 wt	0.12–0.27	Damkjær and Korsbech 1985
	Glacial deposit – diluvial sand, silt & gravel	13	0–0.24 wt	0.09–0.32	Damkjær and Korsbech 1985
	Non-glacial deposit – tertiary morine clay	15	0.13–0.50 wt	0.10–0.61	Damkjær and Korsbech 1985
	Non-glacial deposit – tertiary morine diatomite	6	0.29–0.48 wt	0.14–0.39	Damkjær and Korsbech 1985
	Non-glacial deposit – limestone and chalk	4	0–0.16 wt	0.02–0.17	Damkjær and Korsbech 1985
	Non-glacial deposit – miscellaneous	7	0–0.70 wt	0.03–0.83	Damkjær and Korsbech 1985
Finland	Clay	56	Small amount	0.24	Markkanen and Arvela 1992
	Silt	67	Small amount	0.18	Markkanen and Arvela 1992
	Sand	170	Small amount	0.17	Markkanen and Arvela 1992
	Gravel	175	Small amount	0.19	Markkanen and Arvela 1992
	Till	231	Small amount	0.20	Markkanen and Arvela 1992
	Esker sediment, at a depth of 100–120 cm from the soil surface	–	0	0.19	Breitner et al. 2008
Spain	Stony clay	–	Dry	0.023–0.029	Baixeras et al. 2001
Sweden	Clay	–	Dry	0.139–0.221	Baixeras et al. 2001
	Sand	–	Dry	0.002–0.008	Baixeras et al. 2001
USA	Lava fields	–	–	0.02	Wilkening 1974
	Thin organic soils	–	–	0.55	Wilkening 1974
	Volcanic ash soils (deep agricultural soils)	–	–	0.7	Wilkening 1974
	Sand (mainly quartz)	–	–	0.032	Rama and Moore 1984
	Gravelly sandy loam	–	0.04 wt	0.38	Schery et al. 1984
	Silt loam	–	Moist	0.27	Greeman and Rose 1996

TABLE 4.2.5 (Cont.)

Country	Type of Soil	Number of Samples	Moisture Content	Radon-222 Emanation Coefficient	References
	Alluvium	–	–	0.17	Barretto 1973
	Calcareous	–	–	0.37 (0.24–0.55)	Barretto 1973
	Clayey	–	–	0.28 (0.19–0.36)	Barretto 1973
	Eluvium	–	–	0.20–0.38	Barretto 1973
	Glauconitic	2	–	0.03	Barretto 1973
	Granitic	–	–	0.37–0.46	Barretto 1973
	Sandy (dune and loess)	–	–	0.09–0.19	Barretto 1973
	Lignitic	–	–	0.36	Barretto 1973
	Residual (Baux)	–	–	0.14–0.30	Barretto 1973
	Residual (Calc.)	–	–	0.52–0.54	Barretto 1973
	Volcanic	–	–	0.49	Barretto 1973
	Sand	–	Dry	0.062–0.071	van der Lugt and Scholten 1985
	Gravel	–	Dry	0.111	van der Lugt and Scholten 1985
	Sand	–	0–1 wt	0.13–0.28	van der Spoel et al. 1997, 1999
	Uraniferous soil	–	–	0.25	Gutiérrez et al. 2004

Note: a “–” means number of samples unknown or moisture content not specified.

Source: Adapted from Table A3 of Sakoda et al. (2011).

TABLE 4.2.6 Radon-222 Emanation Coefficients for Seven Common Types of Soil

Soil Type	Moisture Content	Radon-222 Emanation Coefficient	References
Sand	Dry	0.002–0.071	Baixeras et al. 2001, van der Lugt and Scholten 1985
	0.04–0.13 wt	0.09–0.28	Damkjær and Korsbech 1985, Markkanen and Arvela 1992, Van der Spoel et al. 1997, 1999
	–	0.016–0.276	Bossew 2003, Rama and Moore 1984, Barretto 1973
Clay	Dry	0.023–0.221	Baixeras et al. 2001
	0.03–0.27 wt	0.11–0.39	Damkjær and Korsbech 1985, Markkanen and Arvela 1992, Van der Spoel et al. 1997, 1999
	0.13–0.50 wt	0.10–0.61	Damkjær and Korsbech 1985, Markkanen and Arvela 1992, Van der Spoel et al. 1997, 1999
Silt	Small amount	0.18	Markkanen and Arvela 1992
Sandy loam	0.04 wt	0.38	Schery et al. 1984
Silty loam	Moist	0.27	Greeman and Rose 1996
Heavy loam	–	0.17–0.23	Sisigina 1974
Gravel	Dry, small amount	0.111–0.19	Van der Lugt and Scholten 1985, Markkanen and Arvela 1992
	0–0.24 wt	0.09–0.32	Damkjær and Korsbech 1985

Note: a "–" means moisture content not specified.

Using a closed-loop approach, the radon emanation coefficient is calculated with the following equation:

$$\varepsilon = \frac{V \times C}{M \times R} \quad , \tag{4.2.2}$$

where

ε = emanation coefficient,

V = effective volume of the sampling device (m^3),

C = radon concentration in the air (pCi/m^3 or Bq/m^3),

M = total mass of the sample (kg), and

R = radium activity concentration in the sample (pCi/kg or Bq/kg).

If a flow-through approach is used, the calculation of the radon emanation coefficient needs to take into account the flow rate of air:

$$\varepsilon = \frac{(v+\lambda V)\,C}{\lambda\,M\,R}\ , \qquad (4.2.3)$$

where

v = flow rate of air (m^3/s), and

λ = decay constant of radon (s^{-1}).

Another useful nondestructive and indirect method of measuring the radon-222 emanation coefficient involves measuring gamma rays emitted from Pb-214 or Bi-214, progenies generated as radon-222 decays, in the sample. Figure 4.2.3 is a schematic diagram of the method.

TABLE 4.2.7 Radon-222 Emanation Coefficients for Various Types of Uranium Mill Tailings

Type of Tailing	Moisture Content	Radon-222 Emanation Coefficient	Reference
–	0.005 vol.	0.067–0.072	Strong and Levins 1982
–	0.136 vol.	0.264	Strong and Levins 1982
–	0.460 vol.	0.291–0.31	Strong and Levins 1982
–	–	0.153	White and Rood 2001
Mixed (acid-, alkaline-, and carbonate-leached)	0.015 wt.	0.14	Landa 1987a
Acid-leached	0.015 wt.	0.21	Landa 1987a
Acid-leached	0.015 wt.	0.094	Landa 1987a
Acid-leached	0–0.37 wt.	0.08–0.33	Barillon et al. 2005
Carbonate-leached	0.015 wt.	0.21	Landa 1987a
Uraninite – acid-leached	0.009	0.05–0.07	Landa 1987b
Uraninite – alkaline-leached	0.009	0.11–0.12	Landa 1987b
Autunite – acid-leached	0.9	0.02–0.05	Landa 1987b
Autunite – alkaline-leached	0.9	0.12	Landa 1987b
Carnotite – acid-leached	0.9	0.04–0.07	Landa 1987b
Carnotite – alkaline-leached	0.9	0.04–0.06	Landa 1987b

Source: Adapted from Table A4 of Sakoda et al. 2011.

Note: a "–" means moisture content not specified.

TABLE 4.2.8 Radon-222 Emanation Coefficients for Fly Ashes

Moisture Content	Radon-222 Emanation Coefficient	Reference
–	0.0052–0.032	Maraziotis 1987
Dry	< 0.01	Stoulos et al. 2004, Van der Lugt and Scholten 1985
< 0.0012 wt	0.007–0.098	Kalkwarf et al. 1985
0–0.40 wt	0.001–0.154	Barton and Ziemer 1986

Note: a "–" means moisture content not specified.

Source: Adapted from Table A5 of Sakoda et al. (2011).

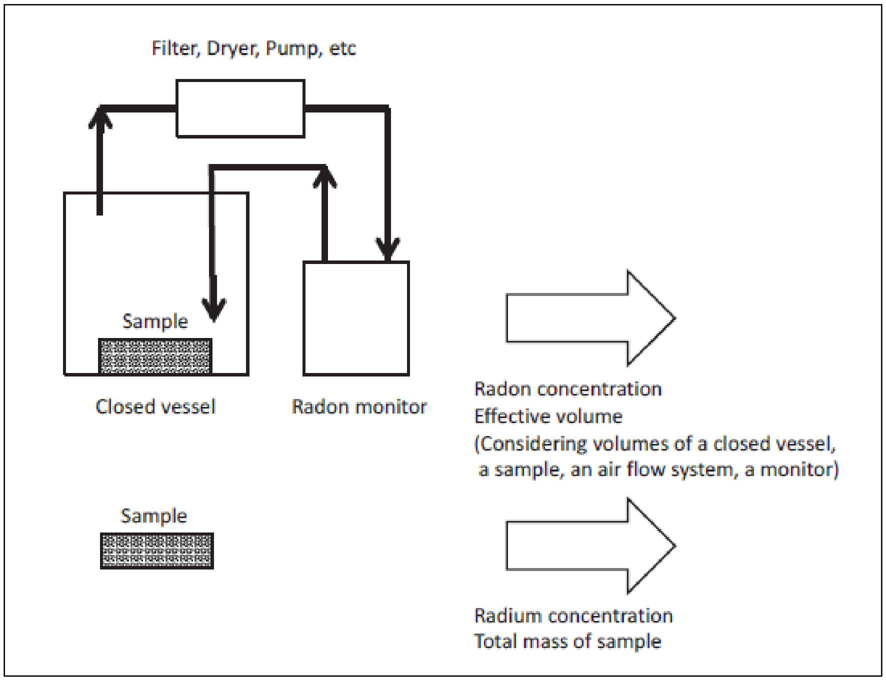


FIGURE 4.2.1 Schematic Presentation of a Closed-Loop Approach to Radon Emanation Coefficient Measurement (Source: Figure 10 of Technical Reports Series No. 474, *Measurement and Calculation of Radon Releases from NORM Residues*, ©IAEA, 2013, page 17)

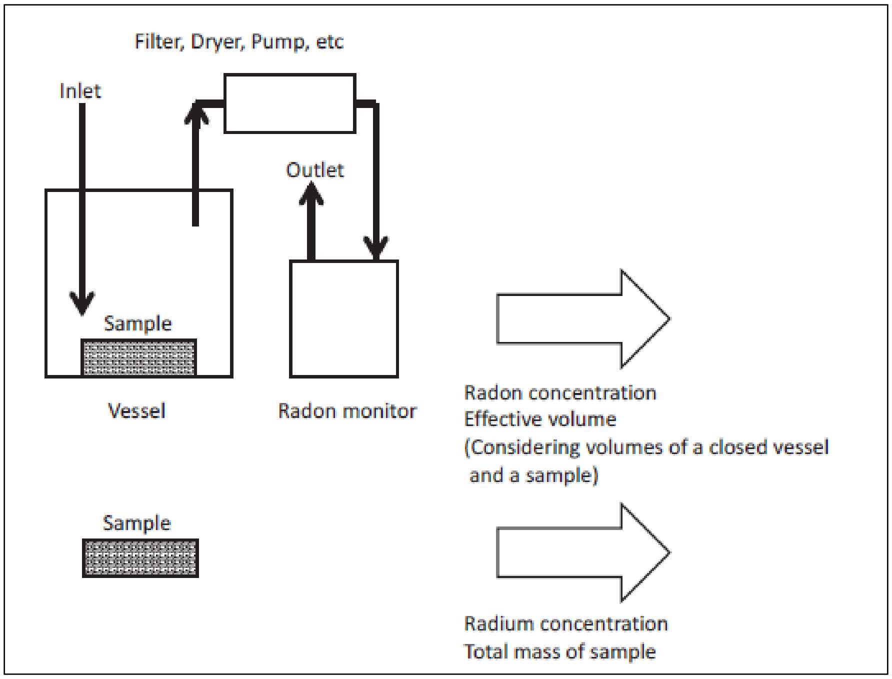


FIGURE 4.2.2 Schematic Presentation of a Flow-Through Approach to Radon Emanation Coefficient Measurement (Source: Figure 11 of Technical Reports Series No. 474, *Measurement and Calculation of Radon Releases from NORM Residues*, ©IAEA, 2013, page 18)

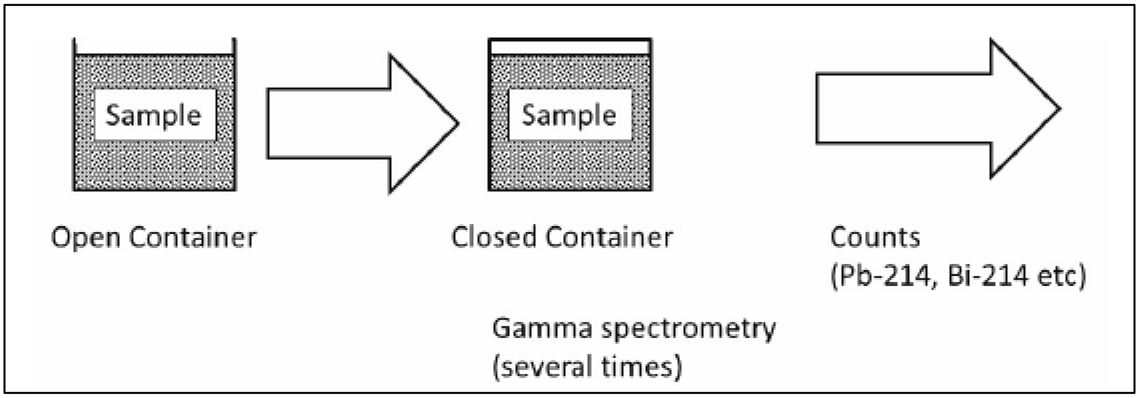


FIGURE 4.2.3 Schematic Presentation of a Gamma Spectrometry Method of Radon Emanation Coefficient Measurement (Source: Figure 12 of Technical Reports Series No. 474, *Measurement and Calculation of Radon Releases from NORM Residues*, ©IAEA, 2013, page 19)

With this method, calibration of the equipment may not be required, because only relative counts are needed (IAEA 2013). Assuming that radon activities are always equal to those of the radon progeny, the following equation is used to obtain the radon-222 emanation coefficient –

$$\varepsilon = \frac{N_{eq} - N_0}{N_{eq}} \;, \tag{4.2.4}$$

where

N_{eq} = specific counts selected from peaks of Pb-214 and Bi-214 in the equilibrium condition, and

N_0 = number of counts corresponding to N_{eq} in the initial condition.

More discussions on improving the precision of measurement can be found in IAEA 2013.

4.2.4 Data Input Requirements

In RESRAD (onsite) and RESRAD-OFFSITE, the user is requested to input a value for the radon emanation coefficient (ε) that is related to the source material in the contaminated zone for the two radon isotopes, radon-222 and radon-220. This parameter is dimensionless and its value should be entered as a fraction (rather than as a percentage).

As shown in Tables 4.2.1 and 4.2.2, the radon emanation coefficient is different for different materials. The values of ε for radon-222 are usually higher than those for radon-220 under the same circumstances. In RESRAD (onsite) and RESRAD-OFFSITE, the adopted default values of the radon emanation coefficient (ε) for the radon isotopes radon-222 and radon-220 are 0.25 and 0.15, respectively, selected on the basis of soils in the contaminated zone. As found in many laboratory measurements, the value of ε is dependent on the source material and the moisture content in the source matrix. If the type of source materials and the moisture content, which may not remain constant, in the source matrix are known, a more accurate estimation of ε can be made with the help of the more detailed listings of ε in Tables 4.2.3 to 4.2.8.

For estimating a radon emanation coefficient, in addition to Tables 4.2.1–4.2.8, an empirical equation (Zhuo et al. 2006) that relates the radon emanation coefficient under the influence of moisture in the source matrix to that under the dry condition may be useful as well:

$$\varepsilon = \varepsilon_0 \{ 1 + 1.85 \times [1 - exp(-18.8 R_s)]\} \;, \tag{4.2.5}$$

where

ε = radon emanation coefficient under a specific moist condition with a saturation ratio of R_s,

ε_0 = radon emanation coefficient under the dry condition, and

R_s = water saturation in the soil (or the fraction of the pore space filled with water, also called the saturation ratio).

In cases in which it is absolutely necessary to have an accurate estimate of ε and there are reasons to suspect that the data in Tables 4.2.1–4.2.8 or the value estimated with Equation (4.2.5) does not reflect the conditions at a specific site, the values of ε for radon-222 and radon-220 can be determined experimentally in the laboratory using one of the methods mentioned in Section 4.2.3.

4.3 RADON VERTICAL DIMENSION OF MIXING

4.3.1 Definition

The radon vertical dimension of mixing is the height of the atmospheric boundary layer near the ground surface into which the radon gas that emanates from the ground is uniformly mixed in the outdoor air. This parameter is measured in units of length (l).

4.3.2 Data Input Requirements

In RESRAD (onsite) and RESRAD-OFFSITE, the user is required to input a value for the radon vertical dimension of mixing that represents conditions at the site. This parameter’s dimension should be given in units of meters (m).

A default value of 2 m was adopted in the RESRAD (onsite) and RESRAD-OFFSITE model for the radon vertical dimension of mixing. This value of 2 m is a conservative value, considering the height of humans. Unless other evidence indicates use of a different value, this value should be used in the RESRAD (onsite) and RESRAD-OFFSITE codes.

This page intentionally left blank.

5 BUILDING CHARACTERISTIC PARAMETERS

5.1 AVERAGE BUILDING AIR EXCHANGE RATE

5.1.1 Definition

The building air exchange (or ventilation) rate is the total volume of air contained in the building that is being exchanged with outside air per unit of time. This parameter expresses the rate at which the total air contained within the building is replaced (or renewed) per unit of time and is measured in units of inverse time (T^{-1}). For example, a building with a ventilation rate of 1 h^{-1} has its volume of air replaced once each hour on average.

Important factors affecting the air exchange rate include construction and operating features of the building (i.e., age, window and door weatherproofing, existence of unbalanced mechanical ventilation, the use of fireplaces, etc.), as well as environmental conditions (i.e., atmospheric pressure, temperature, and wind speed and direction). The total air exchange rate is based on three factors (Nero 1988, Koontz and Rector 1995): (1) infiltration—the airflows through small openings and imperfections in the building structure; (2) natural ventilation—the exchange of air through windows, doors, or any other large openings that are kept partially or temporarily open; and (3) forced ventilation—the mechanically controlled airflows due to the operation of exhaust fans or other similar systems. Each of these factors varies significantly with time and, consequently, the total air exchange rate in a house is also strongly time dependent. In residences, although natural ventilation and forced ventilation contribute at times to the air exchange, infiltration is the dominant mechanism. The use of natural and forced ventilation is mostly seasonal. For larger, nonresidential buildings, forced ventilation is the dominant mechanism. In the United States, the air exchange rates for various housing types generally fall between 0.1 and 3.0 h^{-1}, with most clustered in the range of 0.25–0.75 (Grimsrud et al. 1983).

5.1.2 Summary of Literature Data

Koontz and Rector (1995) analyzed the data on air exchange rate measured with the perfluorocarbon technique (see discussions in Section 5.1.3) for residences in different regions, and derived a median value of 0.45 and a 10th-percentile value of 0.18 for all regions combined. The EPA (2011) recommended using the median value for obtaining a central estimate of potential exposure when evaluating airborne hazardous substances inside a house because the average was skewed by the upper end of the distribution. For a conservative estimate of potential exposure, the 10th-percentile value was recommended. EPA (2011) Section 19.5.1.1.1 presents distributions for the various regions of the country.

For nonresidential buildings (e.g., office space and malls) the building air exchange rates recommended by the EPA (2011) are 1.5 as the mean value and 0.6 as the 10th-percentile value, on the basis of data from Turk et al. (1987).

Tables 5.1.1 and 5.1.2 provide summary statistics on air exchange rates for residential buildings in different regions and for nonresidential buildings of different types, respectively, taken from EPA (2011). Because the data from which these distribution characteristics were derived were from short-term measurements, relying on assumptions that were not easily achieved during measurements, and on residences and buildings that were not selected randomly, the EPA's confidence ratings for its recommendations based on the mean/median and 10th-percentile values were low (EPA 2011). However, these distribution characteristics are believed to be the best information available to represent the air exchange rates for residential and nonresidential buildings across the United States throughout the year.

TABLE 5.1.1 Summary Statistics on Air Exchange Rates (1/hr) for Residential Buildings in Different Regions

Statistical Parameter	West Region	Midwest Region	Northeast Region	South Region	All Regions
Arithmetic Mean	0.66	0.57	0.71	0.61	0.63
Arithmetic Standard Deviation	0.87	0.63	0.6	0.51	0.65
Geometric Mean	0.47	0.39	0.54	0.46	0.46
Geometric Standard Deviation	2.11	2.36	2.14	2.28	2.25
10th Percentile	0.2	0.16	0.23	0.16	0.18
50th Percentile	0.43	0.35	0.49	0.49	0.45
90th Percentile	1.25	1.49	1.33	1.21	1.26
Maximum	23.32	4.52	5.49	3.44	23.32

Adapted from Table 19–24 of EPA (2011). Source of air exchange rate data: Koontz and Rector (1995).

TABLE 5.1.2 Air Exchange Rates (1/hr) for Different Types of Nonresidential Buildings

Building Type	Number of Observations	Mean	Standard Deviation	10th Percentile	Range
Educational	7	1.9			0.8–3.0
Office (<100,000 ft^2)	8	1.5			0.3–4.1
Office (>100,000 ft^2)	14	1.8			0.7–3.6
Library	3	0.6			0.3–1.0
Multi-use	5	1.4			0.6–1.9
Naturally ventilated	3	0.8			0.6–0.9
Total (all buildings)	40	1.5	0.87	0.6	0.3–4.1

Adapted from Table 19-27 of EPA (2011). Source of air exchange rate data: Turk et al. (1987).

5.1.3 Measurement Methodology

Two basic avenues exist for measuring the air exchange rate: (1) pressurization testing to measure the effective leakage area of the building, coupled with modeling to calculate air exchange rate, and (2) tracer-gas technique to monitor the dilution effects of air exchange. While modeling methods exist to relate the effective leakage rate to air exchange rate, the measured effective leakage rate is often used with predicated results obtained on the basis of empirical assumptions (e.g., physical distribution of leakage area on the building surface) and meteorological considerations. The tracer-gas techniques, on the other hand, provide a more direct measure of air exchange rate.

The tracer-gas techniques are generally based on mass-balance considerations, with explicit assumptions that (1) well-mixed conditions prevail and (2) outdoor concentrations are negligible; they take the form (Koontz and Rector 1995)

$$\frac{dC(t)}{dt} = \frac{S(t)}{V} - \frac{Q}{V} C(t) \tag{5.1.1}$$

where

$C(t)$ = tracer-gas concentration at time t (mg/m^3),

$S(t)$ = tracer-gas release rate at time t (mg/h),

V = volume of tested airspace (m^3), and

Q = s the exiting airflow rate at time t (m^3/h).

The ratio of exiting airflow (Q) to indoor volume (V), that is, Q/V, gives the air exchange rate I (h^{-1}).

Two tracer-gas techniques are often used to measure air exchange: concentration decay and constant injection. With the first technique, the measurement of the air exchange rate in a building is accomplished by injecting a tracer gas e.g., sulfur hexafluoride, SF$_6$) into the building and then, after a mixing time, measuring the gas concentration as a function of time by using an infrared analyzer (Nero 1988). The gas concentration, $C(t)$, is related to the air exchange rate, I, by the analytical solution to Equation (5.1.1):

$$C(t_2) = C(t_1)\, e^{-I(t_2-t_1)} \;\; or \;\;\; I = \frac{ln\,[C(t_2)/C(t_1)]}{t_2-t_1}\ . \tag{5.1.2}$$

The constant-injection technique consists of releasing a gaseous tracer, perfluorocarbon, through miniature permeation tubes at a constant rate inside a building and collecting the tracer with passive samplers called Capillary Atmospheric Tracer Samplers. The passive samplers are then analyzed by gas chromatography and quantified by an electron capture detector in an analytical laboratory (Koontz and Rector 1995, Dietz et al. 1986). Under steady state conditions, that is, after the tracer concentration becomes stable, Equation (5.1.1) reduces to:

$$I = \frac{S}{V\,C}\ . \qquad (5.1.3)$$

An increasing popular variant of this technique entails the use of a collecting monitor, consisting of a diffusive tube and an absorber, to measure the average concentration during the time the system is in operation. The average concentration is then proportional to the inverse of the ventilation rate.

Further references for these air exchange rate measurement techniques, as well as some predictive quantitative models, can be found in the work of Nazaroff et al. (1988), Nero (1988), Sherman (1990), Koontz and Rector (1995), and Dietz et al. (1986).

5.1.4 Data Input Requirements

In RESRAD (onsite) and RESRAD-OFFSITE, the user is required to input a value for the average building air exchange rate that represents conditions at the site. The air exchange rate should be given in units of inverse time (T^{-1}). This input parameter is used in determining the indoor radon concentration. In RESRAD-BUILD, an input of building air exchange rate is also required. The input is used in determining the indoor air concentrations of solid radionuclides (attaching to dust particles) as well as radon.

For generic use of the code, a default value of 0.5 h^{-1} was adopted in the RESRAD (onsite) and RESRAD-OFFSITE models and a default value of 0.8 h^{-1} was adopted in the RESRAD-BUILD model for the average building air exchange rate. The default value for RESRAD (onsite) and RESRAD-OFFSITE was based on the mean of residential buildings (see Table 5.1.1), which was increased to also include nonresidential buildings for use in RESRAD-BUILD. For more accurate use of the code, however, site-specific data should be applied whenever possible.

5.2 BUILDING ROOM HEIGHT

5.2.1 Definition

The building room height expresses the average height of the house. More specifically, it is defined as the ratio of the volume of the total internal space of the building to the internal area of its floor surface. This parameter is measured in units of length (l). For one-story houses without a basement, the values for the building room height typically lie within the range of 2.2–3.0 m.

5.2.2 Data Input Requirements

In RESRAD (onsite) and RESRAD-OFFSITE codes, the user is required to input a value for the building room height that represents conditions at the site. The dimensions of the input value of the room height should be given in units of meters (m).

For generic use of the codes, a default value of 2.5 m was adopted in the RESRAD (onsite) and RESRAD-OFFSITE codes for the building room height. For more accurate use of the code, however, site-specific data should be applied whenever possible.

5.3 BUILDING INDOOR AREA FACTOR

5.3.1 Definition

The building indoor area factor is the fraction of the floor area built on the contaminated area.

5.3.2 Data Input Requirements

A value of 1.0 means that the entire floor area was built on the contaminated area. Values greater than 1.0 indicate a contribution from walls extending into the contaminated zone.

If 0 is entered, the code will calculate a time-dependent area factor on the basis of an assumed floor area of 100 m^2 and the amount of wall area extending into the contaminated zone. For example, if the walls extend to a depth of 0.5 m into the contaminated zone, the building indoor area factor is equal to $1+ 0.5 \times 4/\sqrt{100}$, or 1.2. The building indoor area factor is time dependent because of soil erosion of the contaminated zone. The default value used in both codes (RESRAD [onsite] and RESRAD-OFFSITE) for the building indoor area factor is 0.

5.4 BUILDING FOUNDATION THICKNESS

5.4.1 Definition

The building foundation and floor thickness is the average thickness of the building material through which the radon has to diffuse to get into the building. Typical values lie around 0.15 m. If the dwelling is built upon contaminated soil, the radon that is released from contamination can enter the interior by diffusing through the floor or foundation.

5.4.2 Summary of Literature Data

There are three foundation types: deep basement, crawl space, and slab on grade. The factors affecting the choice of foundation type include site conditions (soil type, depth to water table, and depth of bedrock), overall building design, the climate, and construction costs. In the United States, basements are most common in the East and Midwest, crawl spaces are most common in the Northwest and West, and stem walls together with slabs on grade or monolithic slabs are most common in the Southeast (HUD 1999).

Table 5.4.1 lists the housing characteristics from the 2011 American Housing Survey (AHS) and the 2009 Residential Energy Consumption Survey. Of a total of 132.4 million housing units, 114.9 million housing units were occupied in 2011. Of the total, 90.7 million were single-family units, 32.6 million were apartments, and 9 million were mobile homes; 15.8 million of these units were built after 1999. Out of 90.7 million one-unit buildings, 28.7 million had full basements, 9.3 million had partial basements, 20.4 million units were built with crawl space, and 30.4 million units were built on concrete slabs. Fewer homes (113.6 million) were occupied in 2009.

The building footings (foundation thicknesses) are designed with adequate size to distribute the structure load to the soil and are placed beneath the maximum frost penetration depth or insulated to prevent frost penetration. Poured concrete and gravel fill are used as building footings. In a single-story frame house, the footing thickness should be at least 6 in. (Carmody et al. 1991). Basements are normally finished with concrete floors; 4 in. of compacted gravel is used as a base for the concrete floor. For a crawl type of construction, a 4-in.-thick layer of crushed stone or gravel is placed at the bottom and the footing thickness is ≥6 inches of concrete (Carmody et al. 1991). For residential construction, a slab-on-grade floor is approximately 4 inches thick and it rests on fill (HUD 1999).

A default value of 0.15 m was adopted in the RESRAD (onsite) and RESRAD-OFFSITE codes for the building foundation thickness. For more accurate use of the code, however, site-specific data should be applied whenever possible.

A default value of 0.15 m was adopted in the RESRAD (onsite) and RESRAD-OFFSITE codes for the building foundation thickness. For more accurate use of the code, however, site-specific data should be applied whenever possible.

5.4.3 Data Input Requirements

In RESRAD (onsite) and RESRAD-OFFSITE, the user is required to input a value for the building foundation and floor thickness that represents conditions at the site. The input value of the building foundation and floor thickness should be given in units of meters (m).

TABLE 5.4.1 Characteristics of Different Housing Units (in millions)

`		Single-Family Units		Apartments in Buildings		Mobile Units
	Total	Detached	Attached	2 to 4 Units	5 or more Units	Mobile Homes
Total homes[a]	132.4	82.9	7.8	10.7	21.9	9.0
Total occupied[a]	114.9	73.8	6.7	9.0	18.2	7.2
Built 2000–2014[a]	15.8	10.5	1.3	0.5	2.3	1.2
Total homes[b]	113.6	71.8	6.7	9.0	19.1	6.9
Basement[b]	34.9	29.2	3.0	2.7	NA[c]	NA
Crawl space[b]	26.1	24.1	1.1	0.9	NA	NA
Concrete slab[b]	40.7	33.3	3.5	3.9	NA	NA

[a] Data from AHS (2013).

[b] Data from EIA (2014).

[c] NA = not applicable.

5.5 FOUNDATION DEPTH BELOW GROUND SURFACE

5.5.1 Definition

The foundation depth below ground surface is the vertical distance in the soil from the very bottom of the basement floor slab to the ground surface. Typical values lie within the range of 0.0-3.0 m.

5.5.2 Data Input Requirements

In RESRAD (onsite) and RESRAD-OFFSITE, the user is required to input a value for the foundation depth below the ground surface that represents conditions at the site. The input value for the foundation depth should be given in units of meters (m).

A default value of 1.0 m was adopted in the RESRAD (onsite) and RESRAD-OFFSITE codes for the foundation depth. For more accurate use of the code, however, site-specific data should be applied whenever possible.

If a negative value is entered, the absolute value will be adjusted (if needed) so that the foundation depth will not extend into the contaminated zone. Thus, because of erosion of the cover and contaminated zones, the foundation depth can be time dependent and less than the (absolute) specified value.

A default value of −1 m is used in the RESRAD (onsite) and RESRAD-OFFSITE codes for the foundation depth. A negative value of "−1" for this parameter serves as a flag. For more accurate use of the code, however, site-specific data should be applied whenever possible.

5.6 FILTRATION FACTOR FOR INHALATION PATHWAY

5.6.1 Definition

This factor is the ratio of airborne dust concentration indoors on-site to the concentration outdoors on-site. It is based on the fact that a building provides shielding against entry of wind-blown dust particles. Therefore, calculation of the effective dose from the dust inhalation pathway should take into account this dust filtration effect. The filtration factor value is used in calculating the occupancy factor. For example, based on the assumption that 50% of the time is spent indoors with filtration factor of 0.4, 25% is spent outdoors in the contaminated area, and 25% is spent in uncontaminated areas, the occupancy factor for inhalation pathway dose calculations would be as follows:

$$0.5 \times 0.4 + 0.25 \times 1 + 0.25 \times 0 = 0.45 \ . \qquad (5.6.1)$$

The fraction of time spent outdoors on-site is an input parameter discussed in Section 9.6, the fraction of time spent indoors on-site is also an input parameter discussed in Section 9.7, and the fraction of time spent off-site is not an input parameter.

5.6.2 Summary of Literature Data

The filtration factor provides a measure of a building's effectiveness at removing particulate contaminants from the outdoor air that enters the building. This parameter is sometimes referred to as an inhalation shielding factor or inhalation pathway dose reduction factor (DRF). Table 7.1-1 in NUREG/CR-6697 (Yu et al. 2000) lists the measured indoor/outdoor air concentration ratios in different building types for different pollutants. The measured ratio varied from close to 0 to 1 (Yu et al. 2000). For the probabilistic analysis, use the distributions developed for indoor dust filtration factor in NUREG/CR-6697 (Yu et al. 2000). The filtration factor or DRF depends on the building's ventilation rate and the deposition velocity of the particulates. The following equations, derived by Fogh et al. (1997), can be used in estimating the particle size dependent filtration factor:

$$DRF = \frac{\lambda_r}{S/V \times V_d + \lambda_r} \text{, and} \qquad (5.6.2)$$

$$V_d = (0.48 + 0.6 \times d_p) \times 0.36 \ , \qquad (5.6.3)$$

where

S = interior surface area (m^2),

V = interior volume (m^3),

λ_r = building air exchange rate (h^{-1}),

V_p = average deposition velocity (mh^{-1}), and

d_p = particle diameter (μm).

Facciola (2006) studied the infiltration of ultrafine (<0.1 micron) and fine particulate matter into indoor environments. Four mechanically vented buildings, including a typical office space and an elementary school, were included in the study. The study found that the average indoor/outdoor ratio ranged from 0.45 to 0.6 for all particle diameters from 55 to 700 nm and it was not dependent on the time of day when the measurement was taken or the wind speed; however, the ratio was higher when the HVAC system was in use.

5.6.3 Data Input Requirements

The filtration factor should be input as a fraction ranging from 0 to 1. The default filtration factor for dust inhalation in the RESRAD (onsite) and RESRAD-OFFSITE codes is 0.4, which assumes that the dust level indoors is 40% of the outdoor level (Alzona et al. 1979).

5.7 SHIELDING FACTOR FOR EXTERNAL GAMMA RADIATION

5.7.1 Definition

This factor is the ratio of the external gamma radiation level indoors on-site to the radiation level outdoors onsite. It is based on the fact that a building provides shielding against the penetration of gamma radiation. Therefore, the calculation of the effective dose from the ground pathway should take into account this shielding effect.

5.7.2 Summary of Literature Data

The external gamma shielding factor depends on the house construction type, as well as on gamma energy. The external gamma shielding factor is one of the significant parameters affecting the dose in cases where the external exposure pathway dominates the total dose. Many recent studies have used the MCNP5 Monte Carlo radiation transport code to estimate the site-specific external shielding factor. Salinas et al. (2006) studied external gamma shielding factors for typical houses in Brazil from uniform deposited contamination on the ground, walls, and roof. Table 5.7.1 lists the shielding factors for three energies (300, 662, and 3000 keV) for three house types. As expected, the shielding provided depends on the type of construction as well as the gamma energy. Barr et al. (2010) developed site-specific shielding factors for multiple building-source configurations for Th-232 decay chain radionuclides. Table 5.7.2 lists the estimated shielding factor for a receptor located on the ground floor and spending equal times in all areas. Dickson and Hamby (2014) identified the general construction materials used in U.S. homes from the published Residential Energy Consumption Survey (RECS) data and provided the shielding protection factors for these construction materials (Table 5.7.3). The source term used was based on the release from the light water reactor core inventory in NUREG/CR-4467 (Alpert et al. 1986). Cement walls provided the most shielding. Table 5.7.4 lists the average external shielding factors for different types of house construction for deposited contamination on the ground and roof and also for cloud immersion (Dickson 2013). In general, clay brick walls with asphalt roof construction provided the most shielding.

The occupancy factor, FO_1, for the ground pathway can be obtained by using the following equation:

$$FO_1 = TF_1 \times 1 + TF_2 \times 0.7 + TF_3 \times 0 \ , \qquad (5.7.1)$$

where

TF_1 = fraction of time spent outdoors on-site (an input parameter, Section 9.6),

TF_2 = fraction of time spent indoors on-site (an input parameter, Section 9.5), and

TF_3 = fraction of time spent off-site (not an input parameter, Sections 9.5 and 9.6).

5.7.3 Data Input Requirements

The shielding factor for external gamma radiation should be input as a fraction, ranging from 0 to 1. The default shielding factor used in the RESRAD (onsite) and RESRAD-OFFSITE codes for external gamma radiation is 0.7, which assumes that the external gamma radiation level indoors is 30% lower than the outdoor gamma radiation level. For the probabilistic analysis, use the distribution developed for the external shielding factor in NUREG/CR-6697 (Yu et al. 2000).

TABLE 5.7.1 External Gamma Shielding Factors for Deposited Contamination

House Type	Wall Characteristics	Shielding Factor 300 keV	662 keV	3000 keV
Clay brick	Brick: 7.5 cm thick, density = 0.737 g/cm^3	0.36 (0.26)	0.43 (0.31)	0.58 (0.44)
Clay brick with cement on one side	Brick: 9.6 cm thick, density = 0.917 g/cm^3	0.25 (0.18)	0.33 (0.24)	0.51 (0.51)
Clay brick with cement on both sides	Brick: 11.4 cm thick, density = 1.145 g/cm^3	0.17 (0.12)	0.24 (0.17)	0.42 (0.30)

Note: All the houses were of same dimension (length 10 m, width 14 m, and height 3.48 m) and were assumed to have a concrete roof and floor (density 2.4 g/cm^3) with thickness of 15 cm and 33 cm, respectively. The contamination was uniformly deposited outside on a 50-cm layer of soil with a density of 1.8 g/cm^3. The walls and roof were also uniformly contaminated at the same concentration. The values in brackets represent the shielding factor when there was no contamination on walls and roof.

Source: Salinas et al. (2006).

TABLE 5.7.2 External Shielding Factors for Th-232 Series Radionuclides

Building Type	Floor Characteristics	Wall Characteristics	Contamination Configuration	Shielding Factor
Configuration 1	Gravel: 10 cm thick, 1.5 g/cm^3 density + concrete: 10 cm thick, 2.25 g/cm^3 density	Wood: 2.54 cm thick, 0.7 g/cm^3 density + gypsum: 1.25 cm thick, 2.32 g/cm^3 density	House (10 m by 10 m) constructed on top of 2-m-thick contamination and no outside contamination	0.104
Configuration 2	Gravel: 10 cm thick, 1.5 g/cm^3 density + concrete: 10 cm thick, 2.25 g/cm^3 density	Wood: 2.54 cm thick, 0.7 g/cm^3 density + gypsum: 1.25 cm thick, 2.32 g/cm^3 density	Large area (100 m by 100 m) contaminated to 2-m depth and house constructed at the center	0.369
Configuration 3	Gravel: 10 cm thick, 1.5 g/cm^3 density + concrete: 10 cm thick, 2.25 g/cm^3 density	Brick: 10 cm thick, 2.35 g/cm^3 density	House (10 m by 10m) constructed on top of 2-m-thick contamination and no outside contamination	0.105
Configuration 4	Gravel: 10 cm thick, 1.5 g/cm^3 density + concrete: 10 cm thick, 2.25 g/cm^3 density	Brick: 10 cm thick, 2.35 g/cm^3 density	Large area (100 m by 100 m) contaminated to 2-m depth and house constructed at the center	0.241

Table 5.7.2 (Cont.)

Building Type	Floor Characteristics	Wall Characteristics	Contamination Configuration	Shielding Factor
Configuration 5	Basement walls of brick: 10 cm thick, 2.25 g/cm^3 density. Floor between basement and first floor of wood: 2.54 cm thick, density 0.7 g/cm^3	Wood: 2.54 cm thick, 0.7 g/cm^3 density + gypsum: 1.25 cm thick, 2.32 g/cm^3 density	Large area (100 m by 100 m) contaminated to 2-m depth and house constructed with basement at the center. No contamination below the basement floor.	0.359
Configuration 6	Gravel: 5 cm thick, density 1.8 g/cm^3 + wood: 2.5 cm thick, density 0.6 g/cm^3	Wood: 2.54 cm thick, 0.7 g/cm^3 density + gypsum: 1.25 cm thick, 2.32 g/cm^3 density	House constructed on top of 1-m-thick contamination and no outside contamination	0.356

Note: All the building types were of same dimensions: length 10 m, width 10 m, height 2.5 m. Each configuration has one glass window: thickness 1.2 cm, size 4 m by 1.25 m, and density 2.32 g/cm^3.

Source: Barr et al. (2010).

TABLE 5.7.3 Shield Protection Factors per Unit Thickness for Different Construction Materials

Construction Material	Shielding Factor
Cement wall	0.02
Brick wall	0.18
Stucco wall	0.63
Wood wall	0.68
Steel wall	0.72
Vinyl wall	0.74
Internal wall	0.77
Asphalt roof	0.75
Terracotta roof	0.78
Cedar shake roof	0.81
Steel roof	0.85

Source: Dickson and Hamby (2014).

TABLE 5.7.4 Average External Shielding Factors for Different Types of House Construction

		Shielding Factor		
Construction Type	House Type	Ground Deposition	Roof Deposition	Cloud Immersion
Clay brick siding with asphalt roof	One story - NB	0.31	0.69	0.57
	One story - WB	0.18	0.52	0.42
	Two story - NB	0.31	0.73	0.47
	Two story - WB	0.23	0.6	0.36
Vinyl siding with asphalt roof	One story - NB	0.77	0.7	0.79
	One story - WB	0.43	0.51	0.61
	Two story - NB	0.6	0.54	0.79
	Two story - WB	0.56	0.58	0.65
Manufactured houses with vinyl siding and asphalt roof	First floor	0.82	0.76	0.82
Manufactured houses with steel siding and steel roof	First floor	0.84	0.72	0.84

Note: WB = with basement; NB = no basement.

Source: Dickson (2013).

This page intentionally left blank.

6 CROPS AND LIVESTOCK PARAMETERS

6.1 ROOT DEPTH

6.1.1 Definition

This parameter represents the average root depth of various plants grown in the contaminated zone. The plant types consumed by humans are divided into two categories: (1) leafy vegetables and (2) fruit, grain, and nonleafy vegetables. For RESRAD-OFFSITE, the plant types consumed by livestock are divided in two categories: (1) pasture and silage and (2) grain. For RESRAD (onsite), it is assumed that fodder is consumed by livestock.

6.1.2 Summary of Literature Data

The root depth varies for different plants. For some plants, such as beets, carrots, lettuce, and so forth, it does not extend below about 0.3 m. For others, such as fruit trees, the roots may extend 2 or 3 m below the surface; taproots for some crops (e.g., alfalfa) can extend to 5 m. Most of the plant roots from which nutrients are obtained, however, usually extend to less than 1 m below the surface.

Each crop has characteristic rooting habits that it will tend to follow if the soil is deep, uniform, and equally moist throughout. The depth of rooting increases during the growing period. Crops that mature in 2 months usually penetrate only 0.6 to 0.9 m, and crops requiring 6 months to mature may penetrate 1.8 to 3.0 m or more.

When the upper portion of the soil is kept moist, plants will obtain most of their moisture supply from near the surface. As the moisture content of the upper layers decreases, the plants draw more water from the lower layers; this phenomenon encourages more root development in the lower levels. Fewer roots exist in the lower portion of the root zone because of the inability of the root system to extract enough moisture from the lower levels. Generally, the average root-zone depths are reached by the time the foliage of the plant has reached its maximum size. Root-zone depths are limited to the soil depth above the water table.

Root depths can be measured using destructive techniques such as soil coring, in-growth cores, whole root system excavation, and trenching; or non-destructive sampling techniques including rhizotrons and minirhizotrons (Johnson et al. 2001). In the destructive technique, soil samples are taken, the roots are washed out carefully, and root length or surface area is measured (Steingrobe et al. 2001). In the minirhizotron technique, clear acrylic or glass tubes are installed into the soil and a video camera or a fiber optics cable is lowered in the tube to observe the root growth at the soil-tube interface (Johnson et al. 2001).

Table 6.1.1 lists rooting depths for fruits and nuts, grains, and nonleafy vegetables that can be consumed by humans. In general, the grain crops grown during the spring season have shallower rooting depths compared to the crops grown during the winter season. For fruit, grain, and nonleafy vegetables, a uniform distribution with a minimum of 0.3 m and a maximum of 2.4 m is suggested for use in RESRAD-OFFSITE. If specific conditions are known, values from Table 6.1.1 for a specific plant type may be used.

Table 6.1.2 lists rooting depths for leafy vegetables that can be consumed by humans. A uniform distribution with a minimum of 0.3 m and a maximum of 0.9 m is suggested for use in RESRAD-OFFSITE for leafy vegetables. If specific conditions are known, values from Table 6.1.2 for a specific leafy vegetable may be used.

Table 6.1.3 lists rooting depths for a variety of forages that can be used for livestock consumption. Growing conditions (e.g., amount of rainfall or temperature) vary annually and geographically across the United States, and the type of forage consumed by livestock is uncertain. Therefore, for forages, a uniform distribution with a minimum of 0.3 m and a maximum of 3.6 m is suggested for use in RESRAD-OFFSITE. If specific conditions are known, values from Table 6.1.3 for a specific forage type may be used.

Table 6.1.4 lists rooting depths for a variety of grains that can be used for livestock consumption. In general, the grain crops grown during the spring season have shallower rooting depths compared to the crops grown during the winter season. The criteria cited for forage crops also apply to grain crops. Therefore, for grain, a uniform distribution with a minimum of 0.5 m and a maximum of 2.4 m is suggested for use in RESRAD-OFFSITE. If specific conditions are known, values from Table 6.1.4 for a specific grain type may be used.

Minimum and maximum values of 0.3 and 4.0 m for the root depth are suggested as input to RESRAD (onsite), which does not distinguish among the plant types. These values bound those presented in Tables 6.1.1–6.1.4. However, site-specific minimum and maximum root depths should be used, on the basis of the plant types present.

6.1.3 Data Input Requirements

Root depth is used to calculate the cover and depth factor for the plant, meat, and milk pathways because edible plants become contaminated through root uptake of radionuclides. Uptake of radionuclides from plant roots is assumed possible only when the roots extend to the contaminated zone and is limited to the fraction of roots that have direct contact with contaminated soil.

The average root depth should be entered in units of meters (m). In the RESRAD (onsite) code, the default value is 0.9 m. In the RESRAD-OFFSITE code, the default value for fruit, grain, and nonleafy vegetables consumed by humans and grain consumed by livestock is 1.2 m; and the default value for leafy vegetables consumed by humans and pasture and silage consumed by livestock is 0.9 m.

TABLE 6.1.1 Root Depths of Fruits and Nuts, Grains, and Nonleafy Vegetables from Different Sources

Plant Type	Root Depth Range (m)	Allen et al. (1998)	Weaver and Brunner (1927)	Kemble and Sanders (2000)	Evans et al. (1996)	Curwen and Massie (1994)	Weaver (1926)
Fruit and nuts							
Almonds, apples, apricots, cherries, grapes, peaches, and pears	1.0–2.0	1.0–2.0					
Avocados	0.5–1.0	0.5–1.0					
Bananas	0.5–0.9	0.5–0.9					
Berries	0.6–1.2	0.6–1.2				0.6–1.2	
Cantaloupes	0.3–1.5	0.9–1.5	1.1	0.3–0.6			
Citrus fruits	1.2–1.5	1.2–1.5					
Kiwis	0.7–1.3	0.7–1.3					
Olives	1.2–1.7	1.2–1.7					
Pineapples	0.3–0.6	0.3–0.6					
Pistachios	1.0–1.5	1.0–1.5					
Strawberries	0.2–0.6	0.2–0.3	0.3–0.6		0.3	0.3	
Sweet melons, watermelons	0.8–1.5	0.8–1.5	1.1	>0.6		0.6–1.2	
Grain							
Barley	1.0–2.0	1.0–1.5					1.4–2.0
Corn	1.0–2.4	1.0–1.7	1.5–2.4			0.6–1.2	1.5–1.8
Millet	1.0–2.0	1.0–2.0					
Oats	1.0–1.5	1.0–1.5					1.2–1.5
Rice	0.5–1.0	0.5–1.0					
Sorghum	1.0–2.0	1.0–2.0					1.4–1.8
Spring wheat	1.0–1.5	1.0–1.5				0.6	
Winter wheat	1.5–2.1	1.5–1.8					1.5–2.1
Nonleafy vegetables							
Artichokes	0.6–0.9	0.6–0.9					
Asparagus	1.2–3.0	1.2–1.8	1.5–3.0	>0.6			
Carrots	0.3–2.0	0.5–1.0	0.6–2.0	0.3–0.6	0.5		
Chick peas	0.3–1.0	0.6–1.0	0.6–1.0	0.3–0.6	0.5	0.6	
Cucumbers	0.3–1.2	0.7–1.2	1.1	0.3–0.6			
Eggplants	0.3–2.0	0.7–1.2	1.2–2.0	0.3–0.6			
Green beans	0.5–0.7	0.5–0.7		0.5–0.6	0.5		
Lima beans	0.6–1.2	0.8–1.2	0.9–1.2	>0.6		0.6	
Okra	0.5–1.2		0.5–1.2	>0.6			
Onions	0.3–1.0	0.3–0.6	0.5–1.0	0.3–0.5	0.3	0.5	
Potatoes	0.3–0.9	0.4–0.6		0.3–0.5	0.5	0.5	0.6–0.9
Pumpkins	0.6–1.8	1.0–1.5	1.8	>0.6		0.6–1.2	
Radishes	0.3–0.9	0.3–0.5	0.6–0.9				
Squash (zucchini)	0.3–1.8	0.6–1.0	1.8	0.3–0.6		0.6–1.2	
Sugar beets	0.5–2.0	0.7–1.2	1.2–2.0	0.5–0.6	0.5	0.5	1.5–1.8
Sweet peppers	0.3–1.2	0.5–1.0	0.9–1.2	0.3–0.6	0.3	0.6	
Sweet potatoes	0.6–1.5	1.0–1.5	1.2	>0.6			
Tomatoes	0.6–1.7	0.7–1.5	1.0–1.7	>0.6			
Turnips	0.5–1.5	0.5–1.0	1.5				

TABLE 6.1.2 Root Depths of Leafy Vegetables from Different Sources

Leafy Vegetable	Root Depth Range (m)	Curwen and Massie (1994)	Allen et al. (1998)	\Weaver and Brunner (1927)	Kemble and Sanders (2000)	Evans et al. (1996)
Spinach	0.3–0.5		0.3–0.5	0.3	0.3–0.5	0.3–0.5
Lettuce	0.3–0.9	0.5	0.3–0.5		0.6–0.9	0.3–0.5
Broccoli	0.3–0.6		0.4–0.6		0.3–0.5	0.3–0.5
Celery	0.3–0.5		0.3–0.5		0.3–0.5	
Cabbage	0.3–0.9		0.5–0.8	0.9	0.3–0.5	0.3–0.5
Cauliflower	0.3–0.9		0.4–0.7	0.9	0.3–0.5	0.3–0.5
Brussels sprouts	0.3–0.6		0.4–0.6		0.3–0.5	
Mint	0.4–0.8		0.4–0.8			
Collards	0.3–0.5				0.3–0.5	
Mustard	0.3–0.6				0.5–0.6	0.3–0.5

TABLE 6.1.3 Root Depths of Forages from Different Sources

Forage Type	Root Depth Range (m)	Curwen and Massie (1994)	Weaver (1926)	Georgeson and Payne (1897)	Canadell et al. (1996)	Allen et al. (1998)
Alfalfa	0.6–3.6	0.6–1.2	3–3.6	1.5–1.8		1.0–3.0
Bermuda grass	1.0–1.5					1.0–1.5
Bluegrass	0.3–2.1		1.5–2.1	0.3–1.1		
Broome grass	1.1–2.0		1.7–2.0		1.1	
Canary grass	0.6–1.5			0.6–1.5		
Clover, ladino	1.5–2.4		1.5–2.4			
Clover, red	0.6–2.4	0.6	1.5–2.4	1.5		0.6–0.9
Fescue	0.6–1.2		0.6–1.2			
Orchard grass	0.9–1.3		0.9–1.3			
Rye grass	0.6–0.9		0.6–0.9			0.6–1.0
Trefoil	0.6–1.2	0.6–1.2				
Timothy	0.4–0.9		0.4–0.9			
Buffalo grass	0.6–1.9		0.6–0.9	0.9	1.9	
Pasture grasses	0.6–1.5	0.6–1.2				0.5–1.5
Big bluestem	1.5–2.8		1.5–2.7		1.5–2.8	
Little bluestem	0.9–1.8		0.9–1.7		1.5–1.8	

[a] Modified from indicated sources.

TABLE 6.1.4 Root Depths of Grains from Different Sources

Grain Plant	Range Root Depth (m)	Allen et al. (1998)	Weaver and Brunner (1927)	Curwen and Massie (1994)	Weaver (1926)
Barley	1.0–2.0	1.0–1.5			1.4–2.0
Corn	1.0–2.4	1.0–1.7	1.5–2.4	0.6–1.2	1.5–1.8
Millet	1.0–2.0	1.0–2.0			
Oats	1.0–1.5	1.0–1.5			1.2–1.5
Rice	0.5–1.0	0.5–1.0			
Sorghum	1.0–2.0	1.0–2.0			1.4–1.8
Spring wheat	1.0–1.5	1.0–1.5		0.6	
Winter wheat	1.5–2.1	1.5–1.8			1.5–2.1

6.2 LIVESTOCK WATER INTAKE RATE FOR BEEF CATTLE AND MILK COWS

6.2.1 Definition

According to NRC Regulatory Guide 1.109 (NRC 1977), the water ingestion rate for beef cattle is 50 L/d. The water ingestion rate for milk cows is 14 gal/d (approximately 50 L/d) plus 1 gal for every 3 lb of milk produced. If a production rate of 10 gal/d of milk is assumed, then the water ingestion rate for milk cows is about 160 L/d (Gilbert et al. 1983).

Livestock water intake rate depends on many factors including age, weight, reproductive stage, type of cattle, type of feed (green pasture vs. dry fodder), and temperature (in hot weather, livestock use more water for evaporative cooling). Table 6.2.1 shows the variation of total water intake of beef cattle with temperature. For mature bulls, the daily water intake changes from 8.7 gal/d to 20.6 gal/d as the temperature changes from 40°F to 90°F. Table 6.2.2 lists the water consumption by different types of dairy cattle. The water consumption depends on the level of milk produced. The average water consumption of milk cows is 115 L/d. Table 6.2.3 lists the daily water consumption by different types of beef cattle. The average water consumption of beef cattle varies from 25 to 55 L/d.

6.2.2 Data Input Requirements

In the RESRAD (onsite) and RESRAD-OFFSITE codes, the livestock water intake should be entered in units of liters per day (L/d). The default values for beef cattle and milk cows are set to 50 and 160 L/d, respectively, if the user does not specify otherwise. The default values are on the high end compared to the recent data collected in Table 6.2.3.

TABLE 6.2.1 Variations in Total Daily Water Intake of Beef Cattle (gal/d) with Temperature

Beef Cattle Type	40°F	50°F	60°F	70°F	80°F	90°F
Growing heifers, steers, bulls	4.0–6.3	4.3–6.8	5.0–7.9	5.8–9.2	6.7–10.6	9.5–15.0
Finishing cattle	6.0–8.7	6.5–9.4	7.4–10.8	8.7–12.6	10.0–14.5	14.3–20.6
Wintering beef cows	6.0–6.7	6.5–7.2	7.4–8.3	8.7–9.7	Not available	Not available
Lactating cows	11.4	12.6	14.5	16.9	17.9	18.2
Mature bulls	8.0–8.7	8.6–9.4	9.9–10.8	11.7–12.6	13.4–14.5	19.0–20.6

Source: Rasby and Walz (2011).

TABLE 6.2.2 Water Consumption by Dairy Cattle

Dairy Cattle Type	Level of Milk Production (kg/d)	Water Consumption Range (L/d)	Average Water Consumption (L/d)
Dairy calves (1–4 mo.)	0	4.9–13.2	9
Dairy heifers (5–24 mo.)	0	14.4–36.3	25
Milking cows	13.6	68–83	115
	22.7	87–102	
	36.3	114–136	
	45.5	132–155	
Dry cows	0	34–49	41

Source: Ward and McKague (2007).

TABLE 6.2.3 Water Consumption by Beef Cattle

Beef Cattle Type	Weight Range (kg)	Water Consumption Range (L/d)	Average Water Consumption (L/d)
Feedlot cattle: Backgrounder	181–364	15–40	25
Feedlot cattle: Short keep	364–636	27–55	41
Lactating cows with calves	Not available	43–67	55
Dry cows, bred heifers and bulls	Not available	22–54	38

Source: Ward and McKague (2007).

6.3 PLANT TRANSFER FACTORS

6.3.1 Definition

The soil-to-plant transfer factor is defined as the ratio of radionuclide concentration in the plant food product at the time of harvest (fresh weight basis) to that of soil radionuclide concentration (dry weight basis). It is expressed as pCi/kg plant (wet)/pCi/kg dry soil.

6.3.2 Discussion

Comparison of soil-to-plant transfer factors used in RESRAD (onsite) and RESRAD-OFFSITE with those used in other published radiological assessment models can be difficult because the parameters are generally reported in one of two different formats. In RESRAD (onsite) and RESRAD-OFFSITE, soil-to-plant transfer factor is expressed as the ratio pCi per gram plant (wet)/pCi per gram soil (dry). In other published reports, the transfer factors have been based on either the fresh (wet) weight or the dry weight of the vegetation. To aid in converting between the two bases of measurement, representative dry-to-wet weight ratios for food crops and forage plants that have been reported by Baes et al. (1984), NRC (1983), and IAEA (2010a) are listed in Table 6.3.1.

The soil-to-plant transfer factor varies in a complex manner with soil properties and geochemical properties of the radionuclide in soil. After entering the transpiration stream, radionuclides may not be uniformly distributed within a plant, but instead tend to concentrate in certain parts (Grogan 1985). Many studies have shown that the plant transfer factor also varies with crop type and variety, stage of growth, and plant part, as well as with subsoil characteristics and agriculture practices (Baes et al. 1984; IAEA 2010a; Ng et al. 1982a). Comprehensive data on plant transfer factors in different crops grown on various soils are available in the literature for relatively few radionuclides. When there is no data or few data for transfer factors are available, an analogue approach (analogue isotopes, analogue elements, analogue species) can be used in estimating the transfer factors on the basis of the assumption that chemically similar elements act similarly in soil-plant environments (Baes et al. 1984; Staven et al. 2003). Care should be taken not to use very short-lived isotopes to estimate the behavior of long-lived isotopes, because equilibration of a short-lived isotope in environmental media would be influenced by its physical decay whereas biogeochemical processes may determine the equilibration of a long-lived isotope. Elements in the same group of the periodic table have the same number of electrons in their outer shell and exhibit similar chemical behavior; however, this approach should be used with care where essential elements are involved (IAEA 2010a, Napier et al. 2014). Care should also be taken in measuring the transfer factors for very mobile radionuclides (such as chlorine and technetium) that are subject to considerable migration to deeper soil layers, making the soil activities at the end of the vegetation growth period much lower than those at the beginning of the growth period. The activity in the plants is due to uptake from soil during the whole growth period. The time when the soil concentration measurements were taken will affect the value of the transfer factor; therefore, it is advisable to collect soil measurement data throughout the plant growth period and use the average soil concentration in determining the transfer factor.

TABLE 6.3.1 Dry-to-Wet Weight Conversion Factors for Different Plant Types

Plant Group	Crop	Type	Baes et al. (1984)	NRC (1983)	IAEA (2010)
Cereals	Grains	Barley	0.889	0.926	0.87
		Rice	–[a]	0.877	–
		Wheat	0.875	0.87	0.88
		Winter rye	–	–	0.87
		Oats	–	–	0.87
		Millet	–	–	0.88
		Sorghum	–	–	0.87
		Corn	0.895	0.263	0.85
	Stems and shoots	Barley	–	–	0.34
		Rice	–	–	–
		Wheat	–	–	0.18
		Winter rye	–	–	0.23
		Oats	–	–	0.28
		Millet	–	–	0.23
		Sorghum	–	–	0.25
		Corn	–	–	0.19
		Asparagus	0.07	0.083	
Leafy veg.	Leaves	Cabbage	–	0.077	0.12
		Cauliflower	–	0.083	0.11
		Celery	–	0.063	0.06
		Lettuce	–	0.05	0.08
		Rhubarb	–	0.053	–
		Spinach	–	0.083	0.08
		Broccoli	–	0.11	–
		Brussels sprouts	–	0.147	–
		Kale	–	0.125	–
		Turnip greens	–	0.1	0.12
		Leeks	–	–	0.11
		Onions (above-ground part)	–	–	0.11
Non-leafy veg. and fruits	Fruits, heads, berries, buds	Kohlrabi	–	–	0.06
		Apple	0.159	0.149	0.16
		Apricot	–	0.147	0.15
		Banana	–	0.244	–
		Blackberry	–	0.156	–
Leguminous veg.	Seeds and pods	Jerusalem artichokes	–	–	0.22
		Tapioca	–	–	0.38
		Field peas	–	–	0.85
		Garden peas	–	–	0.83
		Grass pea vine	–	–	0.86
		Soya	–	–	0.87
		Lupin yellow	–	–	0.85
		Lupin blue	–	–	0.86
		Broad beans	–	–	0.88
		Lima beans	–	0.322	–
		Peas	0.257	0.169	–
		Green bean	–	0.1	–
		Chestnuts	–	0.476	–
		Peanuts	0.92	0.943	–
	Stems and roots	Field peas	–	–	0.17
		Garden peas	–	–	0.16
		Grass pea vine	–	–	0.21
		Soya	–	–	0.26
		Lupin yellow	–	–	0.14
		Lupin blue	–	–	0.18
		Seradella	–	–	0.22
		Broad beans	–	–	0.18
		Beans	–	–	0.28
		Lentils	–	–	0.25
Leguminous fodder	Stems and roots	Spring vetch	–	–	0.24
		Winter vetch	–	–	0.22
		Field peas	–	–	0.17
		Garden peas	–	–	0.16
		Grass pea vine	–	–	0.21
		Soya	–	–	0.26
		Lupin yellow	–	–	0.14
		Lupin blue	–	–	0.18
		Seradella	–	–	0.22
		Broad beans	–	–	0.18

TABLE 6.3.1 (Cont.)

Plant Group	Crop	Type	Baes et al. (1984)	NRC (1983)	IAEA (2010)
		Blueberry	–	0.167	–
		Cherry	0.17	0.196	–
		Fig	–	0.227	–
		Pear	0.173	0.167	0.17
		Peach	–		0.11
		Pineapple	–	0.147	–
		Plum	0.54	0.189	–
		Raspberry	–	0.175	0.16
		Cucumber	0.039	0.05	0.05
		Eggplant	0.073	0.077	–
		Pepper	–	0.067	–
		Pumpkin	–	0.084	0.08
		Squash	0.082	0.06	–
		Tomato	0.059	0.067	0.06
		Grapefruit	0.112	0.116	–
		Orange	0.128	0.141	0.14
		Peach	0.131	0.109	0.11
		Strawberry	0.101	0.101	0.10
		Cantaloupe	0.06	–	–
		Watermelon	0.079	–	0.07
		Lemon	0.107	–	–
		Grape	–	–	0.18
		Vegetable marrow	–	–	0.09
		Zucchini	–	–	0.05
Root crops	Roots	Beet	–	0.127	0.16
		Sugar beet	–	–	0.22
		Turnip	–	0.085	0.12
		Radish	–	0.056	0.09
		Carrot	0.118	0.118	0.14
		Onion	0.125	0.116	–
Tubers	Tubers	Potato	0.222	0.222	0.21
		Sweet potato	0.315	0.294	–
		Yam	–	0.263	–
		Beans	–	–	0.28
		Lentils	–	–	0.25
		Annual rye grass	–	–	0.20
		Alfalfa	–	–	0.26
		Sickle alfalfa	–	–	0.33
		Bastard lucerne	–	–	0.23
		Red clover	–	–	0.22
		Ladino clover	–	–	0.26
		Sainfoin	–	–	0.23
		White sweet clover	–	–	0.22
		Yellow sweet clover	–	–	0.22
Grass	Grass	Fussian brome grass	–	0.182	0.21
		Slender wheat grass	–	–	0.34
		Couch grass	–	–	0.37
		Standard crested grass	–	–	0.39
		Timothy grass	–	–	0.26
		Meadow fescue	–	–	0.20
		Cock’s foot grass	–	–	0.22
		Meadow grass	–	–	0.22
		Sudan grass	–	–	0.20
Animal feed	Fodder	Concentrate feed	–	–	0.88
		Grass silage	–	–	0.26
		Pasture	–	–	0.20
		Grass hay	–	–	0.86
		Lucerne hay	–	–	0.86
		Lucerne silage	–	–	0.34
		Corn silage	–	–	0.25

[a] Hyphen indicates value not available.

The RESRAD (onsite) and RESRAD-OFFSITE codes use composite values of plant transfer factors. Differences among food crops (such as leafy vegetables, root vegetables, fruits, grain, and forage plants) and consumption groups (such as humans and animals) are not considered. An example of calculating the composite plant transfer factor is provided in Appendix B of the study by Gnanapragasam and Yu (1997).

6.3.3 Data Input Requirements

The soil-to-plant transfer factor is used when the plant ingestion pathway is active. Three plant categories—fruit/grain/nonleafy vegetables; leafy vegetables; and pasture silage—are used in the RESRAD (onsite) code, and four plant categories—fruit/grain/nonleafy vegetables; leafy vegetables; pasture silage; and livestock feed grain—are used in the RESRAD-OFFSITE code. The first two plant categories (fruit/grain/nonleafy vegetables and leafy vegetables) are for human consumption, and fodder, pasture silage, and feed grain are for animal consumption.

The default values of plant transfer factors compiled from multiple sources were reviewed in 1993 (Wang et al. 1993) and were used as default values in the RESRAD (onsite) and RESRAD-OFFSITE codes. Since the publication of that report, a number of new publications on transfer factors have become available. A probabilistic dose analysis capability was developed for the RESRAD codes in 2000 (Yu et al. 2000), and a plant transfer factor distribution was developed using data from the National Council on Radiation Protection and Measurements (NCRP 1999). The NCRP report provides median and geometric standard deviations for composite wet plant/soil transfer factors. Table 6.3.2 lists the lognormal distribution parameter values estimated for plant transfer factors from NCRP (1999). The other published radiological assessments used for comparison of plant transfer factors in this handbook are those of Staven et al. (2003), IAEA (2009, 2010a), and Napier et al. (2014)~~.~~ To select plant transfer factors, the most recent and comprehensive data compilation should be given first priority.

In a Pacific Northwest National Laboratory (PNNL) report, Staven et al. (2003) provide transfer factors for four plant types: leafy vegetables, fruits, grains, and root vegetables. The recommended transfer factors in the PNNL report are provided on a dry weight basis and are listed in Table 6.3.3. The following dry-to-wet conversion factors are used: leafy vegetables 0.2, root vegetables 0.25, fruits 0.18, and grains 0.91. These plant transfer factors are used in the GENII system of computer codes.

The IAES's Technical Report Series No. 472, *Handbook of Parameter Values for the Prediction of Radionuclide Transfer in Terrestrial and Freshwater Environments* (IAEA 2010a), and its supporting document (IAEA 2009) include the full compilation of reviewed data and the methods used to obtain the data values. To decrease the uncertainty associated with plant transfer factors, IAEA (2010a) assigned individual plants to 14 plant groups: cereals, maize, rice, leafy vegetables, nonleafy vegetables, leguminous vegetables, root crops, tubers, fruits, grasses, leguminous fodder, pasture, herbs, and other crops. The IAEA report provides the plant transfer factors in temperate, tropical, and subtropical environments. Tables 6.3.4–6.3.8 list the plant transfer factors (on dry-weight basis) for different plant groups in the temperate environment, an environment appropriate for North America. If the codes are used in other countries where the

TABLE 6.3.2 Plant Transfer Factor Distribution Parameter Values from NCRP (1999)

Element	GM	GSD	μ	σ
Ac	1×10^{-3}	3.0	-6.9	1.1
Ag	4×10^{-3}	2.5	-5.5	0.9
Al	4×10^{-3}	3.0	-5.5	1.1
Am	1×10^{-3}	2.5	-6.9	0.9
As	8×10^{-2}	3.0	-2.5	1.1
At	2×10^{-1}	3.0	-1.6	1.1
Au	1×10^{-1}	3.0	-2.3	1.1
B	1×10^{-2}	3.0	-4.6	1.1
Ba	1×10^{-2}	2.5	-4.6	0.9
Be	4×10^{-3}	3.0	-5.5	1.1
Bi	1×10^{-1}	3.0	-2.3	1.1
Bk	1×10^{-3}	3.0	-6.9	1.1
Br	4×10^{-1}	3.0	-0.9	1.1
Ca	5×10^{-1}	3.0	-0.7	1.1
Cd	5×10^{-1}	3.0	-0.7	1.1
Ce	2×10^{-3}	2.7	-6.2	1.0
Cf	1×10^{-3}	3.0	-6.9	1.1
Cl	2×10^{1}	3.0	3.0	1.1
Cm	1×10^{-3}	2.5	-6.9	0.9
Co	8×10^{-2}	2.5	-2.5	0.9
Cr	1×10^{-2}	2.7	-4.6	1.0
Cs	4×10^{-2}	2.7	-3.2	1.0
Cu	5×10^{-2}	2.7	-3.0	1.0
Dy	2×10^{-3}	3.0	-6.2	1.1
Er	2×10^{-3}	3.0	-6.2	1.1
Es	1×10^{-3}	3.0	-6.9	1.1
Eu	2×10^{-3}	3.0	-6.2	1.1
F	2×10^{-2}	3.0	-3.9	1.1
Fe	1×10^{-3}	2.7	-6.9	1.0
Fm	2×10^{-3}	3.0	-6.2	1.1
Fr	3×10^{-2}	3.0	-3.5	1.1
Ga	3×10^{-3}	3.0	-5.8	1.1
Gd	2×10^{-3}	3.0	-6.2	1.1
Ge	4×10^{-1}	3.0	-0.9	1.1
Ha	2×10^{-3}	3.0	-6.2	1.1
Hf	3×10^{-3}	3.0	-5.8	1.1
Hg	3×10^{-1}	3.0	-1.2	1.1
Ho	2×10^{-3}	3.0	-6.2	1.1
I	2×10^{-2}	2.5	-3.9	0.9
In	3×10^{-3}	3.0	-5.8	1.1
Ir	3×10^{-2}	3.0	-3.5	1.1
K	3×10^{-1}	3.0	-1.2	1.1
La	2×10^{-3}	2.5	-6.2	0.9
Li	1×10^{-3}	3.0	-6.9	1.1
Lr	2×10^{-3}	3.0	-6.2	1.1
Lu	2×10^{-3}	3.0	-6.2	1.1
Md	2×10^{-3}	3.0	-6.2	1.1
Mg	3×10^{-2}	3.0	-3.5	1.1
Mn	3×10^{-1}	2.5	-1.2	0.9

TABLE 6.3.2 (Cont.)

Element	GM	GSD	μ	σ
Mo	1×10^{-1}	2.7	-2.3	1.0
Na	5×10^{-2}	2.7	-3.0	1.0
Nb	1×10^{-2}	2.7	-4.6	1.0
Nd	2×10^{-3}	3.0	-6.2	1.1
Ni	5×10^{-2}	2.5	-3.0	0.9
No	2×10^{-3}	2.7	-6.2	1.0
Np	2×10^{-2}	2.5	-3.9	0.9
Os	3×10^{-2}	3.0	-3.5	1.1
P	1×10^{0}	3.0	0.0	1.1
Pa	1×10^{-2}	3.0	-4.6	1.1
Pb	4×10^{-3}	2.5	-5.5	0.9
Pd	1×10^{-1}	3.0	-2.3	1.1
Pm	2×10^{-3}	3.0	-6.2	1.1
Po	1×10^{-3}	2.5	-6.9	0.9
Pr	2×10^{-3}	2.7	-6.2	1.0
Pt	1×10^{-1}	3.0	-2.3	1.1
Pu	1×10^{-3}	2.5	-6.9	0.9
Ra	4×10^{-2}	2.5	-3.2	0.9
Rb	2×10^{-1}	2.7	-1.6	1.0
Re	2×10^{-1}	3.0	-1.6	1.1
Rf	3×10^{-3}	3.0	-5.8	1.1
Rh	3×10^{-2}	2.7	-3.5	1.0
Ru	3×10^{-2}	2.5	-3.5	0.9
S	6×10^{-1}	3.0	-0.5	1.1
Sb	1×10^{-2}	2.7	-4.6	1.0
Sc	2×10^{-3}	3.0	-6.2	1.1
Se	1×10^{-1}	3.0	-2.3	1.1
Si	2×10^{-2}	3.0	-3.9	1.1
Sm	2×10^{-3}	3.0	-6.2	1.1
Sn	3×10^{-1}	3.0	-1.2	1.1
Sr	3×10^{-1}	2.7	-1.2	1.0
Ta	2×10^{-3}	3.0	-6.2	1.1
Tb	2×10^{-3}	3.0	-6.2	1.1
Tc	5×10^{0}	2.5	1.6	0.9
Te	1×10^{-1}	2.7	-2.3	1.0
Th	1×10^{-3}	2.5	-6.9	0.9
Ti	5×10^{-4}	3.0	-7.6	1.1
Tl	2×10^{-1}	3.0	-1.6	1.1
Tm	2×10^{-3}	3.0	-6.2	1.1
U	2×10^{-3}	2.5	-6.2	0.9
V	2×10^{-3}	3.0	-6.2	1.1
W	8×10^{-1}	2.7	-0.2	1.0
Y	2×10^{-3}	3.0	-6.2	1.1
Yb	2×10^{-3}	2.7	-6.2	1.0
Zn	4×10^{-1}	2.5	-0.9	0.9
Zr	1×10^{-3}	2.7	-6.9	1.0

TABLE 6.3.2 (Cont.)

Note: The values are for wet plant-to-soil transfer factor. The natural logarithm of the plant transfer factor follows a normal distribution.

Abbreviations: GM = geometric mean; GSD = geometric standard deviation; μ = mean of the underlying normal distribution; and σ = standard deviation of the underlying normal distribution.

Source: NCRP (1999).

TABLE 6.3.3 PNNL-Recommended Plant Transfer Factor Values on Dry-Weight Basis for Different Plant Types

Element	Leafy Vegetables	Fruits	Grains	Root Vegetables
Ac	4.7×10^{-4}	2.5×10^{-4}	2.2×10^{-5}	3.5×10^{-4}
Ag	2.7×10^{-4}	8.0×10^{-4}	2.5×10^{-1}	1.3×10^{-3}
Am	4.7×10^{-4}	2.5×10^{-4}	2.2×10^{-5}	3.5×10^{-4}
As	4.0×10^{-2}	6.0×10^{-3}	6.0×10^{-3}	6.0×10^{-3}
Au	1.0×10^{-2}	1.4×10^{-2}	2.5×10^{-1}	1.8×10^{-2}
Ba	1.5×10^{-1}	1.5×10^{-2}	1.5×10^{-2}	1.5×10^{-2}
Be	1.0×10^{-2}	1.5×10^{-3}	2.0×10^{-3}	1.5×10^{-3}
Bi	5.0×10^{-1}	5.0×10^{-1}	5.0×10^{-1}	5.0×10^{-1}
Br	1.5×10^{0}	1.5×10^{0}	1.5×10^{0}	1.5×10^{0}
C	7.0×10^{-1}	7.0×10^{-1}	7.0×10^{-1}	7.0×10^{-1}
Ca	3.5×10^{0}	3.5×10^{-1}	3.5×10^{-1}	3.5×10^{-1}
Cd	5.5×10^{-1}	1.5×10^{-1}	1.5×10^{-1}	1.5×10^{-1}
Ce	2.0×10^{-2}	2.0×10^{-2}	2.0×10^{-2}	2.0×10^{-2}
Cf	4.7×10^{-4}	2.5×10^{-4}	2.2×10^{-5}	3.5×10^{-4}
Cl	7.0×10^{1}	7.0×10^{1}	7.0×10^{1}	7.0×10^{1}
Cm	7.7×10^{-4}	1.5×10^{-5}	2.1×10^{-5}	4.3×10^{-4}
Co	2.3×10^{-1}	7.0×10^{-3}	3.7×10^{-3}	6.7×10^{-2}
Cr	7.5×10^{-3}	4.5×10^{-3}	4.5×10^{-3}	4.5×10^{-3}
Cs	4.6×10^{-1}	2.2×10^{-1}	2.6×10^{-2}	1.3×10^{-1}
Cu	4.0×10^{-1}	2.5×10^{-1}	2.5×10^{-1}	2.5×10^{-1}
Dy	2.0×10^{-2}	2.0×10^{-2}	2.0×10^{-2}	2.0×10^{-2}
Er	2.0×10^{-2}	2.0×10^{-2}	2.0×10^{-2}	2.0×10^{-2}
Eu	2.0×10^{-2}	2.0×10^{-2}	2.0×10^{-2}	2.0×10^{-2}
F	6.0×10^{-2}	6.0×10^{-3}	6.0×10^{-3}	6.0×10^{-3}
Fe	5.0×10^{-2}	5.0×10^{-2}	5.0×10^{-2}	5.0×10^{-2}
Ga	4.0×10^{-3}	4.0×10^{-4}	4.0×10^{-4}	4.0×10^{-4}
Gd	2.0×10^{-2}	2.0×10^{-2}	2.0×10^{-2}	2.0×10^{-2}
Hf	1.0×10^{-3}	1.0×10^{-3}	1.0×10^{-3}	1.0×10^{-3}
Hg	8.5×10^{-1}	3.7×10^{-1}	4.9×10^{-1}	2.0×10^{-1}
Ho	2.0×10^{-2}	2.0×10^{-2}	2.0×10^{-2}	2.0×10^{-2}
In	4.0×10^{-3}	4.0×10^{-4}	4.0×10^{-4}	4.0×10^{-4}
Ir	5.5×10^{-2}	1.5×10^{-2}	1.5×10^{-2}	1.5×10^{-2}

TABLE 6.3.3 (Cont.)

K	1.0×10^{0}	5.5×10^{-1}	5.5×10^{-1}	5.5×10^{-1}
La	5.2×10^{-3}	4.0×10^{-3}	4.0×10^{-3}	3.5×10^{-4}
Mg	1.0×10^{0}	5.5×10^{-1}	5.5×10^{-1}	5.5×10^{-1}
Mn	7.0×10^{-1}	5.0×10^{-2}	3.0×10^{-1}	2.0×10^{-1}
Mo	8.0×10^{-1}	5.0×10^{-2}	8.0×10^{-1}	8.0×10^{-1}
N	5.5×10^{-2}	3.0×10^{-2}	1.3×10^{-1}	4.9×10^{-2}
Na	3.0×10^{-1}	3.0×10^{-1}	3.0×10^{-1}	3.0×10^{-1}
Nb	2.5×10^{-2}	2.5×10^{-2}	2.5×10^{-2}	2.5×10^{-2}
Nd	2.0×10^{-2}	2.0×10^{-2}	2.0×10^{-2}	2.0×10^{-2}
Ni	2.8×10^{-1}	6.0×10^{-2}	3.0×10^{-2}	6.0×10^{-2}
Np	3.2×10^{-2}	1.0×10^{-2}	2.7×10^{-3}	1.3×10^{-2}
Os	1.5×10^{-2}	4.5×10^{-2}	3.5×10^{-3}	3.5×10^{-3}
P	3.5×10^{0}	3.5×10^{0}	3.5×10^{0}	3.5×10^{0}
Pa	4.7×10^{-4}	2.5×10^{-4}	2.2×10^{-5}	3.5×10^{-4}
Pb	1.0×10^{-2}	1.0×10^{-2}	4.7×10^{-3}	6.0×10^{-3}
Pd	1.5×10^{-1}	4.0×10^{-2}	4.0×10^{-2}	4.0×10^{-2}
Pm	2.0×10^{-2}	2.0×10^{-2}	2.0×10^{-2}	2.0×10^{-2}
Po	1.2×10^{-3}	1.2×10^{-3}	2.3×10^{-3}	7.0×10^{-3}
Pr	2.0×10^{-2}	2.0×10^{-2}	2.0×10^{-2}	2.0×10^{-2}
Pu	6.0×10^{-5}	4.5×10^{-5}	8.6×10^{-6}	1.1×10^{-3}
Ra	4.9×10^{-2}	6.1×10^{-3}	1.2×10^{-3}	2.0×10^{-3}
Rb	9.0×10^{-1}	9.0×10^{-1}	9.0×10^{-1}	9.0×10^{-1}
Re	1.5×10^{0}	3.5×10^{-1}	3.5×10^{-1}	3.5×10^{-1}
Rh	1.5×10^{-1}	4.0×10^{-2}	4.0×10^{-2}	4.0×10^{-2}
Ru	4.0×10^{-2}	4.0×10^{-2}	5.0×10^{-3}	4.0×10^{-2}
S	1.5×10^{0}	1.5×10^{0}	1.5×10^{0}	1.5×10^{0}
Sb	1.3×10^{-4}	8.0×10^{-5}	3.0×10^{-2}	5.6×10^{-4}
Sc	6.0×10^{-3}	1.0×10^{-3}	1.0×10^{-3}	1.0×10^{-3}
Se	2.5×10^{-1}	5.0×10^{-2}	2.5×10^{-1}	5.0×10^{-2}
Si	3.5×10^{-1}	7.0×10^{-2}	7.0×10^{-2}	7.0×10^{-2}
Sm	2.0×10^{-2}	2.0×10^{-2}	2.0×10^{-2}	2.0×10^{-2}
Sn	3.0×10^{-2}	6.0×10^{-3}	6.0×10^{-3}	6.0×10^{-3}
Sr	3.0×10^{0}	2.0×10^{-1}	2.1×10^{-1}	5.0×10^{-1}
Ta	2.5×10^{-2}	2.5×10^{-2}	2.5×10^{-2}	2.5×10^{-2}
Tb	2.0×10^{-2}	2.0×10^{-2}	2.0×10^{-2}	2.0×10^{-2}
Tc	2.1×10^{2}	1.5×10^{0}	7.3×10^{-1}	2.4×10^{-1}
Te	2.5×10^{-2}	4.0×10^{-3}	4.0×10^{-3}	4.0×10^{-3}
Th	1.8×10^{-3}	2.5×10^{-4}	3.4×10^{-5}	3.3×10^{-4}
Tl	4.0×10^{-3}	4.0×10^{-4}	4.0×10^{-4}	4.0×10^{-4}
U	8.3×10^{-3}	4.0×10^{-3}	1.3×10^{-3}	1.2×10^{-2}
W	3.0×10^{0}	3.0×10^{0}	3.0×10^{0}	3.0×10^{0}
Y	1.0×10^{-2}	1.0×10^{-2}	1.0×10^{-2}	1.0×10^{-2}
Zn	1.3×10^{0}	9.0×10^{-1}	1.6×10^{0}	3.5×10^{-1}
Zr	1.0×10^{-3}	1.0×10^{-3}	1.0×10^{-3}	1.0×10^{-3}

Note: The following dry-to-wet weight conversion factors can be used: leafy vegetables 0.2, root vegetables 0.25, fruits 0.18, and grains 0.91.

Source: Staven et al. (2003).

TABLE 6.3.4 Plant Transfer Factors on Dry-Weight Basis for Grains in Temperate Environment

	Plant Type = Cereals				Plant Type = Maize			
Element	Recommended	GSD	Minimum	Maximum	Recommended	GSD	Minimum	Maximum
Am	2.2×10^{-5}	11	7.4×10^{-7}	3.4×10^{-2}	–	–	–	–
Ba	1.0×10^{-3}	–[a]	–	–	–	–	–	–
Cd	8.8×10^{-1}	2.7	1.4×10^{-1}	2.9×10^{0}	5.0×10^{-2}	–	–	–
Ce	3.1×10^{-3}	3.7	2.4×10^{-4}	2.0×10^{-2}	–	–	–	–
Cl	3.6×10^{1}	1.6	2.0×10^{1}	8.6×10^{1}	–	–	–	–
Cm	2.3×10^{-5}	3.3	1.4×10^{-6}	2.0×10^{-4}	–	–	–	–
Co	8.5×10^{-3}	5.5	4.0×10^{-4}	6.0×10^{-2}	1.0×10^{-2}	4.1	9.0×10^{-4}	5.6×10^{-1}
Cs	2.9×10^{-2}	4.1	2.0×10^{-4}	9.0×10^{-1}	3.3×10^{-2}	3	3.0×10^{-3}	2.6×10^{-1}
I	6.3×10^{-4}	2.3	1.0×10^{-4}	1.1×10^{-2}	–	–	–	–
K	7.4×10^{-1}	–	7.3×10^{-1}	7.4×10^{-1}	–	–	–	–
La	2.0×10^{-5}	–	–	–	–	–	–	–
Mn	2.8×10^{-1}	3.3	1.4×10^{-2}	2.7×10^{0}	7.5×10^{-2}	2.1	1.8×10^{-2}	$3. \times 10^{-1}$
Mo	8.0×10^{-1}	–	–	–	–	–	–	–
Ni	2.7×10^{-2}	2.7	3.1×10^{-3}	1.7×10^{-1}	–	–	–	–
Np	2.9×10^{-3}	5	2.3×10^{-5}	7.1×10^{-2}	4.8×10^{-3}	–	1.0×10^{-4}	9.4×10^{-3}
Pb	1.1×10^{-2}	3.6	1.9×10^{-3}	4.8×10^{-2}	1.2×10^{-3}	2.3	5.2×10^{-4}	3.8×10^{-3}
Pm	1.4×10^{-2}	6	1.7×10^{-3}	2.4×10^{-1}	–	–	–	–
Po	2.4×10^{-4}	–	2.2×10^{-4}	2.6×10^{-4}	2.4×10^{-4}	–	1.8×10^{-5}	4.7×10^{-4}
Pr	2.0×10^{-2}	–	–	–	–	–	–	–
Pu	9.5×10^{-6}	6.7	2.0×10^{-7}	1.1×10^{-3}	3.0×10^{-6}	–	–	–
Ra	1.7×10^{-2}	12	8.0×10^{-5}	6.7×10^{-1}	2.4×10^{-3}	5.4	1.2×10^{-4}	1.1×10^{-1}
Rb	9.0×10^{-1}	–	–	–	–	–	–	–
Ru	3.0×10^{-3}	2.6	6.0×10^{-4}	1.0×10^{-2}	–	–	–	–
Sb	1.8×10^{-3}	2.7	3.0×10^{-4}	9.0×10^{-3}	–	–	–	–
Sr	1.1×10^{-1}	2.7	3.6×10^{-3}	1.0×10^{0}	3.2×10^{-1}	4.1	2.0×10^{-3}	2.6×10^{0}
Tc	1.3×10^{0}	–	1.8×10^{-1}	2.4×10^{0}	3.8×10^{0}	8.2	5.0×10^{-1}	5.2×10^{1}
Th	2.1×10^{-3}	3.4	1.6×10^{-4}	2.2×10^{-2}	6.4×10^{-5}	9.2	1.2×10^{-6}	1.1×10^{-2}
U	6.2×10^{-3}	7.7	1.6×10^{-4}	8.2×10^{-1}	1.5×10^{-2}	12	5.0×10^{-4}	7.1×10^{-1}
Zn	1.8×10^{0}	2.7	2.0×10^{-2}	1.4×10^{1}	5.8×10^{-1}	1.4	2.8×10^{-1}	9.1×10^{-1}
Zr	$1. \times 10^{-3}$							

Note: GSD = geometric standard deviation.

[a] Hyphen indicates value not available.

Source: TRS 472 (IAEA 2010a)

environment is not temperate, other appropriate environment-specific plant transfer factors should be used. Tables 6.3.4–6.3.8 list the recommended value (observed value in cases when only one data point is available, mean value when two data points are available, or geometric mean when more than two data points are available), along with the geometric standard deviation if applicable, minimum, and maximum value. Probability distributions were developed if more than three data points were available.

TABLE 6.3.5 Plant Transfer Factors on Dry-Weight Basis for Leafy Vegetables in Temperate Environment

Element	Recommended	GSD	Minimum	Maximum
Ag	1.8×10^{-4}	3.3	5.9×10^{-5}	1.3×10^{-3}
Am	2.7×10^{-4}	3.3	4.0×10^{-5}	1.5×10^{-3}
Ba	5.0×10^{-3}	–[a]	–	–
Ce	6.0×10^{-3}	–	–	–
Cl	2.6×10^{1}	1.7	1.4×10^{1}	4.8×10^{1}
Cm	1.4×10^{-3}	4.5	2.0×10^{-4}	8.1×10^{-3}
Co	1.7×10^{-1}	2.7	1.3×10^{-2}	1.0×10^{0}
Cs	6.0×10^{-2}	6	3.0×10^{-4}	9.8×10^{-1}
I	6.5×10^{-3}	3.7	1.1×10^{-3}	8.0×10^{-3}
K	1.3×10^{0}	–	1.2×10^{0}	1.3×10^{0}
La	5.7×10^{-3}	2.7	1.1×10^{-3}	1.5×10^{-2}
Mn	4.1×10^{-1}	2.4	5.2×10^{-2}	3.0×10^{0}
Mo	5.1×10^{-1}	–	2.1×10^{-1}	8.0×10^{-1}
Np	2.7×10^{-2}	3	5.0×10^{-3}	8.0×10^{-2}
Pb	8.0×10^{-2}	13	3.2×10^{-3}	2.5×10^{1}
Po	7.4×10^{-3}	6.9	2.5×10^{-4}	5.0×10^{-2}
Pr	2.0×10^{-2}	–	–	–
Pu	8.3×10^{-5}	2.7	1.0×10^{-5}	2.9×10^{-4}
Ra	9.1×10^{-2}	6.7	1.8×10^{-3}	1.3×10^{1}
Rb	6.2×10^{-1}	–	3.4×10^{-1}	9.0×10^{-1}
Ru	9.0×10^{-2}	3.7	2.0×10^{-2}	2.3×10^{-1}
Sb	9.4×10^{-5}	2.6	2.2×10^{-5}	2.3×10^{-4}
Sr	7.6×10^{-1}	6	3.9×10^{-3}	7.8×10^{0}
Tc	1.8×10^{2}	13.5	4.5×10^{0}	3.4×10^{3}
Th	1.2×10^{-3}	6	9.4×10^{-5}	2.1×10^{-1}
U	2.0×10^{-2}	7.3	7.8×10^{-5}	8.8×10^{0}
Zn	2.4×10^{0}	2.4	1.0×10^{-1}	1.7×10^{1}

Note: GSD = geometric standard deviation

[a] Hyphen indicates value not available.

Source: TRS 472 (IAEA 2010a)

In another recent PNNL report, Napier et al. (2014) provide transfer factors for forage, grain, fruits, and nut trees. For deriving soil-to-plant transfers, samples of stems, leaves, fruits, and nuts, along with the soils that they were growing in (Napier et al. 2014), were taken from a single farm in Nevada. The samples of alfalfa and oats were also taken. The concentrations in the samples were determined for many naturally occurring elements. Table 6.3.9 list the transfer factor values on the dry weight basis.

Table 6.3.10 compares the current plant transfer factor default values with other references. For this comparison, the values from Staven et al. (2003) were converted from the original values reported in dry weight to wet weight using factors provided in the report (Staven et al. 2003). The default values are comparable to values listed in the NCRP (1999) report. The

TABLE 6.3.6 Plant Transfer Factors on Dry-Weight Basis for Root Crops and Tubers in Temperate Environment

	Root Crops				Tubers			
Element	Recommended	GSD	Min	Max	Recommended	GSD	Min	Max
Ag	1.3×10^{-3}	2	5.7×10^{-4}	3.9×10^{-3}	–[a]	–	–	–
Am	6.7×10^{-4}	2.4	2.0×10^{-4}	1.7×10^{-3}	2.1×10^{-4}	6	1.1×10^{-5}	3.4×10^{-2}
Ba	5.0×10^{-3}	–	–	–	5.0×10^{-3}	–	–	–
Cd	–	–	–	–	1.5×10^{0}	–	–	–
Ce	6.0×10^{-3}	–	–	–	4.0×10^{-3}	–	–	–
Cl	1.2×10^{1}	1.8	4.8×10^{0}	3.6×10^{1}	–	–	–	–
Cm	8.5×10^{-4}	3	2.0×10^{-4}	3.9×10^{-3}	1.5×10^{-4}	3.7	1.1×10^{-5}	2.1×10^{-3}
Co	1.1×10^{-1}	2.2	4.7×10^{-2}	7.2×10^{-1}	5.4×10^{-2}	3	1.0×10^{-2}	6.7×10^{-1}
Cs	4.2×10^{-2}	3	1.0×10^{-3}	8.8×10^{-1}	5.6×10^{-2}	3	4.0×10^{-3}	6.0×10^{-1}
I	7.7×10^{-3}	3	1.4×10^{-3}	4.7×10^{-2}	1.0×10^{-1}	–	–	–
La	1.6×10^{-3}	2.7	4.5×10^{-4}	6.0×10^{-3}	3.9×10^{-4}	3.7	7.0×10^{-5}	4.0×10^{-3}
Mn	4.2×10^{-1}	5.5	1.5×10^{-2}	3.9×10^{0}	4.7×10^{-2}	2.2	1.2×10^{-2}	3.0×10^{-1}
Mo	3.2×10^{-1}	–	2.3×10^{-2}	4.2×10^{-1}	–	–	–	–
Np	2.2×10^{-2}	2	5.0×10^{-3}	3.6×10^{-2}	5.7×10^{-3}	2.5	7.1×10^{-4}	2.7×10^{-2}
Pb	1.5×10^{-2}	16	2.4×10^{-4}	3.3×10^{0}	1.5×10^{-3}	7.4	1.5×10^{-4}	2.6×10^{0}
Pm	4.2×10^{-2}	1.2	3.6×10^{-2}	6.0×10^{-2}	1.0×10^{-2}	1.3	7.5×10^{-3}	1.2×10^{-2}
Po	5.8×10^{-3}	4.3	2.4×10^{-4}	4.9×10^{-2}	2.7×10^{-3}	5.8	1.4×10^{-4}	3.4×10^{-2}
Pr	2.0×10^{-2}	–	–	–	–	–	–	–
Pu	3.9×10^{-4}	10	7.0×10^{-5}	5.8×10^{-3}	1.1×10^{-4}	5.5	3.8×10^{-6}	5.0×10^{-3}
Ra	7.0×10^{-2}	9.2	2.0×10^{-3}	5.6×10^{1}	1.1×10^{-2}	6.8	2.4×10^{-4}	3.9×10^{0}
Rb	9.0×10^{-1}	–	–	–	–	–	–	–
Ru	1.0×10^{-2}	–	–	–	5.0×10^{-3}	–	–	–
Sb	6.2×10^{-4}	1.5	4.0×10^{-4}	1.1×10^{-3}	2.0×10^{-3}	–	–	–
Sr	7.2×10^{-1}	4.1	3.0×10^{-2}	4.8×10^{0}	1.6×10^{-1}	3	7.4×10^{-3}	1.6×10^{0}
Tc	4.6×10^{1}	–	1.4×10^{1}	7.9×10^{1}	2.3×10^{-1}	3.7	1.3×10^{-2}	6.5×10^{-1}
Th	8.0×10^{-4}	13	8.2×10^{-6}	9.5×10^{-2}	2.0×10^{-4}	9.9	1.3×10^{-5}	1.8×10^{-2}
U	8.4×10^{-3}	6.2	4.9×10^{-4}	2.6×10^{-1}	5.0×10^{-3}	6.4	1.8×10^{-4}	8.0×10^{-2}
Y	2.0×10^{-3}	–	–	–	1.0×10^{-3}	–	–	–
Zn	–	–	–	–	3.0×10^{-1}	1.8	5.0×10^{-2}	6.3×10^{-1}
Zr	4.0×10^{-3}	–	–	–	2.0×10^{-1}	–	–	–

Note: GSD = geometric standard deviation.

[a] Hyphen indicates value not available.

Source: TRS 472 (IAEA 2010a).

latter report has the comprehensive dataset as required in the RESRAD (onsite) and RESRAD-OFFSITE codes.

TABLE 6.3.7 Plant Transfer Factors on Dry-Weight Basis for Nonleafy Vegetables and Leguminous Vegetables in Temperate Environment

	Nonleafy Vegetables				Leguminous Vegetables			
Element	Recommended	GSD	Min	Max	Recommended	GSD	Min	Max
Ag	6.4×10^{-4}	2.3	2.5×10^{-4}	2.0×10^{-3}	–[a]	–	–	–
Am	3.6×10^{-4}	5	2.3×10^{-5}	1.9×10^{-3}	3.8×10^{-4}	2.6	2.2×10^{-5}	7.9×10^{-4}
Ba	5.0×10^{-3}	–	–	–	–	–	–	–
Cd	–	–	–	–	2.7×10^{-1}	–	8.0×10^{-2}	4.6×10^{-1}
Ce	–	–	–	–	1.3×10^{-2}	–	6.0×10^{-3}	2.0×10^{-2}
Cl	–	–	–	–	1.1×10^{1}	1.3	7.0×10^{0}	1.5×10^{1}
Cm	3.2×10^{-4}	4.5	3.6×10^{-5}	1.4×10^{-3}	7.5×10^{-4}	1.5	4.2×10^{-4}	1.6×10^{-3}
Co	1.4×10^{-1}	1.6	5.7×10^{-2}	2.3×10^{-1}	3.6×10^{-2}	2.3	5.0×10^{-3}	5.0×10^{-1}
Cs	2.1×10^{-2}	4.1	7.0×10^{-4}	7.3×10^{-1}	4.0×10^{-2}	3.7	1.0×10^{-3}	7.1×10^{-1}
I	1.0×10^{-1}	–	–	–	8.5×10^{-3}	7.4	2.0×10^{-4}	1.4×10^{-1}
La	6.0×10^{-3}	–	5.9×10^{-3}	6.0×10^{-3}	4.2×10^{-4}	3	1.6×10^{-4}	1.8×10^{-3}
Mn	3.1×10^{-1}	4.1	1.0×10^{-1}	1.5×10^{0}	2.2×10^{-1}	2.5	2.2×10^{-2}	2.8×10^{0}
Np	1.8×10^{-2}	2.4	4.0×10^{-3}	5.7×10^{-2}	1.7×10^{-2}	1.8	4.0×10^{-3}	3.8×10^{-2}
Pb	1.5×10^{-2}	26	1.5×10^{-3}	3.9×10^{0}	5.3×10^{-3}	12	4.6×10^{-4}	4.9×10^{0}
Pm	–	–	–	–	1.7×10^{-1}	7.4	2.0×10^{-2}	1.2×10^{0}
Po	–	–	–	–	2.7×10^{-4}	3.9	6.0×10^{-5}	1.0×10^{-3}
Pu	6.5×10^{-5}	2.7	6.0×10^{-6}	2.0×10^{-4}	6.3×10^{-5}	1.4	3.7×10^{-5}	1.5×10^{-4}
Ra	1.7×10^{-2}	8.4	2.4×10^{-4}	6.3×10^{0}	1.4×10^{-2}	8.2	3.2×10^{-4}	6.2×10^{0}
Ru	2.0×10^{-2}	–	–	–	1.5×10^{-2}	–	1.0×10^{-2}	2.0×10^{-2}
Sb	1.3×10^{-4}	6.7	1.5×10^{-5}	1.6×10^{-3}	7.0×10^{-3}	–	–	–
Sr	3.6×10^{-1}	5.5	7.1×10^{-3}	7.9×10^{0}	1.4×10^{0}	2.3	1.3×10^{-1}	6.0×10^{0}
Tc	–	–	–	–	4.3×10^{0}	5.2	1.1×10^{0}	3.0×10^{1}
Th	7.8×10^{-4}	6.8	6.2×10^{-5}	1.6×10^{-2}	5.3×10^{-4}	9.4	2.5×10^{-5}	4.8×10^{-1}
U	1.5×10^{-2}	4.2	5.2×10^{-4}	2.0×10^{-1}	2.2×10^{-3}	12	5.4×10^{-5}	1.5×10^{-1}
Zn	4.2×10^{-1}	3.7	1.0×10^{-1}	9.5×10^{-1}	9.1×10^{-1}	2.4	2.5×10^{-1}	1.3×10^{1}
Zr	4.0×10^{-3}	–	–	–	–	–	–	–

Note: GSD = geometric standard deviation.

[a] Hyphen indicates value not available.

Source: TRS 472 (IAEA 2010a).

TABLE 6.3.8 Plant Transfer Factors on Dry-Weight Basis for Stems and Shoots in Different Plant Groups in Temperate Environment

	Cereals				Grasses				Pasture				Maize				Leguminous			
Element	GM	GSD	Min	Max	GM	GSD	Min	Max	GM	GSD	Min	Max	GM	GSD	Min	Max	GM	GSD	Min	Max
Am	7.9×10^{-5}	81.5	3.0×10^{-7}	5.8×10^{-2}	3.3×10^{-2}	9	4.2×10^{-4}	2.6×10^{-1}	1.5×10^{-3}	4.1	1.0×10^{-4}	4.8×10^{-2}	2.6×10^{-4}	5.5	1.1×10^{-5}	1.2×10^{-2}	6.5×10^{-4}	2.7	1.8×10^{-4}	3.1×10^{-3}
Ba	–[a]	–	–	–	2.0×10^{0}	1.3	1.2×10^{0}	3.6×10^{0}	–	–	–	–	–	–	–	–	9.1×10^{-1}	1	2.8×10^{-1}	2.1×10^{0}
Be	–	–	–	–	–	–	–	–	4.2×10^{-1}	–	–	–	–	–	–	–	–	–	–	–
Ca	8.7×10^{0}	3.7	2.3×10^{0}	3.8×10^{1}	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Cd	2.1×10^{0}	2.2	1.9×10^{-1}	5.4×10^{0}	–	–	–	–	–	–	–	–	1.3×10^{0}	–	3.5×10^{-1}	2.2×10^{0}	–	–	–	–
Ce	3.9×10^{-2}	5.5	3.0×10^{-3}	6.8×10^{-1}	2.0×10^{-2}	–	1.0×10^{-2}	3.0×10^{-2}	3.7×10^{-1}	5	2.0×10^{-2}	3.5×10^{0}	–	–	–	–	8.0×10^{-3}	2.1	4.0×10^{-3}	$2.\ 0 \times 10^{-2}$
Cl	3.4×10^{2}	1.5	2.1×10^{2}	6.2×10^{2}	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Cm	–	–	–	–	–	–	–	–	1.0×10^{-3}	2.4	1.0×10^{-4}	3.6×10^{-3}	2.0×10^{-4}	5	5.7×10^{-6}	4.4×10^{-3}	–	–	–	–
Co	1.1×10^{-1}	5	1.0×10^{-2}	4.9×10^{0}	7.7×10^{-2}	2.2	4.0×10^{-2}	1.7×10^{-1}	4.5×10^{-2}	3.7	2.1×10^{-3}	8.4×10^{-1}	3.5×10^{-2}	2.2	6.0×10^{-3}	2.0×10^{-1}	6.6×10^{-2}	3.3	1.0×10^{-3}	7.2×10^{-1}
Cs	1.5×10^{-1}	5	4.3×10^{-3}	3.7×10^{0}	6.3×10^{-2}	36.6	4.8×10^{-3}	9.9×10^{-1}	2.5×10^{-1}	4.1	1.0×10^{-2}	5.0×10^{0}	7.3×10^{-2}	3	3.0×10^{-3}	4.9×10^{-1}	1.6×10^{-1}	3.3	1.0×10^{-2}	1.8×10^{0}
I	5.2×10^{-2}	3.3	7.0×10^{-3}	7.5×10^{-1}	–	–	–	–	3.7×10^{-3}	6	$9. \times 10^{-4}$	5.0×10^{-1}	–	–	–	–	–	–	–	–
K	1.1×10^{0}	–	9.3×10^{-1}	1.2×10^{0}	–	–	–	–	7.3×10^{-1}	–	–	–	–	–	–	–	–	–	–	–
La	–	–	–	–	1.8×10^{-5}	2.3	6.0×10^{-6}	4.7×10^{-5}	2.0×10^{-2}	–	–	–	8.8×10^{-5}	–	7.6×10^{-5}	9.9×10^{-5}	–	–	–	–
Mn	2.2×10^{0}	4.1	2.0×10^{-1}	2.7×10^{1}	–	–	–	–	6.4×10^{-1}	1.9	1.1×10^{-1}	2.7×10^{0}	–	–	–	–	1.5×10^{0}	3.3	2.4×10^{-1}	1.2×10^{1}
Mo	–	–	–	–	–	–	–	–	–	–	–	–	7.3×10^{-1}	–	1.0×10^{0}	3.8×10^{1}	5.4×10^{0}	–	–	–
Ni	–	–	–	–	1.7×10^{-1}	2.6	1.8×10^{-2}	5.8×10^{-1}	–	–	–	–	–	–	–	–	4.0×10^{-1}	2.5	7.3×10^{-2}	2.6×10^{0}
Np	–	–	–	–	–	–	–	–	6.1×10^{-2}	2.7	1.3×10^{-2}	4.7×10^{-1}	1.9×10^{-2}	3.3	1.4×10^{-3}	1.1×10^{-1}	2.5×10^{-2}	3.3	2.0×10^{-3}	1.2×10^{-1}
Pb	2.3×10^{-2}	3.5	5.1×10^{-3}	9.6×10^{-2}	3.1×10^{-1}	1.8	1.1×10^{-1}	1.0×10^{0}	9.2×10^{-2}	4.8	2.2×10^{-3}	1.0×10^{0}	2.8×10^{-3}	6.6	6.0×10^{-4}	2.3×10^{-2}	1.6×10^{-2}	–	–	–
Pm	2.3×10^{-1}	4.1	2.2×10^{-2}	1.4×10^{0}	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Po	–	–	–	–	–	–	–	–	1.2×10^{-1}	4.2	2.2×10^{-2}	1.0×10^{0}	–	–	–	–	1.1×10^{-2}	–	2.6×10^{-5}	2.2×10^{-4}
Pu	4.4×10^{-5}	16.4	4.4×10^{-7}	9.0×10^{-4}	1.6×10^{-4}	–	5.0×10^{-5}	2.7×10^{-4}	5.5×10^{-4}	3	6.3×10^{-5}	3.9×10^{-3}	5.2×10^{-5}	2.7	2.0×10^{-6}	3.2×10^{-4}	4.9×10^{-4}	2.2	1.1×10^{-4}	2.9×10^{-3}
Ra	3.6×10^{-2}	4.8	1.6×10^{-3}	4.3×10^{-1}	1.3×10^{-1}	4	3.6×10^{-3}	1.6×10^{0}	7.1×10^{-2}	7.6	5.1×10^{-5}	1.6×10^{0}	1.8×10^{-2}	5.2	9.6×10^{-4}	8.5×10^{-2}	1.7×10^{-1}	3.1	3.4×10^{-2}	1.5×10^{0}
Ru	1.6×10^{-1}	2.7	3.0×10^{-2}	1.0×10^{0}	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Sb	2.5×10^{-2}	1.6	1.2×10^{-2}	5.3×10^{-2}	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Sr	1.1×10^{0}	2.5	1.5×10^{-1}	9.8×10^{0}	9.1×10^{-1}	1.9	2.5×10^{-1}	2.8×10^{0}	1.3×10^{0}	2.2	5.6×10^{-2}	7.3×10^{0}	7.3×10^{-1}	6	1.2×10^{-1}	3.0×10^{0}	3.7×10^{0}	1.9	1.3×10^{0}	1.8×10^{1}
Tc	–	–	–	–	–	–	–	–	7.6×10^{1}	3	7.9×10^{0}	4.7×10^{2}	6.4×10^{0}	3.3	8.4×10^{-1}	3.7×10^{1}	–	–	–	–
Th	6.1×10^{-3}	2.4	1.6×10^{-3}	3.7×10^{-2}	4.2×10^{-2}	3.1	7.4×10^{-4}	6.5×10^{-1}	9.9×10^{-2}	5.5	2.9×10^{-3}	2.7×10^{0}	1.8×10^{-3}	–	5.4×10^{-3}	3.0×10^{-3}	2.6×10^{-3}	1.6	1.5×10^{-3}	$4.\ 0 \times 10^{-3}$
U	2.7×10^{-2}	7.5	3.0×10^{-5}	3.5×10^{0}	1.7×10^{-2}	9.4	$2. \times 10^{-4}$	5.5×10^{0}	4.6×10^{-2}	5.3	1.3×10^{-3}	1.4×10^{1}	7.8×10^{-3}	14	1.6×10^{-4}	9.6×10^{-1}	1.5×10^{-2}	4.2	2.0×10^{-3}	1.6×10^{0}
Zn	5.3×10^{0}	1.7	$2.\times 10^{0}$	1.5×10^{1}	–	–	–	–	1.0×10^{0}	1.9	5.4×10^{-2}	3.2×10^{0}	5.8×10^{0}	1.8	4.5×10^{0}	7.0×10^{0}	–	–	–	–
Zr	–	–	–	–	–	–	–	–	1.0×10^{-3}	–	–	–	–	–	–	–	–	–	–	–

Note: GM = geometric mean; GSD = geometric standard deviation.

[a] Hyphen indicates value not available.

Source: TRS 472 (IAEA 2010a).

TABLE 6.3.9 Plant Transfer Factor Values on Dry-Weight Basis for Different Plant Types from Naper et al. (2014)

Element	Forage	Grain	Leaf–Fruit	Stem– Fruit	Fruit	Leaf–Nut	Stem–Nut	Nut
Al	0.002	0.004	0.007	0.002	0.001	0.005	0.0026	0.0021
As	0.028	NA[a]	0.083	0.036	0.034	0.1035	0.0875	0.0238
Ba	0.007	0.005	0.013	0.006	0.004	0.0129	0.0073	0.005
Br	0.002	0.004	0.002	0.001	0.001	0.0018	0.0003	0.0002
Ca	0.495	0.108	0.759	0.595	0.18	1.0183	0.6418	0.0689
Ce	0.002	0.002	0.008	0.002	0.001	0.0061	0.0035	0.0024
Cl	0.514	0.302	0.145	0.102	0.103	0.2754	0.0364	0.0457
Co	0.027	0.022	0.021	0.009	0.006	0.0164	0.0088	0.0052
Cr	0.01	0.025	0.05	0.016	0.011	0.0402	0.019	0.0128
Cs	0.003	0.003	0.01	0.003	0.002	0.0095	0.0043	0.0035
Eu	0.001	0.002	0.008	0.002	0.002	0.0067	0.0045	0.0033
Fe	0.005	0.004	0.015	0.004	0.002	0.0127	0.0059	0.0035
Hf	0.001	0.002	0.008	0.002	0.001	0.0057	0.0029	0.0033
K	1.073	0.278	0.548	0.355	0.557	0.6372	0.1892	0.5701
La	0.002	0.002	0.007	0.002	0.001	0.0055	0.0031	0.0016
Lu	NA	NA	0.007	0.002	0.001	0.0068	0.0046	0.0033
Mg	0.342	0.231	0.334	0.161	0.133	0.532	0.1052	0.0989
Mn	0.098	0.149	0.103	0.045	0.023	0.0802	0.0358	0.0139
Na	0.041	0.044	0.015	0.036	0.017	0.043	0.0103	0.0095
Na	0.042	0.047	0.018	0.039	0.018	0.0463	0.0102	0.0088
Nd	NA	NA	0.019	NA	NA	0.0095	0.007	NA
Ni	0.102	0.309	0.292	0.165	0.269	0.2261	0.1721	0.2974
Rb	0.124	0.052	0.108	0.07	0.145	0.0842	0.0283	0.1012
Sb	0.074	0.043	0.047	0.126	0.096	0.0371	0.4106	0.0964
Sc	0.002	0.003	0.013	0.003	0.001	0.0114	0.0051	0.0026
Sm	0.002	0.002	0.009	0.002	0.001	0.0072	0.004	0.0027
Sr	0.277	0.049	0.264	0.232	0.068	0.3177	0.2944	0.0277
Ta	0.003	0.002	0.009	0.003	0.002	0.0067	0.0037	0.0044
Tb	NA	NA	0.009	0.002	NA	0.0063	0.008	0.0043
Th	0.001	0.002	0.008	0.002	0.0009	0.0065	0.0031	0.0026
Ti	NA	NA	NA	NA	NA	0.0054	NA	0.0035
U	NA	NA	0.014	0.003	NA	0.0114	0.0068	0.0042
V	NA	NA	0.026	0.019	NA	0.0136	0.0145	0.0111
Yb	NA	NA	0.016	0.005	NA	0.0121	0.0092	0.009
Zn	0.159	0.211	0.169	0.167	0.137	0.1411	0.0845	0.1476
Zr	NA	NA	0.014	NA	NA	0.0069	0.019	0.006

[a] NA = value not available.

Source: Naper et al. (2014)

TABLE 6.3.10 Comparison of RESRAD Default Plant Transfer Factors (Fresh-Weight Basis) with Those from Other References

			PNNL Fresh Weight (Staven et al. 2003)[a]			
Element	RESRAD (default)	NCRP (1999)	Leafy Vegetables	Fruits	Grains	Root Vegetables
Ac	2.5×10^{-3}	1.0×10^{-3}	9.4×10^{-5}	4.5×10^{-5}	2.0×10^{-5}	8.8×10^{-5}
Ag	1.5×10^{-1}	4.0×10^{-3}	5.4×10^{-5}	1.4×10^{-4}	2.3×10^{-1}	3.3×10^{-4}
Al	4.0×10^{-3}	4.0×10^{-3}	NA[b]	NA	NA	NA
Am	1.0×10^{-3}	1.0×10^{-3}	9.4×10^{-5}	4.5×10^{-5}	2.0×10^{-5}	8.8×10^{-5}
As	8.0×10^{-2}	8.0×10^{-2}	8.0×10^{-3}	1.1×10^{-3}	5.5×10^{-3}	1.5×10^{-3}
At	2.0×10^{-1}	2.0×10^{-1}	NA	NA	NA	NA
Au	1.0×10^{-1}	1.0×10^{-1}	2.0×10^{-3}	2.5×10^{-3}	2.3×10^{-1}	4.5×10^{-3}
B	NA	1.0×10^{-2}	NA	NA	NA	NA
Ba	5.0×10^{-3}	1.0×10^{-2}	3.0×10^{-2}	2.7×10^{-3}	1.4×10^{-2}	3.8×10^{-3}
Be	4.0×10^{-3}	4.0×10^{-3}	2.0×10^{-3}	2.7×10^{-4}	1.8×10^{-3}	3.8×10^{-4}
Bi	1.0×10^{-1}	1.0×10^{-1}	1.0×10^{-1}	9.0×10^{-2}	4.6×10^{-1}	1.3×10^{-1}
Bk	1.0×10^{-3}	1.0×10^{-3}	NA	NA	NA	NA
Br	7.6×10^{-1}	4.0×10^{-1}	3.0×10^{-1}	2.7×10^{-1}	1.4×10^{0}	3.8×10^{-1}
C	5.5×10^{0}	NA	1.4×10^{-1}	1.3×10^{-1}	6.4×10^{-1}	1.8×10^{-1}
Ca	5.0×10^{-1}	5.0×10^{-1}	7.0×10^{-1}	6.3×10^{-2}	3.2×10^{-1}	8.8×10^{-2}
Cd	3.0×10^{-1}	5.0×10^{-1}	1.1×10^{-1}	2.7×10^{-2}	1.4×10^{-1}	3.8×10^{-2}
Ce	2.0×10^{-3}	2.0×10^{-3}	4.0×10^{-3}	3.6×10^{-3}	1.8×10^{-2}	5.0×10^{-3}
Cf	1.0×10^{-3}	1.0×10^{-3}	9.4×10^{-5}	4.5×10^{-5}	2.0×10^{-5}	8.8×10^{-5}
Cl	2.0×10^{1}	2.0×10^{1}	1.4×10^{1}	1.3×10^{1}	6.4×10^{1}	1.8×10^{1}
Cm	1.0×10^{-3}	1.0×10^{-3}	1.5×10^{-4}	2.7×10^{-6}	1.9×10^{-5}	1.1×10^{-4}
Co	8.0×10^{-2}	8.0×10^{-2}	4.6×10^{-2}	1.3×10^{-3}	3.4×10^{-3}	1.7×10^{-2}
Cr	2.5×10^{-4}	1.0×10^{-2}	1.5×10^{-3}	8.1×10^{-4}	4.1×10^{-3}	1.1×10^{-3}
Cs	4.0×10^{-2}	4.0×10^{-2}	9.2×10^{-2}	4.0×10^{-2}	2.4×10^{-2}	3.3×10^{-2}
Cu	1.3×10^{-1}	5.0×10^{-2}	8.0×10^{-2}	4.5×10^{-2}	2.3×10^{-1}	6.3×10^{-2}
Dy	2.0×10^{-3}	2.0×10^{-3}	4.0×10^{-3}	3.6×10^{-3}	1.8×10^{-2}	5.0×10^{-3}
Er	2.0×10^{-3}	2.0×10^{-3}	4.0×10^{-3}	3.6×10^{-3}	1.8×10^{-2}	5.0×10^{-3}
Es	1.0×10^{-3}	1.0×10^{-3}	NA	NA	NA	NA
Eu	2.5×10^{-3}	2.0×10^{-3}	4.0×10^{-3}	3.6×10^{-3}	1.8×10^{-2}	$5. \times 10^{-3}$
F	2.0×10^{-2}	2.0×10^{-2}	1.2×10^{-2}	1.1×10^{-3}	5.5×10^{-3}	1.5×10^{-3}
Fe	1.0×10^{-3}	1.0×10^{-3}	1.0×10^{-2}	9.0×10^{-3}	4.6×10^{-2}	1.3×10^{-2}
Fm	2.0×10^{-3}	2.0×10^{-3}	NA	NA	NA	NA
Fr	3.0×10^{-2}	3.0×10^{-2}	NA	NA	NA	NA
Ga	3.0×10^{-3}	3.0×10^{-3}	8.0×10^{-4}	7.2×10^{-5}	3.6×10^{-4}	1.0×10^{-4}
Gd	2.5×10^{-3}	2.0×10^{-3}	4.0×10^{-3}	3.6×10^{-3}	1.8×10^{-2}	5.0×10^{-3}
Ge	4.0×10^{-1}	4.0×10^{-1}	NA	NA	NA	NA
H	4.8×10^{0}	NA	NA	NA	NA	NA
Ha	NA	2.0×10^{-3}	NA	NA	NA	NA
Hf	3.0×10^{-3}	3.0×10^{-3}	2.0×10^{-4}	1.8×10^{-4}	9.1×10^{-4}	2.5×10^{-4}
Hg	3.8×10^{-1}	3.0×10^{-1}	1.7×10^{-1}	6.7×10^{-2}	4.5×10^{-1}	5.0×10^{-2}
Ho	2.6×10^{-3}	2.0×10^{-3}	4.0×10^{-3}	3.6×10^{-3}	1.8×10^{-2}	5.0×10^{-3}
I	2.0×10^{-2}	2.0×10^{-2}	NA	NA	NA	NA
In	3.0×10^{-3}	3.0×10^{-3}	8.0×10^{-4}	7.2×10^{-5}	3.6×10^{-4}	1.0×10^{-4}
Ir	3.0×10^{-2}	3.0×10^{-2}	8.0×10^{-3}	7.2×10^{-3}	3.6×10^{-2}	1.0×10^{-2}

TABLE 6.3.10 (Cont.)

			PNNL Fresh Weight (Staven et al. 2003)			
Element	RESRAD (default)	NCRP (1999)	Leafy Vegetables	Fruits	Grains	Root Vegetables
K	3.0×10^{-1}	3.0×10^{-1}	2.0×10^{-1}	9.9×10^{-2}	5.0×10^{-1}	1.4×10^{-1}
La	2.5×10^{-3}	2.0×10^{-3}	1.0×10^{-3}	7.2×10^{-4}	3.6×10^{-3}	8.8×10^{-5}
Li	NA	1.0×10^{-3}	NA	NA	NA	NA
Lr	NA	2.0×10^{-3}	NA	NA	NA	NA
Lu	2.0×10^{-3}	2.0×10^{-3}	NA	NA	NA	NA
Md	2.0×10^{-3}	2.0×10^{-3}	NA	NA	NA	NA
Mg	3.0×10^{-2}	3.0×10^{-2}	2.0×10^{-1}	9.9×10^{-2}	5.0×10^{-1}	1.4×10^{-1}
Mn	3.0×10^{-1}	3.0×10^{-1}	1.4×10^{-1}	9.0×10^{-3}	2.7×10^{-1}	5.0×10^{-2}
Mo	1.3×10^{-1}	1.0×10^{-1}	1.6×10^{-1}	9.0×10^{-3}	7.3×10^{-1}	$2.\ 0\times 10^{-1}$
N	7.5×10^{0}	NA	1.1×10^{-2}	5.4×10^{-3}	1.2×10^{-1}	1.2×10^{-2}
Na	5.0×10^{-2}	5.0×10^{-2}	6.0×10^{-2}	5.4×10^{-2}	2.7×10^{-1}	7.5×10^{-2}
Nb	1.0×10^{-2}	1.0×10^{-2}	5.0×10^{-3}	4.5×10^{-3}	2.3×10^{-2}	6.3×10^{-3}
Nd	2.4×10^{-3}	2.0×10^{-3}	4.0×10^{-3}	3.6×10^{-3}	1.8×10^{-2}	5.0×10^{-3}
Ni	5.0×10^{-2}	5.0×10^{-2}	5.6×10^{-2}	1.1×10^{-2}	2.7×10^{-2}	1.5×10^{-2}
No	NA	2.0×10^{-3}	NA	NA	NA	NA
Np	2.0×10^{-2}	2.0×10^{-2}	6.4×10^{-3}	1.8×10^{-3}	2.5×10^{-3}	3.3×10^{-3}
O	6.0×10^{-1}	NA	NA	NA	NA	NA
Os	3.0×10^{-2}	3.0×10^{-2}	3.0×10^{-3}	8.1×10^{-3}	3.2×10^{-3}	8.8×10^{-4}
P	1.0×10^{0}	1.0×10^{0}	7.0×10^{-1}	6.3×10^{-1}	3.2×10^{0}	8.8×10^{-1}
Pa	1.0×10^{-2}	1.0×10^{-2}	9.4×10^{-5}	4.5×10^{-5}	2.0×10^{-5}	8.8×10^{-5}
Pb	1.0×10^{-2}	4.0×10^{-3}	2.0×10^{-3}	1.8×10^{-3}	4.3×10^{-3}	1.5×10^{-3}
Pd	1.0×10^{-1}	1.0×10^{-1}	3.0×10^{-2}	7.2×10^{-3}	3.6×10^{-2}	1.0×10^{-2}
Pm	2.5×10^{-3}	2.0×10^{-3}	4.0×10^{-3}	3.6×10^{-3}	1.8×10^{-2}	5.0×10^{-3}
Po	1.0×10^{-3}	1.0×10^{-3}	2.4×10^{-4}	2.2×10^{-4}	2.1×10^{-3}	1.8×10^{-3}
Pr	2.5×10^{-3}	2.0×10^{-3}	4.0×10^{-3}	3.6×10^{-3}	1.8×10^{-2}	5.0×10^{-3}
Pt	1.0×10^{-1}	1.0×10^{-1}	NA	NA	NA	NA
Pu	1.0×10^{-3}	1.0×10^{-3}	1.2×10^{-5}	8.1×10^{-6}	7.8×10^{-6}	2.8×10^{-4}
Ra	4.0×10^{-2}	4.0×10^{-2}	9.8×10^{-3}	1.1×10^{-3}	1.1×10^{-3}	5.0×10^{-4}
Rb	1.3×10^{-1}	2.0×10^{-1}	1.8×10^{-1}	1.6×10^{-1}	8.2×10^{-1}	2.3×10^{-1}
Re	2.0×10^{-1}	2.0×10^{-1}	3.0×10^{-1}	6.3×10^{-2}	3.2×10^{-1}	8.8×10^{-2}
Rf	NA	3.0×10^{-3}	NA	NA	NA	NA
Rh	1.3×10^{-1}	3.0×10^{-2}	3.0×10^{-2}	7.2×10^{-3}	3.6×10^{-2}	1.0×10^{-2}
Ru	3.0×10^{-2}	3.0×10^{-2}	8.0×10^{-3}	7.2×10^{-3}	4.6×10^{-3}	1.0×10^{-2}
S	6.0×10^{-1}	6.0×10^{-1}	3.0×10^{-1}	2.7×10^{-1}	1.4×10^{0}	3.8×10^{-1}
Sb	1.0×10^{-2}	1.0×10^{-2}	2.6×10^{-5}	1.4×10^{-5}	2.7×10^{-2}	1.4×10^{-4}
Sc	2.0×10^{-3}	2.0×10^{-3}	1.2×10^{-3}	1.8×10^{-4}	9.1×10^{-4}	2.5×10^{-4}
Se	1.0×10^{-1}	1.0×10^{-1}	5.0×10^{-2}	9.0×10^{-3}	2.3×10^{-1}	1.3×10^{-2}
Si	2.0×10^{-2}	2.0×10^{-2}	7.0×10^{-2}	1.3×10^{-2}	6.4×10^{-2}	1.8×10^{-2}
Sm	2.5×10^{-3}	2.0×10^{-3}	4.0×10^{-3}	3.6×10^{-3}	1.8×10^{-2}	5.0×10^{-3}
Sn	2.5×10^{-3}	3.0×10^{-1}	6.0×10^{-3}	1.1×10^{-3}	5.5×10^{-3}	1.5×10^{-3}
Sr	3.0×10^{-1}	3.0×10^{-1}	6.0×10^{-1}	3.6×10^{-2}	1.9×10^{-1}	1.3×10^{-1}
Ta	2.0×10^{-2}	2.0×10^{-3}	5.0×10^{-3}	4.5×10^{-3}	2.3×10^{-2}	6.3×10^{-3}
Tb	2.6×10^{-3}	2.0×10^{-3}	4.0×10^{-3}	3.6×10^{-3}	1.8×10^{-2}	5.0×10^{-3}
Tc	5.0×10^{0}	5.0×10^{0}	4.2×10^{1}	2.7×10^{-1}	6.6×10^{-1}	6.0×10^{-2}
Te	6.0×10^{-1}	1.0×10^{-1}	5.0×10^{-3}	7.2×10^{-4}	3.6×10^{-3}	1.0×10^{-3}

TABLE 6.3.10 (Cont.)

			PNNL Fresh Weight (Staven et al. 2003)			
Element	RESRAD (default)	NCRP (1999)	Leafy Vegetables	Fruits	Grains	Root Vegetables
Th	1.0×10^{-3}	1.0×10^{-3}	3.6×10^{-4}	4.5×10^{-5}	3.1×10^{-5}	8.3×10^{-5}
Ti	1.0×10^{-3}	5.0×10^{-4}	NA	NA	NA	NA
Tl	2.0×10^{-1}	2.0×10^{-1}	8.0×10^{-4}	7.2×10^{-5}	3.6×10^{-4}	1.0×10^{-4}
Tm	2.0×10^{-3}	2.0×10^{-3}	NA	NA	NA	NA
U	2.5×10^{-3}	2.0×10^{-3}	1.7×10^{-3}	7.2×10^{-4}	1.2×10^{-3}	3.0×10^{-3}
V	2.0×10^{-3}	2.0×10^{-3}	NA	NA	NA	NA
W	1.8×10^{-2}	8.0×10^{-1}	6.0×10^{-1}	5.4×10^{-1}	2.7×10^{0}	7.5×10^{-1}
Y	2.5×10^{-3}	2.0×10^{-3}	2.0×10^{-3}	1.8×10^{-3}	9.1×10^{-3}	2.5×10^{-3}
Yb	2.0×10^{-3}	2.0×10^{-3}	NA	NA	NA	NA
Zn	4.0×10^{-1}	4.0×10^{-1}	2.6×10^{-1}	1.6×10^{-1}	1.5×10^{0}	8.8×10^{-2}
Zr	1.0×10^{-3}	1.0×10^{-3}	2.0×10^{-4}	1.8×10^{-4}	9.1×10^{-4}	2.5×10^{-4}

[a] PNNL fresh weight values were obtained by converting the original values reported in the dry weight to wet weight basis using conversion factors provided in Staven et al. (2003).

[b] NA = value not available.

6.4 MEAT TRANSFER FACTORS

6.4.1 Definition

The radionuclide transfer factor for meat is the equilibrium ratio of the concentration of a radionuclide in fresh meat (pCi/kg) to the rate of daily dietary intake of that radionuclide (pCi/d) by the meat animal. It is reported that this transfer factor is perhaps the least well documented in the literature because of the obvious practical difficulty: the need to sacrifice the meat-producing animals to collect the required experimental data (IAEA 1982).

6.4.2 Discussion

There are three routes of contamination intake in animals: absorption through skin, inhalation, and ingestion. The most important transfer pathways of contaminants to animals are through the ingestion of contaminated feed, soil, and contaminated water. The concentration of radionuclides in animal products depends on the rate of food intake, gastro-intestinal absorption and turnover in tissues. Absorption of most essential elements is controlled by dietary supply and the animal's requirement and, in some instances, interaction with other essential elements such as calcium, phosphorous, iron, sodium, and zinc (IAEA 2009, Robertson et al. 2003). Table 6.4.1 lists the grouping of fractional absorption values for different elements in ruminants. After absorption, radionuclides enter the circulatory systems and are distributed to various tissues

TABLE 6.4.1 Fractional Absorption for Different Elements in Ruminants (e.g., Cow, Beef Cattle, Goat)

Fractional Absorption	Element
0.1–1	I, Cl, Na, Cs, P, Se, Ca, Te, Zn, Sr, Fe
0.01–0.1	Ag, Ba, Co, Pb, U
0.00–0.01	Mn, Ru, Cd, Y
0.0001–0.001	Zr, Ce, Pm, Am, Nb
0.00001–0.0001	Pu

of the body. The site of deposition in many cases depends on the biological role of the corresponding stable element or analogue.

For many elements and/or radionuclides, the meat transfer factor is derived from other sources, such as stable element concentrations in feed and animal tissues, extrapolations from single-dose tracer experiments, and comparison of elemental concentrations in associated or unassociated meat, or milk, and feed (Staven et al. 2003). Some of the difficulties in deriving the beef/feed transfer factor include the following:

- *Intake estimation*: The relative proportion of grass, grain, and other dietary constituents is important in estimating the dietary intake. The dietary composition varies according to the feeding strategies (indoors or grazing) and with season. Variability will be less in experimental studies than for field studies (IAEA 2009).

- *The need for equilibrium*: With a few exceptions, the time required for a radionuclide to reach equilibrium in many animal products (e.g., beef) is so long that few experiments can be continued long enough to approach equilibrium conditions (IAEA 2010a). Hence, a transfer factor derived from comparatively short experiments will underestimate the equilibrium transfer factor.

- *Effect of chemical and physical forms of radionuclide and composition of diet:* The availability of a radionuclide for gut uptake differs markedly, depending on the chemical and physical forms of the radionuclide and on the constituents of the diet (IAEA 2009, 2010a). Higher radionuclide concentrations are often found in tissues other than muscle, particularly liver (e.g., Pu, Am, Co, Ag, Ce, Tc) and bone (e.g., Pu, Am, Ce, Sr) (IAEA 2009). Radionuclide transfer models often underestimate soil adhesion on vegetation ingested by animals. The extent of soil ingestion will be influenced by the species of animal, season, soil type, stocking rates, and pasture management. Consequently, values for soil ingestion will be highly site specific.

- *Influence of age*: The intake of radionuclides by an animal is dependent on the animal's species, mass, age, and growth rate, as well as on the digestibility of the feed. Young animals often have enhanced gut uptake and, hence, higher transfer coefficients than adults. Few available transfer coefficient data take these factors into account.

6.4.3 Data Input Requirements

This parameters is used when the meat ingestion pathway is active. The default transfer factors in the RESRAD (onsite) and RESRAD-OFFSITE codes are for beef. If the human food source is other than beef, the transfer factors should be changed accordingly.

The default values of meat transfer factors compiled from multiple sources were reviewed in 1993 (Wang el al. 1993) and were used as default values in the RESRAD (onsite) code. Since the publication of that report, a number of new publications on transfer factors have become available. The probabilistic dose analysis capability was developed for the RESRAD (onsite) code in 2000 (Yu et al. 2000), and a meat transfer factor distribution was developed using data from NCRP (1999). The other published radiological assessments used for comparison of meat transfer factors in this handbook are Staven et al. (2003), and IAEA (2009, 2010a).

To select values for meat transfer factors, the most recent and comprehensive references should be given first priority. The first reference chosen is the IAEA Technical Report Series No. 472, *Handbook of Parameter Values for the Prediction of Radionuclide Transfer in Terrestrial and Freshwater Environments* (IAEA 2010a) and its supporting document (IAEA 2009), which includes the full compilation of reviewed data and the methods used to obtain the data values. The data included in the IAEA (2010a) report relate mainly to equilibrium conditions. The second reference used is the PNNL report, *A Compendium of Transfer Factors for Agricultural and Animal Products* (Staven et al. 2003). The transfer factors in this report are used in the GENII system of computer codes. The transfer parameters were selected from recommended values listed by national or international organizations and if no reference document was available, the values were derived based on chemical groupings in the periodic table of the elements. NCRP Report No. 129 (NCRP 1999), *Recommended Screening Limits for Contaminated Surface Soil and Review of Factors Relevant to Site-Specific Studies*, was chosen because of its general acceptance in the dose assessment community.

Table 6.4.2 compares the current RESRAD (onsite) meat (beef) transfer factor default values with those from other references. In general, elements with higher fractional absorption (Table 6.4.1) tend to have higher meat transfer factors. The IAEA handbook of parameter values (IAEA 2010a) also provides the meat transfer factors for goat meat, mutton, pork, poultry, and egg content. Pork and egg transfer factors are also provided in the PNNL report on transfer factors (Staven et al. 2003). Table 6.4.3 compares these values for different types of meat.

TABLE 6.4.2 Comparison of RESRAD (onsite) Meat (Beef) Transfer Factors with Those from Other References

Element	RESRAD (onsite)	PNNL (Staven et al. 2003)	NCRP (1999)	IAEA (2010)
Ac	2.0×10^{-5}	4.0×10^{-4}	2.0×10^{-5}	NA
Ag	3.0×10^{-3}	3.0×10^{-3}	3.0×10^{-3}	NA
Al	5.0×10^{-4}	NA[a]	5.0×10^{-4}	NA
Am	5.0×10^{-5}	4.0×10^{-5}	5.0×10^{-5}	5.0×10^{-4}
As	1.5×10^{-3}	2.0×10^{-3}	2.0×10^{-2}	NA
At	1.0×10^{-2}	NA	1.0×10^{-2}	NA
Au	5.0×10^{-3}	5.0×10^{-3}	5.0×10^{-3}	NA
B	NA	NA	8.0×10^{-4}	NA
Ba	2.0×10^{-4}	2.0×10^{-4}	2.0×10^{-4}	1.4×10^{-4}
Be	1.0×10^{-3}	1.0×10^{-3}	5.0×10^{-3}	NA
Bi	2.0×10^{-3}	4.0×10^{-4}	2.0×10^{-3}	NA
Bk	2.0×10^{-5}	NA	2.0×10^{-5}	NA
Br	2.0×10^{-2}	2.5×10^{-2}	5.0×10^{-2}	NA
C	3.1×10^{-2}	NA	NA	NA
Ca	1.6×10^{-3}	2.0×10^{-3}	2.0×10^{-3}	1.3×10^{-2}
Cd	4.0×10^{-4}	4.0×10^{-4}	1.0×10^{-3}	5.8×10^{-3}
Ce	2.0×10^{-5}	2.0×10^{-5}	2.0×10^{-5}	NA
Cf	6.0×10^{-5}	4.0×10^{-5}	6.0×10^{-5}	NA
Cl	6.0×10^{-2}	2.0×10^{-2}	4.0×10^{-2}	1.7×10^{-2}
Cm	2.0×10^{-5}	4.0×10^{-5}	2.0×10^{-5}	NA
Co	2.0×10^{-2}	1.0×10^{-2}	3.0×10^{-2}	4.3×10^{-4}
Cr	9.0×10^{-3}	9.0×10^{-3}	3.0×10^{-2}	NA
Cs	3.0×10^{-2}	5.0×10^{-2}	5.0×10^{-2}	2.2×10^{-2}
Cu	1.0×10^{-2}	9.0×10^{-3}	1.0×10^{-2}	NA
Dy	2.0×10^{-3}	2.0×10^{-5}	2.0×10^{-3}	NA
Er	2.0×10^{-3}	2.0×10^{-5}	2.0×10^{-3}	NA
Es	2.0×10^{-5}	NA	2.0×10^{-5}	NA
Eu	2.0×10^{-3}	2.0×10^{-5}	2.0×10^{-3}	NA
F	2.0×10^{-2}	NA	2.0×10^{-2}	NA
Fe	2.0×10^{-2}	2.0×10^{-2}	3.0×10^{-2}	1.4×10^{-2}
Fm	2.0×10^{-4}	NA	2.0×10^{-4}	NA
Fr	3.0×10^{-2}	NA	3.0×10^{-2}	NA
Ga	3.0×10^{-4}	5.0×10^{-4}	3.0×10^{-4}	NA
Gd	2.0×10^{-3}	2.0×10^{-5}	2.0×10^{-3}	NA
Ge	2.0×10^{-1}	NA	2.0×10^{-1}	NA
H	1.2×10^{-2}	NA	NA	NA
Ha	NA	NA	5.0×10^{-6}	NA
Hf	4.0×10^{-4}	1.0×10^{-3}	4.0×10^{-4}	NA
Hg	1.0×10^{-1}	2.5×10^{-1}	1.0×10^{-2}	NA
Ho	2.0×10^{-3}	2.0×10^{-5}	2.0×10^{-3}	NA
I	7.0×10^{-3}	4.0×10^{-2}	4.0×10^{-2}	6.7×10^{-3}
In	4.0×10^{-3}	8.0×10^{-3}	4.0×10^{-3}	NA
Ir	2.0×10^{-3}	1.5×10^{-3}	2.0×10^{-3}	NA
K	2.0×10^{-2}	2.0×10^{-2}	2.0×10^{-2}	NA
La	2.0×10^{-3}	2.0×10^{-3}	2.0×10^{-3}	1.3×10^{-4}

TABLE 6.4.2 (Cont.)

Element	RESRAD (onsite)	PNNL (Staven et al. 2003)	NCRP (1999)	IAEA (2010)
Li	NA	NA	2.0×10^{-2}	NA
Lr	NA	NA	2.0×10^{-4}	NA
Lu	2.0×10^{-3}	NA	2.0×10^{-3}	NA
Md	2.0×10^{-4}	NA	2.0×10^{-4}	NA
Mg	3.0×10^{-3}	2.0×10^{-2}	3.0×10^{-3}	NA
Mn	5.0×10^{-4}	5.0×10^{-4}	1.0×10^{-3}	6.0×10^{-4}
Mo	1.0×10^{-3}	1.0×10^{-3}	1.0×10^{-3}	1.0×10^{-3}
N	1.0×10^{-2}	7.5×10^{-2}	NA	NA
Na	8.0×10^{-2}	8.0×10^{-2}	8.0×10^{-2}	1.5×10^{-2}
Nb	3.0×10^{-7}	3.0×10^{-7}	1.0×10^{-6}	2.6×10^{-7}
Nd	2.0×10^{-3}	2.0×10^{-5}	2.0×10^{-3}	NA
Ni	5.0×10^{-3}	5.0×10^{-3}	5.0×10^{-3}	NA
No	NA	NA	2.0×10^{-4}	NA
Np	1.0×10^{-3}	1.0×10^{-3}	1.0×10^{-3}	NA
O	2.0×10^{-1}	NA	NA	NA
Os	2.0×10^{-3}	4.0×10^{-1}	2.0×10^{-3}	NA
P	5.0×10^{-2}	5.0×10^{-2}	5.0×10^{-2}	5.5×10^{-2}
Pa	5.0×10^{-3}	4.0×10^{-5}	5.0×10^{-6}	NA
Pb	8.0×10^{-4}	4.0×10^{-4}	8.0×10^{-4}	7.0×10^{-4}
Pd	1.0×10^{-3}	4.0×10^{-3}	2.0×10^{-4}	NA
Pm	2.0×10^{-3}	2.0×10^{-5}	2.0×10^{-3}	NA
Po	5.0×10^{-3}	5.0×10^{-3}	5.0×10^{-3}	NA
Pr	2.0×10^{-3}	2.0×10^{-5}	2.0×10^{-3}	NA
Pt	2.0×10^{-4}	NA	2.0×10^{-4}	NA
Pu	1.0×10^{-4}	1.0×10^{-5}	1.0×10^{-4}	1.1×10^{-6}
Ra	1.0×10^{-3}	9.0×10^{-4}	1.0×10^{-3}	1.7×10^{-3}
Rb	1.5×10^{-2}	1.0×10^{-2}	3.0×10^{-2}	NA
Re	1.0×10^{-2}	8.0×10^{-3}	1.0×10^{-2}	NA
Rf	NA	NA	4.0×10^{-4}	NA
Rh	1.0×10^{-3}	2.0×10^{-3}	2.0×10^{-3}	NA
Ru	2.0×10^{-3}	5.0×10^{-2}	2.0×10^{-3}	3.3×10^{-3}
S	2.0×10^{-1}	2.0×10^{-1}	2.0×10^{-1}	NA
Sb	1.0×10^{-3}	1.0×10^{-3}	1.0×10^{-3}	1.2×10^{-3}
Sc	1.5×10^{-2}	1.5×10^{-2}	2.0×10^{-3}	NA
Se	1.0×10^{-1}	1.5×10^{-2}	1.0×10^{-1}	NA
Si	3.0×10^{-4}	4.0×10^{-5}	3.0×10^{-4}	NA
Sm	2.0×10^{-3}	2.0×10^{-5}	2.0×10^{-3}	NA
Sn	1.0×10^{-2}	8.0×10^{-2}	1.0×10^{-2}	NA
Sr	8.0×10^{-3}	8.0×10^{-3}	1.0×10^{-2}	1.3×10^{-3}
Ta	5.0×10^{-6}	3.0×10^{-7}	5.0×10^{-6}	NA
Tb	2.0×10^{-3}	2.0×10^{-5}	2.0×10^{-3}	NA
Tc	1.0×10^{-4}	1.0×10^{-4}	1.0×10^{-4}	NA
Te	7.0×10^{-3}	7.0×10^{-3}	7.0×10^{-3}	7.0×10^{-3}
Th	1.0×10^{-4}	4.0×10^{-5}	1.0×10^{-4}	2.3×10^{-4}
Ti	2.0×10^{-2}	NA	2.0×10^{-2}	NA
Tl	2.0×10^{-2}	4.0×10^{-2}	2.0×10^{-2}	NA
Tm	2.0×10^{-3}	NA	2.0×10^{-3}	NA

TABLE 6.4.2 (Cont.)

Element	RESRAD (onsite)	PNNL (Staven et al. 2003)	NCRP (1999)	IAEA (2010)
U	3.4×10^{-4}	3.0×10^{-4}	8.0×10^{-4}	3.9×10^{-4}
V	1.0×10^{-2}	NA	1.0×10^{-2}	NA
W	4.0×10^{-2}	4.0×10^{-2}	4.0×10^{-2}	NA
Y	2.0×10^{-3}	1.0×10^{-3}	2.0×10^{-3}	NA
Yb	2.0×10^{-3}	NA	2.0×10^{-3}	NA
Zn	1.0×10^{-1}	1.0×10^{-1}	1.0×10^{-1}	1.6×10^{-1}
Zr	1.0×10^{-6}	1.0×10^{-6}	1.0×10^{-6}	1.2×10^{-6}

[a] NA = value not available.

The variability in the transfer factor is assumed to follow a lognormal probability distribution. Table 6.4.4 compares the current RESRAD (onsite) meat transfer factor distributions parameter values with other references. The RESRAD (onsite) and RESRAD-OFFSITE codes have similar default values.

6.5 MILK TRANSFER FACTORS

6.5.1 Definition

The radionuclide transfer factor for milk is the ratio of the concentration of a radionuclide in fresh milk (pCi/L) to the rate of daily dietary intake of that radionuclide (pCi/d) by the milk animal.

6.5.2 Discussion

There are three routes of contamination intake in animals, as discussed in Section 6.4.2. After absorption, radionuclides enter the circulatory systems and are distributed to various tissues of the body. The site of deposition in many cases depends on the biological role of the corresponding stable element or analogue. In addition, the secretion of contaminants in milk is influenced by other factors, such as breed of dairy animals, age, nutritional status, and stage of lactation.

For many elements and/or radionuclides, the milk transfer factor is derived from other sources, such as stable element concentrations in feed and milk, extrapolations from single-dose tracer experiments, and comparison of elemental concentrations in milk and feed

TABLE 6.4.3 Comparison of Meat Transfer Factors for Different Types of Meat

	IAEA 2010a						PNNL (Staven et al. 2003)		
Element	Beef	Goat	Mutton	Pork	Poultry	Egg	Beef	Poultry	Egg
Ac	NA[a]	NA	NA	NA`	NA	NA	4.0×10^{-4}	6.0×10^{-3}	4.0×10^{-3}
Ag	NA	NA	4.8×10^{-4}	NA`	NA	NA	3.0×10^{-3}	2.0×10^{0}	5.0×10^{-1}
Am	5.0×10^{-4}	NA	1.1×10^{-4}	NA	NA	3.0×10^{-3}	4.0×10^{-5}	6.0×10^{-3}	4.0×10^{-3}
As	NA`	NA	NA	NA`	NA	NA	2.0×10^{-3}	8.3×10^{-1}	2.6×10^{-1}
Au	NA`	NA	NA	NA`	NA	NA	5.0×10^{-3}	1.0×10^{0}	5.0×10^{-1}
Ba	1.4×10^{-4}	1.3×10^{-5}	NA	NA	1.9×10^{-2}	8.7×10^{-1}	2.0×10^{-4}	9.0×10^{-3}	9.0×10^{-1}
Be	NA`	NA	NA	NA`	NA	NA	1.0×10^{-3}	4.0×10^{-1}	2.0×10^{-2}
Bi	NA`	NA	NA	NA`	NA	NA	4.0×10^{-4}	9.8×10^{-2}	2.6×10^{-1}
Br	NA`	NA	NA	NA`	NA	NA	2.5×10^{-2}	4.0×10^{-3}	1.6×10^{0}
Ca	1.3×10^{-2}	NA	NA	2.0×10^{-3}	4.4×10^{-2}	4.4×10^{-1}	2.0×10^{-3}	4.0×10^{-2}	4.0×10^{-1}
Cd	5.8×10^{-3}	NA	1.2×10^{-3}	NA	1.75×10^{0}	NA	4.0×10^{-4}	8.0×10^{-1}	1.0×10^{-1}
Ce	NA	NA	2.5×10^{-4}	NA	NA	3.1×10^{-3}	2.0×10^{-5}	2.0×10^{-3}	4.0×10^{-5}
Cf	NA`	NA	NA	NA`	NA	NA	4.0×10^{-5}	6.0×10^{-3}	4.0×10^{-3}
Cl	1.7×10^{-2}	NA	NA	NA`	NA	NA	2.0×10^{-2}	3.0×10^{-2}	2.7×10^{0}
Cm	NA`	NA	NA	NA`	NA	NA	4.0×10^{-5}	6.0×10^{-3}	4.0×10^{-3}
Co	4.3×10^{-4}	NA	1.2×10^{-2}	NA	9.7×10^{-1}	3.3×10^{-2}	1.0×10^{-2}	2.0×10^{0}	1.0×10^{-1}
Cr	NA`	NA	NA	NA`	NA	NA	9.0×10^{-3}	2.0×10^{-1}	9.0×10^{-1}
Cs	2.2×10^{-2}	3.2×10^{-1}	1.9×10^{-1}	2.0×10^{-1}	2.7×10^{0}	4.0×10^{-1}	5.0×10^{-2}	3.0×10^{0}	4.0×10^{-1}
Cu	NA`	NA	NA	NA`	NA	NA	9.0×10^{-3}	5.0×10^{-1}	5.0×10^{-1}
Dy	NA`	NA	NA	NA`	NA	NA	2.0×10^{-5}	2.0×10^{-3}	4.0×10^{-5}
Er	NA`	NA	NA	NA`	NA	NA	2.0×10^{-5}	2.0×10^{-3}	4.0×10^{-5}
Eu	NA`	NA	NA	NA`	NA	NA	2.0×10^{-5}	2.0×10^{-3}	4.0×10^{-5}
F	NA`	NA	NA	NA`	NA	NA	1.5×10^{-1}	1.4×10^{-2}	2.7×10^{0}
Fe	1.4×10^{-2}	NA	NA	3.0×10^{-3}	NA	1.8×10^{0}	2.0×10^{-2}	1.0×10^{0}	1.0×10^{0}
Ga	NA`	NA	NA	NA`	NA	NA	5.0×10^{-4}	8.0×10^{-1}	1.0×10^{0}
Gd	NA`	NA	NA	NA`	NA	NA	2.0×10^{-5}	2.0×10^{-3}	4.0×10^{-5}
Hf	NA`	NA	NA	NA`	NA	NA	1.0×10^{-3}	6.0×10^{-5}	2.0×10^{-4}
Hg	NA`	NA	NA	NA`	NA	NA	2.5×10^{-1}	3.0×10^{-2}	5.0×10^{-1}
Ho	NA`	NA	NA	NA`	NA	NA	2.0×10^{-5}	2.0×10^{-3}	4.0×10^{-5}
I	6.7×10^{-3}	NA	3.0×10^{-2}	4.1×10^{-2}	8.7×10^{-3}	2.4×10^{0}	4.0×10^{-2}	5.0×10^{-2}	4.4×10^{0}
In	NA`	NA	NA	NA`	NA	NA	8.0×10^{-3}	8.0×10^{-1}	1.0×10^{0}
Ir	NA`	NA	NA	NA`	NA	NA	1.5×10^{-3}	2.0×10^{0}	1.0×10^{-1}
K	NA`	NA	NA	NA`	NA	NA	2.0×10^{-2}	4.0×10^{-1}	1.0×10^{0}
La	1.3×10^{-4}	NA	NA	NA`	NA	NA	2.0×10^{-3}	1.0×10^{-1}	9.0×10^{-3}
Mg	NA`	NA	NA	NA`	NA	NA	2.0×10^{-2}	3.0×10^{-2}	2.0×10^{0}
Mn	6.0×10^{-4}	NA	9.0×10^{-3}	5.3×10^{-3}	1.9×10^{-3}	4.2×10^{-2}	5.0×10^{-4}	5.0×10^{-2}	6.0×10^{-2}
Mo	1.0×10^{-3}	NA	NA	NA	1.8×10^{-1}	6.4×10^{-1}	1.0×10^{-3}	1.8×10^{-1}	9.0×10^{-1}
N	NA	NA	NA	NA	NA	NA	7.5×10^{-2}	9.8×10^{-2}	2.6×10^{-1}
Na	1.5×10^{-2}	NA	1.1×10^{-1}	NA	7.0×10^{0}	4.0×10^{0}	8.0×10^{-2}	1.0×10^{-2}	6.0×10^{0}
Nb	2.60×10^{-7}	6.0×10^{-5}	NA	NA	3.0×10^{-4}	1.0×10^{-3}	3.00×10^{-7}	3.0×10^{-4}	1.0×10^{-3}
Nd	NA`	NA	NA	NA`	NA	NA	2.0×10^{-5}	2.0×10^{-3}	4.0×10^{-5}
Ni	NA`	NA	NA	NA`	NA	NA	5.0×10^{-3}	1.0×10^{-3}	1.0×10^{-1}
Np	NA`	NA	NA	NA`	NA	NA	1.0×10^{-3}	6.0×10^{-3}	4.0×10^{-3}
Os	NA`	NA	NA	NA`	NA	NA	4.0×10^{-1}	8.4×10^{-2}	7.1×10^{-2}
P	5.5×10^{-2}	NA	NA	2.7×10^{-2}	NA	6.4×10^{-1}	5.0×10^{-2}	1.9×10^{-1}	1.0×10^{0}
Pa	NA`	NA	NA	NA`	NA	NA	4.0×10^{-5}	6.0×10^{-3}	4.0×10^{-3}
Pb	7.0×10^{-4}	NA	7.1×10^{-3}	NA	NA	NA	4.0×10^{-4}	8.0×10^{-1}	1.0×10^{0}
Pd	NA`	NA	NA	NA`	NA	NA	4.0×10^{-3}	3.0×10^{-4}	4.0×10^{-3}
Pm	NA`	NA	NA	NA`	NA	NA	2.0×10^{-5}	2.0×10^{-3}	4.0×10^{-5}
Po	NA	NA	NA	NA	2.4×10^{0}	3.1×10^{0}	5.0×10^{-3}	2.3×10^{0}	7.0×10^{0}
Pr	NA	NA	NA`	NA	NA	NA	2.0×10^{-5}	2.0×10^{-3}	4.0×10^{-5}
Pu	1.1×10^{-6}	NA	5.3×10^{-5}	NA	NA	1.2×10^{-3}	1.0×10^{-5}	3.0×10^{-3}	5.0×10^{-4}
Ra	1.7×10^{-3}	NA	NA	NA`	NA	NA	9.0×10^{-4}	3.0×10^{-2}	3.1×10^{-1}
Rb	NA`	NA	NA	NA`	NA	NA	1.0×10^{-2}	2.0×10^{0}	3.0×10^{0}
Re	NA`	NA	NA	NA`	NA	NA	8.0×10^{-3}	4.0×10^{-2}	4.2×10^{-1}

TABLE 6.4.3 (Cont.)

	IAEA 2010a						PNNL (Staven et al. 2003)		
Element	Beef	Goat	Mutton	Pork	Poultry	Egg	Beef	Poultry	Egg
Rh	NA\`	NA	NA	NA\`	NA	NA	2.0×10^{-3}	2.0×10^{0}	1.0×10^{-1}
Ru	3.3×10^{-3}	NA	2.1×10^{-3}	3.0×10^{-3}	NA	4.0×10^{-3}	5.0×10^{-2}	7.0×10^{-3}	5.0×10^{-3}
S	NA	NA	1.7×10^{0}	NA	NA	NA	2.0×10^{-1}	2.3×10^{0}	7.0×10^{0}
Sb	1.2×10^{-3}	NA	NA	NA	NA	NA	1.0×10^{-3}	6.0×10^{-3}	7.0×10^{-2}
Sc	NA\`	NA	NA	NA\`	NA	NA	1.5×10^{-2}	4.0×10^{-3}	4.2×10^{-3}
Se	NA	NA	NA	3.2×10^{-1}	9.7×10^{0}	1.6×10^{1}	1.5×10^{-2}	9.0×10^{0}	9.0×10^{0}
Si	NA\`	NA	NA	NA\`	NA	NA	4.0×10^{-5}	8.0×10^{-1}	1.0×10^{0}
Sm	NA\`	NA	NA	NA\`	NA	NA	2.0×10^{-5}	2.0×10^{-3}	4.0×10^{-5}
Sn	NA\`	NA	NA	NA\`	NA	NA	8.0×10^{-2}	8.0×10^{-1}	1.0×10^{0}
Sr	1.3×10^{-3}	2.9×10^{-3}	1.5×10^{-3}	2.5×10^{-3}	2.0×10^{-2}	3.5×10^{-1}	8.0×10^{-3}	8.0×10^{-2}	2.0×10^{-1}
Ta	NA\`	NA	NA	NA\`	NA	NA	3.00×10^{-7}	3.0×10^{-4}	1.0×10^{-3}
Tb	NA\`	NA	NA	NA\`	NA	NA	2.0×10^{-5}	2.0×10^{-3}	4.0×10^{-5}
Tc	NA\`	NA	NA	NA\`	NA	NA	1.0×10^{-4}	3.0×10^{-2}	3.0×10^{0}
Te	7.0×10^{-3}	2.4×10^{-3}	NA	NA	6.0×10^{-1}	5.1×10^{0}	7.0×10^{-3}	6.0×10^{-1}	5.0×10^{0}
Th	2.3×10^{-4}	NA	NA	NA	NA	NA	4.0×10^{-5}	6.0×10^{-3}	4.0×10^{-3}
Tl	NA\`	NA	NA	NA\`	NA	NA	4.0×10^{-2}	8.0×10^{-1}	1.0×10^{0}
U	3.9×10^{-4}	NA	NA	4.4×10^{-2}	7.5×10^{-1}	1.1×10^{0}	3.0×10^{-4}	1.0×10^{0}	1.0×10^{0}
W	NA\`	NA	NA	NA\`	NA	NA	4.0×10^{-2}	2.0×10^{-1}	9.0×10^{-1}
Y	NA	5.4×10^{-2}	NA	NA	NA	NA	1.0×10^{-3}	1.0×10^{-2}	2.0×10^{-3}
Zn	1.6×10^{-1}	NA	4.5×10^{-2}	1.7×10^{-1}	4.7×10^{-1}	1.4×10^{0}	1.0×10^{-1}	7.0×10^{0}	3.0×10^{0}
Zr	1.2×10^{-6}	2.0×10^{-5}	NA	NA	6.0×10^{-5}	2.0×10^{-4}	1.0×10^{-6}	6.0×10^{-5}	2.0×10^{-4}

[a] NA = value not available.

(Staven et al. 2003). Some of the difficulties in deriving the milk transfer factor include the following:

- *Intake estimation*: The relative proportion of grass, grain, and other dietary constituents is important in estimating the dietary intake. The dietary composition varies according to the feeding strategy (indoors or grazing) and with season. Variability will be less in experimental studies than in field studies (IAEA 2009).

- *The need for equilibrium*: With a few exceptions, the time required for a radionuclide to reach equilibrium in many animal products (e.g., milk) is so long that few experiments can be continued long enough to approach equilibrium conditions (IAEA 2010a). Hence, a transfer factor derived from comparatively short experiments will underestimate the equilibrium transfer factor.

- *Effect of chemical and physical forms of diet and composition*: The availability of a radionuclide for gut uptake differs markedly, depending on the chemical and physical forms of the radionuclide and on the constituents of the diet (IAEA 2009, 2010a). Higher radionuclide concentrations are often found in tissues other than muscle, particularly liver (e.g., Pu, Am, Co, Ag, Ce, Tc) and bone (e.g., Pu, Am, Ce, Sr)

TABLE 6.4.4 Comparison of RESRAD (onsite) Default Meat Transfer Factor Distributions with Those from Other References

	RESRAD (onsite)[a] (Yu et al. 2000)		NCRP (1999)		IAEA (2010)	
Element	µ	σ	µ	σ	µ	σ
Ac	-10.82	1.0	-10.82	1.0	NA[b]	NA
Ag	-6.21	0.7	-5.81	0.7	NA	NA
Al	-7.60	1.0	-7.60	1.0	NA	NA
Am	-9.90	0.4	-9.90	0.4	NA	NA
As	-3.91	1.0	-3.91	1.0	NA	NA
At	NA	NA	-4.61	1.0	NA	NA
Au	-5.30	1.0	-5.30	1.0	NA	NA
B	NA	NA	-7.13	1.0	NA	NA
Ba	-8.52	0.9	-8.52	0.9	NA	NA
Be	-5.30	1.0	-5.30	1.0	NA	NA
Bi	-6.21	1.0	-6.21	1.0	NA	NA
Bk	NA	NA	-10.82	1.0	NA	NA
Br	-3.00	1.0	-3.00	1.0	NA	NA
C	-3.47	1.0	NA	NA	NA	NA
Ca	-6.21	0.2	-6.21	0.2	-4.34	3.4
Cd	-6.91	0.9	-6.91	0.9	-5.15	2.1
Ce	-10.82	0.9	-10.82	0.9	NA	NA
Cf	-9.72	1.0	-9.72	1.0	NA	NA
Cl	-3.22	0.7	-3.22	0.7	NA	NA
Cm	-10.82	1.0	-10.82	1.0	NA	NA
Co	-3.51	1.0	-3.51	1.0	-7.75	0.8
Cr	-3.51	0.4	-3.51	0.4	NA	NA
Cs	-3.00	0.4	-3.00	0.4	-3.82	0.9
Cu	-4.61	0.4	-4.61	0.4	NA	NA
Dy	NA	NA	-6.21	1.0	NA	NA
Er	NA	NA	-6.21	1.0	NA	NA
Es	NA	NA	-10.82	1.0	NA	NA
Eu	-6.21	1.0	-6.21	1.0	NA	NA
F	NA	NA	-3.91	1.0	NA	NA
Fe	-3.51	0.4	-3.51	0.4	-4.27	0.4
Fm	NA	NA	-8.52	1.0	NA	NA
Fr	NA	NA	-3.51	1.0	NA	NA
Ga	NA	NA	-8.11	1.0	NA	NA
Gd	-6.21	1.0	-6.21	1.0	NA	NA
Ge	-1.61	1.0	-1.61	1.0	NA	NA
H	-4.42	1.0	NA	NA	NA	NA
Ha	NA	NA	-12.21	1.0	NA	NA
Hf	NA	NA	-7.82	1.0	NA	NA
Hg	-4.61	1.0	-4.61	1.0	NA	NA
Ho	-6.21	1.0	-6.21	1.0	NA	NA
I	-3.22	0.4	-3.22	0.4	-5.01	1.2
In	-5.52	1.0	-5.52	1.0	NA	NA
Ir	-6.21	1.0	-6.21	1.0	NA	NA

TABLE 6.4.4 (Cont.)

	RESRAD (onsite)[a] (Yu et al. 2000)		NCRP (1999)		IAEA (2010)	
Element	μ	σ	μ	σ	μ	σ
K	-3.91	0.2	-3.91	0.2	NA	NA
La	-6.21	1.0	-6.21	1.0	-8.95	0.2
Li	NA	NA	-3.91	1.0	NA	NA
Lr	NA	NA	-8.52	1.0	NA	NA
Lu	NA	NA	-6.21	1.0	NA	NA
Md	NA	NA	-8.52	1.0	NA	NA
Mg	-5.81	0.2	-5.81	0.2	NA	NA
Mn	-6.91	0.7	-6.91	0.7	NA	NA
Mo	-13.82	0.9	-6.91	0.9	NA	NA
N	NA	NA	NA	NA	NA	NA
Na	-2.53	0.2	-2.53	0.2	NA	NA
Nb	-13.82	0.9	-13.82	0.9	NA	NA
Nd	-6.21	1.0	-6.21	1.0	NA	NA
Ni	-5.30	0.9	-5.30	0.9	NA	NA
No	NA	NA	-8.52	1.0	NA	NA
Np	-6.91	0.7	-6.91	0.7	NA	NA
O	NA	NA	NA	NA	NA	NA
Os	NA	NA	-6.21	1.0	NA	NA
P	-3.00	0.2	-3.00	0.2	NA	NA
Pa	-12.21	1.0	-12.21	1.0	NA	NA
Pb	-7.13	0.7	-7.13	0.7	-7.26	0.9
Pd	-8.52	1.0	-8.52	1.0	NA	NA
Pm	-6.21	1.0	-6.21	1.0	NA	NA
Po	-5.30	0.7	-5.30	0.7	NA	NA
Pr	-6.21	1.0	-6.21	1.0	NA	NA
Pt	NA	NA	-8.52	1.0	NA	NA
Pu	-9.21	0.2	-9.21	0.4	-13.72	3.2
Ra	-6.91	0.7	-6.91	0.7	NA	NA
Rb	-3.51	0.7	-3.51	0.7	NA	NA
Re	NA	NA	-4.61	1.0	NA	NA
Rf	NA	NA	-7.82	1.0	NA	NA
Rh	-6.21	1.0	-6.21	1.0	NA	NA
Ru	-6.21	0.9	-6.21	0.9	-5.71	0.6
S	-1.61	1.0	-1.61	1.0	NA	NA
Sb	-6.91	0.9	-6.91	0.9	NA	NA
Sc	-6.21	1.0	-6.21	1.0	NA	NA
Se	-2.30	0.9	-2.30	0.9	NA	NA
Si	-8.11	1.0	-8.11	1.0	NA	NA
Sm	-6.21	1.0	-6.21	1.0	NA	NA
Sn	-4.61	1.0	-4.61	1.0	NA	NA
Sr	-4.61	0.4	-4.61	0.4	-6.65	1.1
Ta	-12.21	1.0	-12.21	1.0	NA	NA
Tb	-6.21	1.0	-6.21	1.0	NA	NA
Tc	-9.21	0.7	-9.21	0.7	NA	NA
Te	-4.96	0.9	-4.96	0.9	NA	NA

TABLE 6.4.4 (Cont.)

	RESRAD (onsite)[a] (Yu et al. 2000)		NCRP (1999)		IAEA (2010)	
Element	μ	σ	μ	σ	μ	σ
Th	-9.21	1.0	-9.21	1.0	-8.38	1.1
Ti	NA	NA	-3.91	1.0	NA	NA
Tl	-3.91	1.0	-3.91	1.0	NA	NA
Tm	NA	NA	-6.21	1.0	NA	NA
U	-7.13	0.7	-7.13	0.7	-7.85	0.5
V	NA	NA	-4.61	1.0	NA	NA
W	-3.22	0.9	-3.22	0.9	NA	NA
Y	-6.21	0.9	-6.21	0.9	NA	NA
Yb	NA	NA	-6.21	1.0	NA	NA
Zn	-2.30	0.3	-2.30	0.3	-1.83	1.2
Zr	-13.82	0.9	-13.82	0.9	NA	NA

[a] For H and C, transfer factor distribution is derived from Hoffman et al. (1982).

[b] NA = value not available.

Note: μ = mean of the underlying normal distribution and σ = standard deviation of the underlying normal distribution.

(IAEA 2009). Some elements are actively taken up by mammary glands (e.g., I, Cs, and Sr) and transferred to milk. Radionuclide transfer models often underestimate soil adhesion on vegetation ingested by animals. The extent of soil ingestion will be influenced by the species of animal, season, soil type, stocking rates, and pasture management. Consequently, values for soil ingestion will be highly site specific.

- *Influence of age*: The intake of radionuclides by an animal is dependent on the animal's species, mass, age, and growth rate, as well as on the digestibility of the feed. Young animals often have enhanced gut uptake and, hence, higher transfer coefficients than adults. Few available transfer coefficient data take these factors into account.

6.5.3 Data Input Requirements

This parameter is used when the milk ingestion pathway is active. The default transfer factors in the RESRAD (onsite) and RESRAD-OFFSITE codes are for cow's milk, since cow's milk is generally consumed in larger quantities in the United States than other milk types (NCRP 1999). If the human food source in the dose assessment is other than cow's milk (e.g., goat's milk), the transfer factors should be changed accordingly.

The default values of milk transfer factors compiled from multiple sources were reviewed in 1993 (Wang el al. 1993) and were used as default values in the RESRAD (onsite) code. Since the publication of that report, a number of new publications on transfer factors have become available. The probabilistic dose analysis capability was developed for the RESRAD (onsite) code in 2000 (Yu et al. 2000), and the milk transfer factor distribution was developed using data from NCRP (1999). The other published radiological assessments used for comparison of milk transfer factors in this report are those of Staven et al. (2003) and IAEA (2009, 2010a).

To select values for milk transfer factors, the most recent and comprehensive references should be given first priority. The first reference chosen for this report was the IAEA Technical Report Series No. 472, *Handbook of Parameter Values for the Prediction of Radionuclide Transfer in Terrestrial and Freshwater Environments* (IAEA 2010a) and its supporting document (IAEA 2009), which includes the full compilation of reviewed data and the methods used to obtain the data values. The data included in IAEA reports relate mainly to equilibrium conditions. The second reference chosen was the PNNL report *A Compendium of Transfer Factors for Agricultural and Animal Products* (Staven et al. 2003). The transfer factors in the PNNL report were used in the GENII system of computer codes. The transfer parameters in the PNNL report were selected from recommended values listed by national or international organizations, and if no reference document was available, the values were derived on the basis of chemical groupings in the periodic table of the elements. NCRP Report No. 129 (NCRP 1999), *Recommended Screening Limits for Contaminated Surface Soil and Review of Factors Relevant to Site-Specific Studies,* was chosen because of its general acceptance in dose assessment.

Table 6.5.1 compares the current RESRAD (onsite) and RESRAD-OFFSITE default values for milk (cow) transfer factors with those from other references. In general, elements with higher fractional absorption (Table 6.4.1) tend to have higher milk transfer factors. The IAEA handbook of parameter values (IAEA 2010a) also provided milk transfer factors for goat’s milk and sheep’s milk. Table 6.5.2 compares, for different types of milk, the mean or geometric mean (if the number of data points was greater than 2) and geometric standard deviation values (probability distributions were developed if the number of data points was greater than 3).

The variability in the transfer factor is assumed to follow a lognormal probability distribution. Table 6.5.3 compares the current RESRAD (onsite) and RESRAD-OFFSITE milk transfer factor distributions with those from other references.

TABLE 6.5.1 Comparison of RESRAD (onsite) Default Milk (Cow) Transfer Factors with Those from Other References

Element	RESRAD Default	PNNL (Staven et al. 2003)	NCRP (1999)	IAEA (2010a)
Ac	2.0×10^{-5}	2.0×10^{-5}	2.0×10^{-6}	NA[a]
Ag	2.5×10^{-2}	5.0×10^{-5}	6.0×10^{-3}	NA
Al	2.0×10^{-4}	NA	2.0×10^{-4}	NA
Am	2.0×10^{-6}	1.5×10^{-6}	2.0×10^{-6}	4.2×10^{-7}
As	1.0×10^{-4}	6.0×10^{-5}	1.0×10^{-4}	NA
At	1.0×10^{-2}	NA	1.0×10^{-2}	NA
Au	1.0×10^{-5}	5.5×10^{-6}	1.0×10^{-5}	NA
B	NA	NA	3.0×10^{-3}	NA
Ba	5.0×10^{-4}	4.8×10^{-4}	5.0×10^{-4}	1.6×10^{-4}
Be	2.0×10^{-6}	9.0×10^{-7}	2.0×10^{-6}	8.3×10^{-7}
Bi	5.0×10^{-4}	5.0×10^{-4}	1.0×10^{-3}	NA
Bk	2.0×10^{-6}	NA	2.0×10^{-6}	NA
Br	2.0×10^{-2}	2.0×10^{-2}	2.0×10^{-2}	NA
C	1.2×10^{-2}	NA	NA	NA
Ca	3.0×10^{-3}	3.0×10^{-3}	3.0×10^{-3}	1.0×10^{-2}
Cd	1.0×10^{-3}	1.0×10^{-3}	2.0×10^{-3}	1.9×10^{-4}
Ce	3.0×10^{-5}	3.0×10^{-5}	3.0×10^{-5}	2.0×10^{-5}
Cf	7.5×10^{-7}	1.5×10^{-6}	2.0×10^{-6}	NA
Cl	2.0×10^{-2}	1.7×10^{-2}	2.0×10^{-2}	NA
Cm	2.0×10^{-6}	2.0×10^{-5}	2.0×10^{-6}	NA
Co	2.0×10^{-3}	3.0×10^{-4}	2.0×10^{-3}	1.1×10^{-4}
Cr	2.0×10^{-3}	1.0×10^{-5}	2.0×10^{-3}	4.3×10^{-4}
Cs	8.0×10^{-3}	7.9×10^{-3}	1.0×10^{-2}	4.6×10^{-3}
Cu	2.0×10^{-3}	2.0×10^{-3}	2.0×10^{-3}	NA
Dy	6.0×10^{-5}	3.0×10^{-5}	6.0×10^{-5}	NA
Er	6.0×10^{-5}	3.0×10^{-5}	6.0×10^{-5}	NA
Es	2.0×10^{-6}	NA	4.0×10^{-7}	NA
Eu	2.0×10^{-5}	3.0×10^{-5}	6.0×10^{-5}	NA
F	7.0×10^{-3}	1.0×10^{-3}	7.0×10^{-3}	NA
Fe	3.0×10^{-4}	3.0×10^{-5}	3.0×10^{-4}	3.5×10^{-5}
Fm	8.0×10^{-6}	NA	8.0×10^{-6}	NA
Fr	8.0×10^{-3}	NA	8.0×10^{-3}	NA
Ga	1.0×10^{-5}	5.0×10^{-5}	1.0×10^{-5}	NA
Gd	2.0×10^{-5}	3.0×10^{-5}	6.0×10^{-5}	NA
Ge	1.0×10^{-2}	NA	1.0×10^{-2}	NA
H	1.0×10^{-2}	1.5×10^{-2}	NA	NA
Ha	NA	NA	5.0×10^{-6}	NA
Hf	2.0×10^{-5}	5.5×10^{-7}	2.0×10^{-5}	NA
Hg	5.0×10^{-4}	4.7×10^{-4}	5.0×10^{-4}	NA
Ho	2.0×10^{-5}	3.0×10^{-5}	6.0×10^{-5}	NA
I	1.0×10^{-2}	9.0×10^{-3}	1.0×10^{-2}	5.4×10^{-3}
In	2.0×10^{-4}	2.0×10^{-4}	2.0×10^{-4}	NA
Ir	2.0×10^{-6}	2.0×10^{-6}	2.0×10^{-6}	NA
K	7.0×10^{-3}	7.2×10^{-3}	7.0×10^{-3}	NA
La	2.0×10^{-5}	2.0×10^{-5}	6.0×10^{-5}	NA

TABLE 6.5.1 (Cont.)

Element	RESRAD Default	PNNL (Staven et al. 2003)	NCRP (1999)	IAEA (2010a)
Li	NA	NA	5.0×10^{-2}	NA
Lr	NA	NA	5.0×10^{-6}	NA
Lu	6.0×10^{-5}	NA	6.0×10^{-5}	NA
Md	5.0×10^{-6}	NA	5.0×10^{-6}	NA
Mg	8.0×10^{-3}	3.9×10^{-3}	8.0×10^{-3}	NA
Mn	3.0×10^{-4}	3.0×10^{-5}	3.0×10^{-4}	4.1×10^{-5}
Mo	1.7×10^{-3}	1.7×10^{-3}	2.0×10^{-3}	1.1×10^{-3}
N	1.0×10^{-2}	2.5×10^{-2}	NA	NA
Na	4.0×10^{-2}	1.6×10^{-2}	4.0×10^{-2}	1.3×10^{-2}
Nb	2.0×10^{-6}	4.10×10^{-7}	2.0×10^{-6}	4.10×10^{-7}
Nd	2.0×10^{-5}	3.0×10^{-5}	6.0×10^{-5}	NA
Ni	2.0×10^{-2}	1.6×10^{-2}	2.0×10^{-2}	9.5×10^{-4}
No	NA	NA	5.0×10^{-6}	NA
Np	5.0×10^{-6}	5.0×10^{-6}	1.0×10^{-5}	NA
O	2.0×10^{-2}	NA	NA	NA
Os	1.0×10^{-4}	5.0×10^{-3}	1.0×10^{-4}	NA
P	1.6×10^{-2}	1.6×10^{-2}	2.0×10^{-2}	2.0×10^{-2}
Pa	5.0×10^{-6}	5.0×10^{-6}	5.0×10^{-6}	NA
Pb	3.0×10^{-4}	2.6×10^{-4}	3.0×10^{-4}	1.9×10^{-4}
Pd	5.0×10^{-3}	1.0×10^{-2}	1.0×10^{-4}	NA
Pm	2.0×10^{-5}	3.0×10^{-5}	6.0×10^{-5}	NA
Po	3.4×10^{-4}	3.4×10^{-4}	4.0×10^{-4}	2.1×10^{-4}
Pr	2.0×10^{-5}	3.0×10^{-5}	6.0×10^{-5}	NA
Pt	1.0×10^{-4}	NA	1.0×10^{-4}	NA
Pu	1.0×10^{-6}	1.1×10^{-6}	1.0×10^{-6}	1.0×10^{-5}
Ra	1.0×10^{-3}	1.3×10^{-3}	1.0×10^{-3}	3.8×10^{-4}
Rb	1.0×10^{-2}	1.2×10^{-2}	1.0×10^{-2}	NA
Re	2.0×10^{-3}	1.5×10^{-3}	2.0×10^{-3}	NA
Rf	NA	NA	2.0×10^{-5}	NA
Rh	5.0×10^{-3}	1.0×10^{-2}	5.0×10^{-4}	NA
Ru	3.3×10^{-6}	3.3×10^{-6}	2.0×10^{-5}	9.4×10^{-6}
S	2.0×10^{-2}	1.6×10^{-2}	2.0×10^{-2}	7.9×10^{-3}
Sb	1.0×10^{-4}	2.5×10^{-5}	1.0×10^{-4}	3.8×10^{-5}
Sc	5.0×10^{-6}	5.0×10^{-6}	6.0×10^{-5}	NA
Se	1.0×10^{-2}	4.0×10^{-3}	1.0×10^{-2}	4.0×10^{-3}
Si	2.0×10^{-5}	2.0×10^{-5}	2.0×10^{-5}	NA
Sm	2.0×10^{-5}	3.0×10^{-5}	6.0×10^{-5}	NA
Sn	1.0×10^{-3}	1.0×10^{-3}	1.0×10^{-3}	NA
Sr	2.0×10^{-3}	2.8×10^{-3}	2.0×10^{-3}	1.3×10^{-3}
Ta	5.0×10^{-6}	4.10×10^{-7}	5.0×10^{-6}	NA
Tb	2.0×10^{-5}	3.0×10^{-5}	6.0×10^{-5}	NA
Tc	1.0×10^{-3}	1.4×10^{-4}	1.0×10^{-3}	NA
Te	5.0×10^{-4}	4.5×10^{-4}	5.0×10^{-4}	3.4×10^{-4}
Th	5.0×10^{-6}	5.0×10^{-6}	5.0×10^{-6}	NA
Ti	1.0×10^{-2}	NA	1.0×10^{-2}	NA
Tl	3.0×10^{-3}	2.0×10^{-3}	3.0×10^{-3}	NA
Tm	6.0×10^{-5}	NA	6.0×10^{-5}	NA

TABLE 6.5.1 (Cont.)

Element	RESRAD Default	PNNL (Staven et al. 2003)	NCRP (1999)	IAEA (2010a)
U	6.0×10^{-4}	4.0×10^{-4}	4.0×10^{-4}	1.8×10^{-3}
V	5.0×10^{-4}	NA	5.0×10^{-4}	NA
W	3.0×10^{-4}	3.0×10^{-4}	3.0×10^{-4}	1.9×10^{-4}
Y	2.0×10^{-5}	2.0×10^{-5}	6.0×10^{-5}	NA
Yb	6.0×10^{-5}	NA	6.0×10^{-5}	NA
Zn	1.0×10^{-2}	1.0×10^{-2}	1.0×10^{-2}	2.7×10^{-3}
Zr	6.0×10^{-7}	5.5×10^{-7}	6.0×10^{-7}	3.6×10^{-6}

[a] NA = value not available.

6.6 BIOACCUMULATION FACTORS FOR AQUATIC ORGANISMS

6.6.1 Definition

The bioaccumulation factor for an aquatic organism or tissue is the ratio of the concentration of a radionuclide in the whole organism or tissue (pCi/kg) to the concentration of that same radionuclide in water (pCi/L).

6.6.2 Discussion

A bioaccumulation factor is used to calculate the transfer of a radionuclide from contaminated water through various trophic levels of aquatic foodstuffs consumed by humans. The factor is normally expressed as the ratio of radioactivity in animal tissue to that in water at equilibrium conditions (pCi/kg wet or dry weight organism per pCi/L water).

The physicochemical form of the radionuclide is generally more important in aquatic ecosystems than in terrestrial ecosystems, where most of the affected food products are produced in situations in which most of the factors can be controlled. In aquatic and marine environments, numerous species in the food chain are mobile and can move over considerable distances. Therefore, the concentration of a radionuclide can change much faster with time in aquatic ecosystems than in terrestrial ecosystems, and the equilibrium condition is less likely to be achieved in the former. A radionuclide may exist in water in a truly dissolved state, as a colloid, or adsorbed to particulate matter. Reactions between radionuclides and chemical species present in the water determine the biological availability of the radionuclide for uptake in aquatic environments. For example, a dissolved radionuclide might precipitate out of solution and become less available for uptake if the concentrations of ligands in the water system are sufficiently high that the corresponding solubility product is exceeded. A radionuclide that is adsorbed to particulate matter might dissolve and become available for uptake if the

TABLE 6.5.2 Comparison of Milk Transfer Factors for Different Types of Milk

	Cow Milk		Goat Milk		Sheep Milk	
Element	Mean/GM	GSD	Mean/GM	GSD	Mean/GM	GSD
Am	4.2×10^{-7}	NA[a]	6.9×10^{-6}	NA	NA	NA
Ba	1.6×10^{-4}	2.7	0.011	9.9	4.1×10^{-2}	NA
Be	8.3×10^{-7}	NA	NA	NA	NA	NA
Ca	1.0×10^{-2}	1.7	7.3×10^{-2}	1.9	2.3×10^{-1}	NA
Cd	1.9×10^{-4}	15	1.6×10^{-2}	NA	4.9×10^{-2}	NA
Ce	2.0×10^{-5}	5.8	4.0×10^{-5}	NA	NA	NA
Co	1.1×10^{-4}	2	5.0×10^{-3}	NA	2.7×10^{-3}	NA
Cr	4.3×10^{-4}	26	1.5×10^{-2}	NA	2.0×10^{-2}	NA
Cs	4.6×10^{-3}	2	1.1×10^{-1}	2.2	5.8×10^{-2}	2.3
Fe	3.5×10^{-5}	2	5.2×10^{-2}	NA	7.9×10^{-2}	NA
I	5.4×10^{-3}	2.4	2.2×10^{-1}	2.9	2.3×10^{-1}	3.3
Mn	4.1×10^{-5}	4.9	1.0×10^{-3}	NA	2.4×10^{-3}	NA
Mo	1.1×10^{-3}	2.3	8.2×10^{-3}	1.4	NA	NA
Na	1.3×10^{-2}	2	1.2×10^{-1}	NA	1.0×10^{-1}	NA
Nb	4.1×10^{-7}	NA	6.4×10^{-6}	NA	NA	NA
Ni	9.5×10^{-4}	NA	8.3×10^{-2}	NA	2.8×10^{-1}	NA
Np	NA	NA	5.3×10^{-5}	NA	NA	NA
P	2.0×10^{-2}	NA	2.9×10^{-1}	NA	3.1×10^{-1}	NA
Pb	1.9×10^{-4}	1	6.0×10^{-3}	NA	3.5×10^{-2}	NA
Pm	NA	NA	2.7×10^{-5}	NA	NA	NA
Po	2.1×10^{-4}	1.8	2.3×10^{-3}	NA	NA	NA
Pu	1.0×10^{-5}	NA	NA	NA	1.0×10^{-4}	NA
Ra	3.8×10^{-4}	2.3	NA	NA	NA	NA
Ru	9.4×10^{-6}	8.5	NA	NA	NA	NA
S	7.9×10^{-3}	NA	3.8×10^{-2}	1.7	1.5×10^{-1}	NA
Sb	3.8×10^{-5}	2.5	NA	NA	NA	NA
Se	4.0×10^{-3}	2.1	6.9×10^{-2}	NA	NA	NA
Sr	1.3×10^{-3}	1.7	1.6×10^{-2}	2.0	2.7×10^{-2}	1.2
Te	3.4×10^{-4}	2.4	4.4×10^{-3}	NA	2.9×10^{-3}	NA
U	1.8×10^{-3}	3.5	1.4×10^{-3}	NA	NA	NA
W	1.9×10^{-4}	3.1	NA	NA	NA	NA
Y	NA	NA	2.0×10^{-5}	NA	NA	NA
Zn	2.7×10^{-3}	3.9	6.4×10^{-2}	NA	8.1×10^{-2}	NA
Zr	3.6×10^{-6}	4.3	5.5×10^{-6}	NA	NA	NA

[a] NA = value not available.

Note: GM = geometric mean; GSD = geometric standard deviation.

TABLE 6.5.3 Comparison of RESRAD (onsite) Default Milk Transfer Factor Distributions with Those from Other References

	RESRAD (onsite) Default[a] (Yu et al. 2000)		NCRP (1999)		IAEA (2010a)	
Element	μ	σ	μ	σ	μ	σ
Ac	-13.12	0.9	-13.12	0.9	NA[b]	NA
Ag	-5.12	0.7	-5.12	0.7	NA	NA
Al	-8.52	0.9	-8.52	0.9	NA	NA
Am	-13.12	0.7	-13.12	0.7	-14.68	NA
As	-9.21	0.9	-9.21	0.9	NA	NA
At	NA	NA	-4.61	0.9	NA	NA
Au	-11.51	0.9	-11.51	0.9	NA	NA
B	NA	NA	-5.81	0.9	NA	NA
Ba	-7.60	0.7	-7.60	0.7	-8.74	1.0
Be	-13.12	0.9	-13.12	0.9	-14.00	NA
Bi	-6.91	0.9	-6.91	0.9	NA	NA
Bk	NA	NA	-13.12	0.9	NA	NA
Br	-3.91	0.9	-3.91	0.9	NA	NA
C	-4.40	0.9	NA	NA	NA	NA
Ca	-5.81	0.5	-5.81	0.5	-4.61	0.5
Cd	-6.21	0.9	-6.21	0.9	-8.57	2.7
Ce	-10.41	0.7	-10.41	0.7	-10.82	1.8
Cf	-13.12	0.9	-13.12	0.9	NA	NA
Cl	-3.91	0.5	-3.91	0.5	NA	NA
Cm	-13.12	0.9	-13.12	0.9	NA	NA
Co	-6.21	0.7	-6.21	0.7	-9.12	0.7
Cr	-6.21	0.7	-6.21	0.7	-7.75	3.3
Cs	-4.61	0.5	-4.61	0.5	-5.38	0.7
Cu	-6.21	0.9	-6.21	0.9	NA	NA
Dy	NA	NA	-9.72	0.9	NA	NA
Er	NA	NA	-9.72	0.9	NA	NA
Es	NA	NA	-14.73	0.9	NA	NA
Eu	-9.72	0.9	-9.72	0.9	NA	NA
F	NA	NA	-4.96	0.9	NA	NA
Fe	-8.11	0.7	-8.11	0.7	-10.26	0.7
Fm	NA	NA	-11.74	0.9	NA	NA
Gr	NA	NA	-4.83	0.9	NA	NA
Ga	NA	NA	-11.51	0.9	NA	NA
Gd	-9.72	0.9	-9.72	0.9	NA	NA
Ge	-4.61	0.9	-4.61	0.9	NA	NA
H	-4.60	0.9	NA	NA	NA	NA
Ha	NA	NA	-12.21	0.9	NA	NA
Hf	NA	NA	-10.82	0.9	NA	NA
Hg	-7.60	0.7	-7.60	0.7	NA	NA
Ho	-9.72	0.9	-9.72	0.9	NA	NA
I	-4.61	0.5	-4.61	0.5	-5.22	0.9
In	-8.52	0.9	-8.52	0.9	NA	NA
Ir	-13.12	0.9	-13.12	0.9	NA	NA

TABLE 6.5.3 (Cont.)

	RESRAD (onsite) Default[a] (Yu et al. 2000)		NCRP (1999)		IAEA (2010a)	
Element	μ	σ	μ	σ	μ	σ
K	-4.96	0.5	-4.96	0.5	NA	NA
La	-9.72	0.9	-9.72	0.9	NA	NA
Li	NA	NA	-3.00	0.9	NA	NA
Lr	NA	NA	-12.21	0.9	NA	NA
Lu	NA	NA	-9.72	0.9	NA	NA
Md	NA	NA	-12.21	0.9	NA	NA
Mg	NA	NA	-4.83	0.7	NA	NA
Mn	-8.11	0.7	-8.11	0.7	-10.10	1.6
Mo	-6.21	0.7	-6.21	0.7	-6.81	0.8
N	NA	NA	NA	NA	NA	NA
Na	-3.22	0.5	-3.22	0.5	-4.34	0.7
Nb	-13.12	0.7	-13.12	0.7	-14.71	NA
Nd	-9.72	0.9	-9.72	0.9	NA	NA
Ni	-3.91	0.7	-3.91	0.7	-6.96	NA
No	NA	NA	-12.21	0.9	NA	NA
Np	-11.51	0.7	-11.51	0.7	NA	NA
O	NA	NA	NA	NA	NA	NA
Os	NA	NA	-9.21	0.9	NA	NA
P	-3.91	0.7	-3.91	0.7	-3.91	NA
Pa	-12.21	0.9	-12.21	0.9	NA	NA
Pb	-8.11	0.9	-8.11	0.9	-8.57	0.0
Pd	-9.21	0.9	-9.21	0.9	NA	NA
Pm	-9.72	0.9	-9.72	0.9	NA	NA
Po	-7.82	0.7	-7.82	0.7	-8.47	0.6
Pr	-9.72	0.9	-9.72	0.9	NA	NA
Pt	NA	NA	-9.21	0.9	NA	NA
Pu	-13.82	0.5	-13.82	0.5	-11.51	NA
Ra	-6.91	0.5	-6.91	0.5	-7.88	0.8
Rb	-4.61	0.7	-4.61	0.7	NA	NA
Re	NA	NA	-6.21	0.9	NA	NA
Rf	NA	NA	-10.82	0.9	NA	NA
Rh	-7.60	0.9	-7.60	0.9	NA	NA
Ru	-10.82	0.6	-10.82	0.6	-11.57	2.1
S	-3.91	0.7	-3.91	0.7	-4.84	NA
Sb	-9.72	0.9	-9.21	0.7	-10.18	0.9
Sc	-5.12	0.9	-9.72	0.9	NA	NA
Se	-4.61	0.9	-4.61	0.9	-5.52	0.7
Si	NA	NA	-10.82	0.9	NA	NA
Sm	-9.72	0.9	-9.72	0.9	NA	NA
Sn	-6.91	0.9	-6.91	0.9	NA	NA
Sr	-6.21	0.5	-6.21	0.5	-6.65	0.5
Ta	-12.21	0.9	-12.21	0.9	NA	NA
Tb	-9.72	0.9	-9.72	0.9	NA	NA
Tc	-6.91	0.7	-6.91	0.7	NA	NA

TABLE 6.5.3 (Cont.)

	RESRAD (onsite) Default[a] (Yu et al. 2000)		NCRP (1999)		IAEA (2010a)	
Element	μ	σ	μ	σ	μ	σ
Te	-7.60	0.6	-7.60	0.6	-7.99	0.9
Th	-12.21	0.9	-12.21	0.9	NA	NA
Ti	NA	NA	-4.61	0.9	NA	NA
Tl	-5.81	0.9	-5.81	0.9	NA	NA
Tm	NA	NA	-9.72	0.9	NA	NA
U	-7.82	0.6	-7.82	0.6	-6.32	1.3
V	NA	NA	-7.60	0.9	NA	NA
W	-8.11	0.9	-8.11	0.9	-8.57	1.1
Y	-9.72	0.9	-9.72	0.9	NA	NA
Yb	NA	NA	-9.72	0.9	NA	NA
Zn	-4.61	0.9	-4.61	0.9	-5.91	1.4
Zr	-14.33	0.7	-14.33	0.7	-12.53	1.5

[a] For H and C, transfer factor distributions are derived from Hoffman et al. (1982).

[b] NA = value not available.

Note: μ = mean of the underlying normal distribution and σ = standard deviation of the underlying normal distribution.

concentrations of ligands and stable isotopes of the radionuclide are such that the solubility product is not exceeded.

In the literature, bioaccumulation factors are derived by a number of methods, and the reported values vary widely. Historically, radioactivity in animal tissue is estimated on the basis of ash weight, dry weight, wet weight, whole body burdens, and/or muscle tissue. Radioactivity in water is estimated on the basis of filtered or unfiltered water. The wet weight to dry weight to ash weight ratio can vary as a function of the age, size, and species of fish. These different measurement methods can affect the computed bioaccumulation factor of the fish. For radionuclides that partition into soluble (liquid) and particulate phases, the degree of partitioning must be considered. A higher bioaccumulation factor will be obtained if the radionuclide concentration in the soluble (filtrate) fraction is measured than if it is measured in an unfiltered sample. For instance, if 1% of the radionuclide is present as soluble species, and the rest is in the solid phase, the transfer factor for a filtered water sample would be estimated to be 100 times greater than the factor for an unfiltered water sample (Poston and Klopfer 1986).

6.6.3 Data Input Requirements

The bioaccumulation factor is used when the aquatic food pathway is active. The default transfer factors in the RESRAD (onsite) and RESRAD-OFFSITE codes include two aquatic food types—fish and crustacea—and are for freshwater (FW) applications only.

To select values for aquatic food bioaccumulation factors, the most recent and comprehensive references should be given first priority. The first reference chosen in this report was the IAEA's Technical Report Series No. 472, *Handbook of Parameter Values for the Prediction of Radionuclide Transfer in Terrestrial and Freshwater Environments* (IAEA 2010a) and its supporting document (IAEA 2009), which includes the full compilation of reviewed data and the methods used to obtain the data values. The data included in IAEA reports relate mainly to equilibrium conditions. The second reference used was the PNNL report *A Compendium of Transfer Factors for Agricultural and Animal Products* (Staven et al. 2003). The transfer factors in the PNNL report were used in the GENII system of computer codes. The transfer parameters in the PNNL report were selected from recommended values listed by national or international organizations, if no reference document was available; the values were derived on the basis of chemical groupings in the periodic table of the elements. The third reference used was the NCRP report, *Screening Models for Releases of Radionuclides to Atmosphere, Surface water, and Ground* (NCRP 1996).

6.6.4 Fish Bioaccumulation Factors

The physiological status of fish plays an important role in their uptake of radionuclides. Young, rapidly growing fish may accumulate higher levels of biologically active radionuclides than fish in a stationary growth period. The different osmoregulatory problems faced by FW fish and marine fish also create differences in the route of radionuclide uptake. In seawater, the salt concentration is high, and marine fish drink large amounts of water and expend considerable energy to excrete salt against a concentration gradient. In fresh water, fish retain salt and excrete a large amount of water. Therefore, radionuclides found in the water column, either as dissolved species or adsorbed to particulate matter, are more prone to gastrointestinal absorption in marine species than in FW species.

The default values of fish bioaccumulation factors compiled from multiple sources were reviewed in 1993 (Wang et al. 1993). On the basis of that data review, the default values in the RESRAD (onsite) code were revised. Since the publication of the 1993 report, a number of new publications on transfer factors have become available. In 2000, a probabilistic dose analysis capability was developed for the RESRAD (onsite) code (Yu et al. 2000). The distribution for fish bioaccumulation factors was developed for the code using data from Wang et al. (1993).

Table 6.6.1 compares the current RESRAD (onsite) and RESRAD-OFFSITE default values for FW fish bioaccumulation factors with factors published by NCRP (1996), PNNL (Staven et al. 2003), and IAEA (2009, 2010a). The PNNL (Staven et al. 2003) and NCRP (1996) reports also contained saltwater (SW) fish bioaccumulation factors. The IAEA handbook of

TABLE 6.6.1 Comparison of RESRAD (onsite) Freshwater Fish Bioaccumulation Factors with Those from Other References

	RESRAD	PNNL (Staven et al. 2003)		NCRP 1996			IAEA 2010a	
Element	Freshwater Fish	Freshwater Fish	Saltwater Fish	Freshwater Fish	Saltwater Fish	Saltwater Shellfish	Freshwater Whole Fish	Freshwater Fish Muscle
Ac	15	25	30	15	10	50	NA	NA
Ag	5	5	300	10	1000	5000	110	110
Al	500	NA[a]	NA	500	500	1000	66	51
Am	30	30	2500	30	10	2000	NA	240
As	300	1700	300	400	1000	1000	390	330
At	15	NA	NA	15	10	50	NA	NA
Au	35	33	NA	35	100	1000	290	240
B	NA	NA	NA	5	200	200	NA	NA
Ba	4	4	10	4	10	100	47	1.2
Be	100	100	1000	100	200	200	NA	NA
Bi	15	15	15	15	20	1000	NA	NA
Bk	25	NA	NA	25	25	1000	NA	NA
Br	420	400	10	400	3	10	160	91
C	50000	50000	20000	50000	2000	2000	NA	400000
Ca	1000	40	2	1000	1	1	1000	12
Cd	200	200	2000	200	3000	250000	NA	NA
Ce	30	30	100	30	10	10	12	25
Cf	25	25	50	25	25	1000	NA	NA
Cl	1000	50	1	1000	1	1	95	47
Cm	30	30	2500	30	10	200	NA	NA
Co	300	300	100	300	100	1000	400	76
Cr	200	4	600	200	400	500	210	40
Cs	2000	2000	100	2000	50	30	3000	2500
Cu	200	200	1000	200	700	1700	270	230
Dy	25	30	500	25	100	1000	300	650
Er	12000	30	100	12000	30	1000	NA	NA
Es	25	NA	NA	25	25	1000	NA	NA
Eu	50	30	300	50	100	1000	150	130
F	10	10	4	10	4	4	NA	NA
Fe	200	200	3000	200	3000	10000	140	170
Fm	10	NA	NA	10	10	10	NA	NA
Fr	30	NA	NA	30	10000	1000	NA	NA
Ga	400	400	1000	400	700	700	NA	NA
Gd	25	30	500	30	25	1000	NA	NA
Ge	4000	NA	NA	4000	4000	20000	NA	NA
H	1	NA	NA	1	1	1	NA	NA
Ha	NA	NA	NA	40	30	30	NA	NA
Hf	40	300	40	40	30	30	2100	1100
Hg	1000	1000	20000	1000	2000	1000	4500	6100
Ho	25	30	300	12000	30	30	NA	NA
I	40	40	10	40	10	100	650	30

TABLE 6.6.1 (Cont.)

	RESRAD	PNNL (Staven et al. 2003)		NCRP 1996			IAEA 2010a	
Element	Fresh-water Fish	Fresh-water Fish	Salt-water Fish	Fresh-water Fish	Salt-water Fish	Saltwater Shellfish	Fresh-water Whole Fish	Freshwater Fish Muscle
In	10000	10000	1000	10000	10000	15000	NA	NA
Ir	10	10	20	10	10	2000	NA	NA
K	1000	1000		10000	5000	5000	4000	3200
La	30	30	30	30	100	1000	16	37
Li	NA	NA	NA	1	1	1	NA	NA
Lr	NA	NA	NA	10	10	10	NA	NA
Lu	25	NA	NA	25	25	1000	NA	NA
Md	10	NA	NA	10	10	10	NA	NA
Mg	50	50		50	1	1	110	37
Mn	400	400	400	500	500	1000	450	240
Mo	10	10	40	10	10	100	27	1.9
N	150000	200000	1	150000	60000	60000	NA	NA
Na	20	20	1	20	10	10	140	76
Nb	300	300	100	300	100	1000	NA	NA
Nd	100	30	100	100	100	1000	NA	NA
Ni	100	100	100	100	500	100	71	21
Np	30	21	2500	30	10	1000	NA	NA
O	1	1	NA	1	1	1	NA	NA
Os	35	10	NA	35	100	1000	NA	NA
P	50000	50000	28000	50000	30000	20000	NA	140000
Pa	10	10	300	10	1000	10	NA	NA
Pb	300	300	200	300	300	100	370	25
Pd	10	10	50	10	10	2000	NA	NA
Pm	30	30	500	30	100	1000	NA	NA
Po	100	50	2000	100	2000	20000	NA	36
Pr	100	30	100	100	10	10	NA	NA
Pt	35	NA	NA	35	100	1000	NA	NA
Pu	30	30	1000	30	1	100	NA	21000
Ra	50	50	950	50	100	100	210	4
Rb	2000	2000	500	2000	10	20	6100	4900
Re	12000	120	NA	12000	30	30	NA	NA
Rf	NA	NA	NA	40	30	30	NA	NA
Rh	10	10	10	300	100	1000	NA	NA
Ru	10	100	10	10	1	2000	NA	55
S	1000	800	2	1000	5	5	NA	NA
Sb	100	100	1000	100	1000	300	71	37
Sc	100	100	750	100	2	10000	930	190
Se	200	170	6000	200	4000	1000	6900	6000
Si	20	20	50000	20	10	30	NA	NA
Sm	25	30	500	25	100	1000	NA	NA
Sn	3000	3000	50000	3000	1000	300	NA	NA
Sr	60	60	4	60	1	10	190	2.9

TABLE 6.6.1 (Cont.)

	RESRAD	PNNL (Staven et al. 2003)		NCRP 1996			IAEA 2010a	
Element	Freshwater Fish	Freshwater Fish	Saltwater Fish	Freshwater Fish	Saltwater Fish	Saltwater Shellfish	Freshwater Whole Fish	Freshwater Fish Muscle
Ta	100	300	60	100	30	30	NA	NA
Tb	25	30	60	25	25	1000	750	410
Tc	20	20	30	20	10	1000	NA	NA
Te	400	400	1000	400	1000	10000	420	150
Th	100	100	600	100	10000	1000	190	6
Ti	1000	NA	NA	1000	1000	1000	370	190
Tl	10000	10000	5000	10000	10000	15000	590	900
Tm	25	NA	NA	25	25	1000	NA	NA
U	10	10	50	10	1	10	2.4	0.96
V	200	NA	NA	200	400	500	290	97
W	1200	10	10	12000	30	30	NA	NA
Y	30	30	20	30	10	1000	31	40
Yb	200	NA	NA	200	500	500	NA	NA
Zn	1000	350	1000	1000	2000	20000	4700	3400
Zr	300	300	50	300	100	1000	95	22

[a] NA = value not available.

parameter values (IAEA 2010a) included both whole-body fish bioaccumulation factors and fish muscle bioaccumulation factors. The variability in the fish bioaccumulation factor was assumed to follow a lognormal probability distribution. Table 6.6.2 compares the RESRAD (onsite) default fish bioaccumulation factor distributions with those of other references.

6.6.5 Crustacea and Mollusc Bioaccumulation Factors

The default values of crustacea bioaccumulation factors compiled from multiple sources were reviewed in 1993 (Wang el al. 1993). On the basis of that data review, the default values in the RESRAD (onsite) code were revised. Since the publication of the 1993 report, many more radionuclides have been added to the RESRAD family of codes, and a number of new publications on bioaccumulation factors have become available. The other published radiological assessments used for comparison of crustacean and mollusc bioaccumulation factors in this handbook are those of Staven et al. (2003) and the IAEA (2009, 2010a, and the IAEA wildlife transfer factor database (IAEA 2014). To fill in the missing data on FW invertebrates, data from two older references—Simpson and McGill (1980) and NRC Regulatory Guide 1.109, Rev. 1 (NRC 1977) are used.

TABLE 6.6.2 Comparison of RESRAD Default Fish Transfer Factor Distributions with Those Published by IAEA

	Yu et al. 2000 RESRAD (onsite) Default		IAEA 2010a Whole Fish		IAEA 2010a Fish Muscle	
Element	μ	σ	μ	σ	μ	Σ
Ac	2.7	1.1	NA[a]	NA	NA	NA
Ag	1.6	1.1	4.7	0.3	4.7	0.4
Al	6.2	1.1	4.2	2.0	3.9	1.4
Am	3.4	1.1	NA	NA	NA	NA
As	5.7	1.1	6.0	0.8	5.8	0.7
Au	3.5	1.1	5.7	0.8	5.5	0.7
Ba	1.4	1.1	3.9	0.5	0.2	1.2
Be	4.6	1.1	NA	NA	NA	NA
Bi	2.7	1.1	NA	NA	NA	NA
Br	6	1.1	5.1	0.8	4.5	0.8
C	10.8	1.1	NA	NA	12.9	1.1
Ca	6.9	1.1	6.9	1.2	2.5	0.9
Cd	5.3	1.1	NA	NA	NA	NA
Ce	3.4	1.1	2.5	1.0	3.2	2.3
Cf	3.2	1.1	NA	NA	NA	NA
Cl	6.9	1.1	4.6	0.5	3.9	0.8
Cm	3.4	1.1	NA	NA	NA	NA
Co	5.7	1.1	6.0	0.5	4.3	0.9
Cr	5.3	1.1	5.3	0.7	3.7	0.7
Cs	7.6	0.7	8.0	1.0	7.8	0.9
Cu	5.3	1.1	5.6	0.4	5.4	0.5
Eu	3.9	1.1	5.0	1.2	4.9	1.6
F	2.3	1.1	NA	NA	NA	NA
Fe	5.3	1.1	4.9	1.7	5.1	1.9
Ga	8.3	1.1	NA	NA	NA	NA
Gd	3.2	1.1	NA	NA	NA	NA
H	0	0.1	NA	NA	NA	NA
Hf	NA	NA	7.6	1.2	7.0	0.6
Hg	6.9	1.1	8.4	0.8	8.7	0.6
Ho	3.2	1.1	NA	NA	NA	NA
I	3.7	1.1	6.5	0.7	3.4	0.9
In	9.2	1.1	NA	NA	NA	NA
Ir	2.3	1.1	NA	NA	NA	NA
K	6.9	1.1	8.3	0.7	8.1	0.5
La	3.4	1.1	2.8	1.2	3.6	1.6
Mg	NA	NA	4.7	1.1	3.6	0.8
Mn	6	1.1	6.1	1.4	5.5	1.9
Mo	2.3	1.1	3.3	0.6	0.6	0.7
N	12	1.1	NA	NA	NA	NA
Na	3	1.1	4.9	0.7	4.3	1.1
Nb	5.7	1.1	NA	NA	NA	NA

TABLE 6.6.2 (Cont.)

	Yu et al. 2000 RESRAD (onsite) Default		IAEA 2010a			
			Whole Fish		Fish Muscle	
Element	μ	σ	μ	σ	μ	Σ
Nd	4.6	1.1	NA	NA	NA	NA
Ni	4.6	1.1	4.3	0.7	3.0	0.6
Np	3.4	1.1	NA	NA	NA	NA
Os	NA	NA	NA	NA	11.8	0.1
P	10.8	1.1	NA	NA	NA	NA
Pa	2.3	1.1	NA	NA	NA	NA
Pb	5.7	1.1	5.9	1.1	3.2	1.1
Pd	3	1.1	NA	NA	NA	NA
Pm	3.4	1.1	NA	NA	NA	NA
Po	4.6	1.1	NA	NA	3.6	1.5
Pr	4.6	1.1	NA	NA	NA	NA
Pu	3.4	1.1	NA	NA	10.0	1.0
Ra	3.9	1.1	NA	NA	1.4	1.9
Rb	7.6	1.1	8.7	0.5	8.5	0.5
Rh	3	1.1	NA	NA	NA	NA
Ru	3	1.1	NA	NA	NA	NA
S	6.9	1.1	NA	NA	NA	NA
Sb	4.6	1.1	4.3	2.3	3.6	1.5
Sc	4.6	1.1	6.8	1.3	5.2	0.7
Se	5.1	1.1	8.8	0.3	8.7	0.3
Sm	3.2	1.1	NA	NA	NA	NA
Sn	8	1.1	NA	NA	NA	NA
Sr	4.1	1.1	5.2	0.8	1.1	1.4
Ta	4.6	1.1	NA	NA	NA	NA
Tb	3.2	1.1	6.6	1.0	6.0	0.6
Tc	3	1.1	NA	NA	NA	NA
Te	6	1.1	6.0	0.4	5.0	0.4
Th	4.6	1.1	NA	NA	NA	NA
Ti	NA	NA	5.9	0.6	5.2	0.3
Tl	9.2	1.1	6.4	0.6	6.8	1.0
U	2.3	1.1	NA	NA	0.0	2.5
V	NA	NA	5.7	0.7	4.6	0.6
W	3.2	1.1	NA	NA	NA	NA
Y	3.4	1.1	3.4	0.5	3.7	0.9
Zn	6.9	1.1	8.5	0.6	8.1	1.1
Zr	5.7	1.1	4.6	0.4	3.1	0.9

[a] NA = value not available.

Note: μ = mean of the underlying normal distribution and σ = standard deviation of the underlying normal distribution.

Table 6.6.3 compares the current RESRAD (onsite) default values for FW crustacea bioaccumulation factor with those from other references. The PNNL (Staven et al. 2003) report also included SW crustacea and FW and SW mollusk bioaccumulation factors. The IAEA handbook of parameter values (IAEA 2010a) compiled FW invertebrate bioaccumulation factors and suggested that those can be used for crustacea and molluscs. If a default value for a crustacea bioaccumulation factor is not available, −1 is assigned in the code.

TABLE 6.6.3 Comparison of RESRAD Default Freshwater Crustacea Bioaccumulation Factors with Those from Other References

	RESRAD Default	PNNL (Staven et al. 2003)		IAEA 2010a	IAEA 2014	NUREG/CR-1276 (Simpson and McGill 1980)	NRC 1977
Element	Freshwater Crustacea	Freshwater Crustacea	Freshwater Mollusc	Freshwater Invertebrate	Freshwater Mollusc	Freshwater Invertebrate	Freshwater Invertebrate
Ac	1000	1000	1000	NA[a]	NA	NA	1000
Ag	770	200	200	230	NA	NA	770
Al	1000	NA	NA	3400	NA	NA	63
Am	1000	100	100	2400	6600	NA	1000
As	300	300	300	1500	NA	NA	40
At	-1	NA	NA	NA	NA	NA	5
Au	1000	NA	NA	1400	NA	NA	50
B	-1	NA	NA	NA	NA	NA	50
Ba	200	200	200	140	NA	200	200
Be	10	50	50	NA	NA	NA	10
Bi	10	100000	100000	NA	NA	NA	24
Bk	-1	NA	NA	NA	NA	NA	1000
Br	330	330	330	1300	NA	330	330
C	9100	9000	9000	65000	NA	9100	9100
Ca	330	2000	2000	34	1100	NA	330
Cd	2000	10000	10000	100	21000	NA	2000
Ce	1000	1000	1000	430	660	1000	1000
Cf	1000	1000	1000	NA	NA	NA	1000
Cl	190	50	50	160	NA	NA	100
Cm	1000	1000	1000	9500	17	NA	1000
Co	200	2000	2000	22	960	200	200
Cr	2000	2000	2000	300	NA	2000	2000
Cs	100	500	500	23	100	1000	100
Cu	400	400	400	42	NA	400	400
Dy	-1	1000	5000	NA	700	NA	1000
Er	-1	1000	1000	NA	460	NA	1000
Es	-1	NA	NA	NA	NA	NA	100
Eu	1000	3000	3000	220	1400	NA	1000
F	100	100	100	NA	NA	NA	100
Fe	3200	100	100	2000	1800	3200	3200

TABLE 6.6.3 (Cont.)

	RESRAD Default	PNNL (Staven et al. 2003)		IAEA 2010a	IAEA 2014	NUREG/ CR-1276 (Simpson and McGill 1980)	NRC 1977
Element	Freshwater Crustacea	Freshwater Crustacea	Freshwater Mollusc	Freshwater Invertebrate	Freshwater Mollusc	Freshwater Invertebrate	Freshwater Invertebrate
Fm	-1	NA	NA	NA	NA	NA	100
Fr	-1	NA	NA	NA	NA	NA	100
Ga	-1	10000	10000	NA	NA	NA	670
Gd	1000	2000	5000	NA	940	NA	1000
Ge	20000	NA	NA	NA	NA	NA	33
H	1	NA	NA	NA	NA	0.9	0.9
Ha	-1	NA	NA	NA	NA	NA	NA
Hf	-1	1000	3000	1400	NA	NA	6.7
Hg	20000	20000	20000	750	NA	NA	100000
Ho	1000	3000	3000	NA	480	NA	1000
I	5	100	100	17	79	5	5
In	15000	10000	10000	NA	NA	NA	100000
Ir	200	200	200	NA	NA	NA	300
K	200	NA	NA	590	NA	NA	830
La	1000	1000	1000	350	2900	1000	1000
Li	-1	NA	NA	NA	NA	NA	40
Lr	-1	NA	NA	NA	NA	NA	
Lu	-1	NA	NA	1100	180	NA	1000
Md	-1	NA	NA	NA	NA	NA	
Mg	100	NA	NA	32	25	NA	100
Mn	90000	100000	100000	21	10000	90000	90000
Mo	10	100	100	0.45	250	10	10
N	0	1	1	NA	NA	NA	150000
Na	200	100	100	3.4	67	200	200
Nb	100	50	50	NA	NA	100	100
Nd	1000	NA	NA	NA	1400	1000	1000
Ni	100	500	500	NA	120	100	100
Np	400	30	30	9500	NA	400	400
O	-1	NA	NA	NA	NA	NA	-0.92
Os	-1	NA	NA	NA	NA	NA	300
P	20000	100000	100000	NA	NA	NA	20000
Pa	110	30	30	NA	NA	NA	110
Pb	100	500	500	22	2300	NA	100
Pd	300	2000	2000	NA	NA	NA	300
Pm	1000	3000	3000	NA	NA	NA	1000
Po	20000	20000	20000	NA	110000	NA	20000
Pr	1000	1000	1000	NA	1600	1000	1000
Pt	-1	NA	NA	NA	NA	NA	300
Pu	100	100	100	7400	2300	NA	100
Ra	250	1000	1000	100	14000	NA	250
Rb	1000	1000	1000	2000	94	1000	1000

TABLE 6.6.3 (Cont.)

Element	RESRAD Default	PNNL (Staven et al. 2003)		IAEA 2010a	IAEA 2014	NUREG/ CR-1276 (Simpson and McGill 1980)	NRC 1977
	Freshwater Crustacea	Freshwater Crustacea	Freshwater Mollusc	Freshwater Invertebrate	Freshwater Mollusc	Freshwater Invertebrate	Freshwater Invertebrate
Re	-1	NA	NA	NA	NA	NA	60
Rf	-1	NA	NA	NA	NA	NA	
Rh	300	300	300	NA	NA	300	300
Ru	300	300	300	0.039	NA	300	300
S	240	100	100	NA	NA	NA	100
Sb	10	100	100	210	49	NA	10
Sc	1000	1000	1000	3500	NA	NA	1000
Se	170	2000	2000	570	2400	NA	170
Si	-1	10000	10000	NA	NA	NA	25
Sm	1000	3000	3000	1600	1300	NA	1000
Sn	1000	10000	10000	NA	NA	NA	1000
Sr	100	100	100	270	350	100	100
Ta	30	3000	3000	NA	NA	NA	670
Tb	1000	1000	1000	NA	NA	NA	1000
Tc	5	100	100	26	NA	5	5
Te	75	6100	6100	NA	NA	6100	100000
Th	500	100	100	2900	NA	NA	500
Ti	-1	NA	NA	NA	NA	NA	3000
Tl	15000	1000	5000	NA	NA	NA	15000
Tm	-1	NA	NA	NA	310	NA	1000
U	60	100	100	170	540	NA	60
V	3000	NA	NA	390	560	NA	3000
W	10	10	10	NA	NA	10	10
Y	1000	1000	1000	NA	2100	NA	1000
Yb	-1	NA	NA	NA	400	NA	1000
Zn	10000	10000	10000	92	NA	10000	10000
Zr	6.7	50	50	NA	NA	6.7	6.7

[a] NA = value not available.

7 HUMAN INTAKE PARAMETERS

7.1 DRINKING WATER INTAKE RATE

7.1.1 Definition

This a dietary factor for water intake by an individual in one year. The default water intake rate is a national average that is site-independent. It includes both contaminated and uncontaminated intake rates; the contaminated fraction is specified as a separate input.

7.1.2 Summary of Literature Review

The EPA used 2 L/d as the average amount of water consumed by an adult (EPA 1990a); this includes juices and beverages containing tap water (e.g., coffee). However, this value was established by the U.S. Army in determining the amount of water needed per person in the field and is believed to be an overestimate.

The National Academy of Sciences (NAS) (1977) calculated the average consumption rate of water to be 1.63 L/d per person. It is reasonable to assume that people in physically oriented occupations or living in warmer regions may have an intake rate exceeding this level. Although the consumption rate of 1.63 L/d seems to have a more scientific basis than the 2 L/d rate, the NAS (1977) still adopted the larger volume (i.e., 2 L/d) to represent the intake rate for the majority of people.

Several other drinking water intake rates have been suggested. The National Cancer Institute (NCI), in an investigation of the possible relationship between bladder cancer and drinking water, interviewed approximately 9,000 individuals by using a standardized questionnaire and suggests that the overall average tap water consumption rate is 1.39 L/d (Cantor et al. 1987). According to the NCI’s distribution data, 1.3 L/d is the approximate value of the 50th percentile and 2.0 L/d is the approximate value of the 90th percentile.

Gillies and Paulin (1983) suggest an average rate of 1.256 (+ 0.39) L/d and a 90th percentile rate of 1.9 L/d on the basis of a survey conducted in New Zealand. On the basis of data from the U.S. Food and Drug Administration’s total diet study, Pennington (1983) reported an average daily fluid consumption rate for water and water-based foods of 1.2 L/d. The International Commission on Radiological Protection (ICRP) has summarized the intake levels for adults as ranging from about 0.4 L/d to about 2.2 L/d under normal conditions (ICRP 1975).

The EPA (1984a) used data collected by the USDA in its 1977–1978 nationwide food consumption survey to determine daily beverage intake levels by age. The daily beverage intake level for adults ranged from 1.24 to 1.73 L. The EPA (1990a) has suggested that the average adult drinking water consumption rate is 1.4 L/d; the reasonable worst-case value is 2.0 L/d on the basis of the above studies. These values correspond to 510 and 730 L/yr, respectively, if a

365-day year is used. Further evidence to support these values is provided by Pennington (1983) and Cantor et al. (1987), who reported average total fluid intake rates of 1.7 and 1.87 L/d, respectively, among adults. Thus, the average water consumption rate should be less than the 2.0 L/d commonly used. Although data are available for the intake rate for the reasonable worst case, from the reported value of 1.90 L/d for the 90th percentile by Gillies and Paulin (1983) and 2.0 L/d by Cantor et al. (1987), it is reasonable to assume a worst-case value of 2.0 L/d in risk assessment.

Some of the recent nationwide surveys such as the Continuing Survey of Food Intake by Individuals (CSFII) or the National Health and Nutrition Examination Survey (NHANES) collect data on dietary intake in the United States population. EPA (2011) used the survey data to estimate the overall per capita consumption rate of different food types. The EPA (2011) has recommended that the average adult (>21 years) per-capita drinking water intake rate is 1 L/d, the 90th-percentile drinking water intake rate is about 2.5 L/d, and the 95th-percentile drinking water intake rate is about 3 L/d. Table 7.1.1 lists the EPA-recommended drinking water intake rates for different age groups.

The drinking water intake rate used by RESRAD does not differentiate the contaminated fraction from the uncontaminated fraction. A separate input parameter, that is, fraction of drinking water from a specific site, is used to adjust the contaminated fraction, and site-specific data can be used to reflect more realistic conditions.

TABLE 7.1.1 Recommended Drinking Water Intake Rates

	Per-capita Intake (mL/day)			
Age Group	Mean	50th Percentile	90th Percentile	95th Percentile
Birth to 1 month	184	0	687	839
1 to <3 months	227	0	804	896
3 to <6 months	362	148	928	1056
6 to <12 months	360	218	885	1055
1 to <2 years	271	188	624	837
2 to <3 years	317	246	683	877
3 to <6 years	327	245	746	959
6 to <11 years	414	297	1000	1316
11 to <16 years	520	329	1338	1821
16 to <18 years	573	375	1378	1783
18 to <21 years	681	355	1808	2368
>21 years	1043	787	2414	2958
>65 years	1046	886	2272	2730
All ages	869	560	2170	2717

Source: EPA (2011).

7.1.3 Data Input Requirements

In the RESRAD (onsite) and RESRAD-OFFSITE codes, the drinking water intake should be entered in units of liters per year (L/yr). The default value currently used is 510 L/yr. The default drinking water intake is more conservative than the EPA-recommended mean values in all age groups (Table 7.1.1). For the probabilistic analysis, use the distribution developed for drinking water intake in NUREG/CR-6697 (Yu et al. 2000).

7.2 INHALATION RATE

7.2.1 Definition

Inhalation rate is the annual air intake in m^3/yr. It varies with activity level, age, weight, sex, and general physical condition. Anthropometric data (EPA 1985) have been used to propose several formulas for calculating the inhalation rate for a human at rest. However, in general, the formulas are based on measurements from relatively small sample sizes and are limited to calculating the inhalation rate at rest only.

7.2.2 Summary of Literature Review

EPA (1985) has compiled the available data, most of which are from early studies, and has derived inhalation rates expressed in cubic meters per hour (m^3/h). Inhalation rates were compiled for each age/sex group at rest and at light, moderate, and heavy activity levels. The activity levels were categorized according to criteria developed by the Environmental Criteria and Assessment Office of the EPA for the air quality document for ozone. A male adult with a body weight of 70 kg was used as a reference base. Activity level categories for the other age/sex groups were extrapolated from the criteria for male adults on the basis of body weight (AIHA 1971). Table 7.2.1 gives a summary of human inhalation rates at different age/sex/activity levels (EPA 1985). Resting is characterized by such activities as watching television, reading, or sleeping. Light activity includes level walking, meal cleanup, care of laundry and clothes, domestic work and other miscellaneous household chores, attending to personal needs, photography, hobbies, and conducting minor indoor repairs and home improvements. Moderate activity includes climbing stairs, heavy indoor cleanup, and performing major indoor repairs and alterations (e.g., remodeling). Heavy activity consists of vigorous physical exercise such as weight lifting, dancing, or riding an exercise bike.

Assuming 16 hours of light activity and 8 hours of resting, the ICRP (1981) has reported a 23-m^3/d inhalation rate for adult males and a 21-m^3/d rate for adult females, yielding an average value of 22 m^3/d (8,030 m^3/yr) for adults.

Data presented by the EPA (1985) suggest lower inhalation rates for light and resting activity levels. Using the same assumption as the ICRP (1981), the daily inhalation rate would be about 14 m^3/d (5,110 m^3/yr). EPA also estimated the daily inhalation rate for moderate and

TABLE 7.2.1 Summary of Human Inhalation Rates (m^3/h) for Men, Women, and Children by Activity Level

	Activity Level			
	Resting	Light	Moderate	Heavy
Adult male	0.7	0.8	2.5	4.8
Adult female	0.3	0.5	1.6	2.9
Average adult	0.5	0.6	2.1	3.9
Child, age 6	0.4	0.8	2.0	2.4
Child, age 10	0.4	1.0	3.2	4.2

Source: EPA (1985).

heavy activity levels; therefore, it is possible to estimate the total inhalation rate for any combination of activity levels. The EPA's data suggest that the maximum inhalation rate is roughly twice the reported mean rates for all activity levels.

The EPA (1990a) made the following recommendations on the basis of the above-mentioned data: 20 m^3/d (7,300 m^3/yr) should be used as the average adult daily inhalation rate and 30 m^3/d (11,000 m^3/yr) as the reasonable worst-case inhalation rate, when activity patterns are unknown. For exposure scenarios in which the distribution of activity patterns is known, the values in Table 7.2.1 should be used for calculations because they are more representative rates.

For an individual performing outdoor activities, a typical activity mix would consist of 37% at a moderate activity level, 28% at both resting and light activity levels, and 7% at a heavy activity level, which results in a 1.4 m^3/h (12,300 m^3/yr) inhalation rate. A reasonable worst-case outdoor inhalation rate would consist of 50% at a heavy activity level and 50% at a moderate activity level, with an inhalation rate of 3.0 m^3/h (26,300 m^3/yr), according to Table 7.2.1.

For an individual performing indoor activities, an average assumption would consist of 48% of the time at both a resting and light activity level, 3% at a moderate activity level, and 1% at a heavy activity level. A reasonable worst-case level would consist of 25% at a resting activity level, 60% at a light activity level, 10% at a moderate activity level, and 5% at a heavy activity level. The first assumption yields an average inhalation rate of 0.63 m^3/h (5,500 m^3/yr), and the second one yields a reasonable worst-case inhalation rate of 0.89 m^3/h (7,800 m^3/yr).

Many new data on inhalation rate have been reported since ICRP Publication 23 was issued in 1975 (ICRP 1975), and the scope for radiological protection has been broadened to extend the assessment to different age groups. ICRP Publication 89 (ICRP 2002) provides a comprehensive and consistent set of age- and gender-specific reference values for many physiological parameters, including inhalation rate. The reference values are provided for both males and females in six age groups: newborn, 1 year, 5 years, 10 years, 15 years, and adults. Table 7.2.2 lists the reference values for inhalation rate at different levels of physical activity.

TABLE 7.2.2 Reference Values for Inhalation Rates (m^3/h) at Different Physical Activity Levels

Age Group	Resting (Sleeping)	Sitting Awake	Light Exercise	Heavy Exercise
3 months	0.1	NA	0.2	NA
1 year	0.2	0.2	0.4	NA
5 years	0.2	0.3	0.6	NA
10 years, male	0.3	0.4	1.1	2.2
10 years, female	0.3	0.4	1.1	1.8
15 years, male	0.4	0.5	1.4	2.9
15 years, female	0.4	0.4	1.3	2.6
Adult, male	0.5	0.5	1.5	3.0
Adult, female	0.3	0.4	1.3	2.7

Source: ICRP (2002).

For an adult male, the inhalation rate during heavy exercise is 6 times higher than the resting inhalation rate. An individual may be involved in various different activities in a day, resulting in different daily inhalation rates. Table 7.2.3 lists the reference values of the time spent in different activities and the associated reference inhalation rates for different age groups. The daily average inhalation rate varies from 2.8 m^3/d for a 3-month-old to 22.2 m^3/d for an adult male. The daily inhalation rate in sedentary workers and heavy workers may be different because of the difference in physical activities performed. Table 7.2.4 lists the reference inhalation rates for both sedentary and heavy workers and their associated physical activities.

The available studies on inhalation rates have been summarized by the EPA in the *Exposure Factors Handbook* published in 2011 (EPA 2011). EPA has used some recent key studies (EPA 2009; Brochu et al. 2006; Arcus-Arth and Blaisdell 2007; and Stifelman 2007) to recommend the mean and 95th percentile values for inhalation in different age groups. Table 7.2.5 lists the recommended inhalation values for long-term exposure along with the inhalation values from other studies. The EPA-recommended mean values for inhalation rate vary from 5.4 m^3/d (for birth to <1 year) to 16.3 m^3/d (16 to <21 years). The ICRP reference values for inhalation rates are, in general, greater than the EPA recommended mean values for an individual over 10 years old. Table 7.2.6 lists the EPA-recommended values for short-term exposure in different physical activities. The physical activities are categorized as sleep, sedentary, light, moderate, and heavy.

7.2.3 Data Input Requirements

In the RESRAD (onsite) and RESRAD-OFFSITE codes, the inhalation rate should be entered in units of cubic meters per year (m^3/yr). The default value used in the codes is 8,400 m^3/yr.

TABLE 7.2.3 Reference Values for Inhalation Rates and the Time Spent in Different Physical Activities

Age Group	Inhalation Rate (m^3/day)	Time Spent (h/day) Sleep	Sitting	Light Exercise	Heavy Exercise
3 months	2.8	17		7	NA
1 year	5.1	14	3.3	6.7	NA
5 years	8.8	12	4	8	NA
10 years, male	15.2	10	4.7	9.3	NA
10 years, female	15.2	10	4.7	9.3	NA
15 years, male	20.1	10	5.5	7.5	1
15 years, female	15.8	10	7	6.8	0.25
Adult, male	22.2	8	6	9.8	0.25
Adult, female	18.2	8.5	5.4	9.9	0.19

Source: ICRP (2002).

TABLE 7.2.4 Reference Values for Inhalation Rates of Sedentary and Heavy Workers

	Inhalation Rate (m^3/day)		
	Sedentary Worker		
Activity	Male	Female	Heavy Worker
Sleeping (8 h)	3.6	2.6	3.6
Occupational (8 h)[a]	9.6	7.9	13.5
Nonoccupational (8 h)[b]	9.7	8	9.7
Total inhalation rate	22.9	18.5	26.8

[a] For sedentary workers, it is assumed that 1/3 of occupational time is spent in sitting and 2/3 in light exercise. For heavy workers, it is assumed that 7/8 of occupational time is spent in light exercise and 1/8 in heavy exercise

[b] It is assumed that 1/2 of nonoccupational time is spent in sitting, 3/8 in light exercise, and 1/8 in heavy exercise.

Source: ICRP (2002).

TABLE 7.2.5 EPA-Recommended Inhalation Values for Long-Term Exposure

		Mean (m^3/day) from Key Studies					95th Percentile (m^3/day) from Key Studies		
Age Group	Recommended Mean (m^3/day)	EPA (2009)	Brochu et al. (2006)	Arcus-Arth and Blaisdell (2007)	Stifelman (2007)	Recommended 95th Percentile	EPA (2009)	Brochu et al. (2006)	Arcus-Arth and Blaisdell (2007)
Birth to <1 year	5.4	8.64	3.72	5.7	3.4	9.2	12.67	4.9	9.95
1 to <2 years	8	13.41	4.9	8.77	4.9	12.8	18.22	6.43	13.79
2 to <3 years	8.9	12.99	7.28	9.76	5.7	13.7	17.04	9.27	14.81
3 to <6 years	10.1	12.4	7.28	11.22	9.3	13.8	15.17	9.27	17.09
6 to <11 years	12	12.93	9.98	13.42	11.5	16.6	17.05	12.85	19.86
11 to <16 years	15.2	14.34	14.29	16.98	15	21.9	19.23	19.02	27.53
16 to <21 years	16.3	15.44	14.29	18.29	17	24.6	20.89	19.02	33.99
21 to <31 years	15.7	16.3	14.59	NA[a]	NA	21.3	23.57	19	NA
31 to <41 years	16	17.4	14.99	NA	NA	21.4	24.3	18.39	NA
41 to <51 years	16	18.55	13.74	NA	NA	21.2	24.83	17.5	NA
51 to <61 years	15.7	18.56	13.74	NA	NA	21.3	25.17	17.5	NA
61 to <71 years	14.2	15.43	12.57	NA	NA	18.1	19.76	16.37	NA
71 to <81 years	12.9	14.25	11.46	NA	NA	16.6	17.88	15.3	NA
81 years	12.2	12.97	11.46	NA	NA	15.7	16.1	15.3	NA

Note: Recommended values are the weighted average of key studies.

[a] NA = value not available.

Source: EPA (2011),

TABLE 7.2.6 EPA-Recommended Inhalation Values for Short-Term Exposure

Age Group	Sleep (m^3/minute)		Sedentary/Passive (m^3/minute)		Light Intensity (m^3/minute)		Moderate Intensity (m^3/minute)		High Intensity (m^3/minute)	
	Mean	95th	Mean	95th	Mean	95th	Mean	95th	Mean	95th
Birth to <1 year	3.0×10^{-3}	4.6×10^{-3}	3.1×10^{-3}	4.7×10^{-3}	7.6×10^{-3}	1.1×10^{-2}	1.4×10^{-2}	2.2×10^{-2}	2.6×10^{-2}	4.1×10^{-2}
1 to <2 years	4.5×10^{-3}	6.4×10^{-3}	4.7×10^{-3}	6.5×10^{-3}	1.2×10^{-2}	1.6×10^{-2}	2.1×10^{-2}	2.9×10^{-2}	3.8×10^{-2}	5.2×10^{-2}
2 to <3 years	4.6×10^{-3}	6.4×10^{-3}	4.8×10^{-3}	6.5×10^{-3}	1.2×10^{-2}	1.6×10^{-2}	2.1×10^{-2}	2.9×10^{-2}	3.9×10^{-2}	5.3×10^{-2}
3 to <6 years	4.3×10^{-3}	5.8×10^{-3}	4.5×10^{-3}	5.8×10^{-3}	1.1×10^{-2}	1.4×10^{-2}	2.1×10^{-2}	2.7×10^{-2}	3.7×10^{-2}	4.8×10^{-2}
6 to <11 years	4.5×10^{-3}	6.3×10^{-3}	4.8×10^{-3}	6.4×10^{-3}	1.1×10^{-2}	1.5×10^{-2}	2.2×10^{-2}	2.9×10^{-2}	4.2×10^{-2}	5.9×10^{-2}
11 to <16 years	5.0×10^{-3}	7.4×10^{-3}	5.4×10^{-3}	7.5×10^{-3}	1.3×10^{-2}	1.7×10^{-2}	2.5×10^{-2}	3.4×10^{-2}	4.9×10^{-2}	7.0×10^{-2}
16 to <21 years	4.9×10^{-3}	7.1×10^{-3}	5.3×10^{-3}	7.2×10^{-3}	1.2×10^{-2}	1.6×10^{-2}	2.6×10^{-2}	3.7×10^{-2}	4.9×10^{-2}	7.3×10^{-2}
21 to <31 years	4.3×10^{-3}	6.5×10^{-3}	4.2×10^{-3}	6.5×10^{-3}	1.2×10^{-2}	1.6×10^{-2}	2.6×10^{-2}	3.8×10^{-2}	5.0×10^{-2}	7.6×10^{-2}
31 to <41 years	4.6×10^{-3}	6.6×10^{-3}	4.3×10^{-3}	6.6×10^{-3}	1.2×10^{-2}	1.6×10^{-2}	2.7×10^{-2}	3.7×10^{-2}	4.9×10^{-2}	7.2×10^{-2}
41 to <51 years	5.0×10^{-3}	7.1×10^{-3}	4.8×10^{-3}	7.0×10^{-3}	1.3×10^{-2}	1.6×10^{-2}	2.8×10^{-2}	3.9×10^{-2}	5.2×10^{-2}	7.6×10^{-2}
51 to <61 years	5.2×10^{-3}	7.5×10^{-3}	5.0×10^{-3}	7.3×10^{-3}	1.3×10^{-2}	1.7×10^{-2}	2.9×10^{-2}	4.0×10^{-2}	5.3×10^{-2}	7.8×10^{-2}
61 to <71 years	5.2×10^{-3}	7.2×10^{-3}	4.9×10^{-3}	7.3×10^{-3}	1.2×10^{-2}	1.6×10^{-2}	2.6×10^{-2}	3.4×10^{-2}	4.7×10^{-2}	6.6×10^{-2}
71 to <81 years	5.3×10^{-3}	7.2×10^{-3}	5.0×10^{-3}	7.2×10^{-3}	1.2×10^{-2}	1.5×10^{-2}	2.5×10^{-2}	3.2×10^{-2}	4.7×10^{-2}	6.5×10^{-2}
81 years	5.2×10^{-3}	7.0×10^{-3}	4.9×10^{-3}	7.0×10^{-3}	1.2×10^{-2}	1.5×10^{-2}	2.5×10^{-2}	3.1×10^{-2}	4.8×10^{-2}	6.8×10^{-2}

Source: EPA (2011).

In the codes, the yearly inhalation rate is used. This rate is an average value that accounts for different activity levels both indoors and outdoors. Therefore, a site-specific value can be obtained with the assumed exposure scenario and an activity profile. The fraction of time spent on-site and off-site should not affect this input parameter, however, because in the RESRAD (onsite) and RESRAD-OFFSITE dose calculations, an occupancy factor is automatically derived and used for adjusting the calculated dose.

7.3 SOIL AND DUST INGESTION RATE

7.3.1 Definition

This parameter is the accidental ingestion rate of soil material or soil dust.

7.3.2 Summary of Literature Review

Soil or dust ingestion results from different behaviors such as mouthing (ingesting substances that are not considered to be food), contacting dirty hands, eating dropped food, or consuming soil or dust directly. Children, especially those under 18 months, tend to mouth. When this behavior extends beyond the age of 18 months, the child is said to practice pica (Barltrop 1966; Robischon 1971; Ziai 1983). Soil-pica is recurrent ingestion of high amounts of soil. Many factors, such as nutrition, quality of care, and parental relationship (Bicknell 1974; Glickman 1981; Danford et al. 1982; Behrman and Vaughan 1983; Forfar and Arneil 1984; Bellinger et al. 1986), influence the extent of this behavior. However, it is believed that a child who practices pica is no different from one who does not, because pica cannot be consistently predicted (Feldman 1986), even though severe pica usually occurs among grossly disturbed or mentally retarded children. The intentional ingestion of earths that is usually associated with cultural practices is called geophagy (EPA 2011).

According to the *Exposure Factors Handbook* (EPA 2011), soil ingestion includes both soil and outdoor settled dust, and dust ingestion includes indoor settled dust only. Soil is defined as “particles of unconsolidated mineral and/or organic matter from the earth’s surface that are located outdoors, or are used indoors to support plant growth.” It includes outdoor settled dust. Indoor settled dust is defined as “particles in building interiors that have settled onto objects, surfaces, floor, and carpeting.” Outdoor settled dust is defined as “particles that have settled onto outdoor objects and surfaces due to either wet or dry deposition.”

According to the literature, a wide variety of substances are ingested: soil, clay, sand, dust, grass, leaves, plaster, hair, starch, paint chips, string, soap, wood, powders, chalk, and paper. According to Gavrelis et al. (2011), pica behavior is most prevalent among children aged 1 to <3 years, whereas geophagy behavior is extremely rare among children.

On the basis of observational data, children are most likely to ingest soil from the age of 1 to 6 (Gavrelis et al. 2011; Cooper 1957; Sayre et al. 1974; Charney et al. 1980;

Walter et al. 1980). Beyond age 6 or 7, ingestion of nonfood substances is usually caused by inadvertent ingestion or developmental problems. Paustenbach et al. (1986) summarized the normal amount of soil ingested by children on the basis of the age of the child. Vermeer and Frate (1979) pointed out that the environmental setting is also an important factor for children in rural areas, who tend to ingest a higher amount of soil. Hawley (1985) used data from the literature to develop scenarios to estimate ingestion amounts for young and older children and adults. He divided each year into two activity periods: May through October, when individuals spend more time outdoors and November through April, when most of the time, weather conditions eliminate outdoor activities. Hawley's study indicated that the amount ingested by young children (2.5 years old, weighing 13.2 kg) during outdoor activity between May and October (5 d/week) is 250 mg/d. During November through April, the ingestion during indoor activity is 100 mg/d. For 6-year-old children, weighing approximately 20.8 kg, the ingestion amount is 50 mg/d during outdoor activity from May through October and 3 mg/d year-round for indoor activity. Working in attics or other uncleaned areas of a house can cause adults (weighing 70 kg) to ingest 110 mg/d of soil for an assumed duration of 12 d/yr. For living-space activities, the ingestion amount is 0.56 mg/d. For outdoor activities from May through October, the ingestion amount is 480 mg/active day, assuming 8 hours is spent outdoors per day, 2 d/week.

According to Binder et al. (1986), the average quantity of soil ingested by children is about 108 mg/d (within a range of 4–708 mg/d). Clausing et al. (1987) estimated that the ingestion rate of children is 105 mg/d, with a range of 23–362 mg/d. Binder et al. (1986) and Clausing et al. (1987) have also provided some limited information on the upper limit of the soil ingestion rate on the basis of evidence that the upper range of the ingestion rate for children is around 800 mg/d or more.

An amount has not been estimated for abnormal soil ingestion behavior among children. However, some evidence suggests that a rate of 5 to 10 g/d may not be unreasonable. ATSDR (2001) estimated that 33% of children ingest more than 10 grams of soil 1 or 2 days a year. The EPA used 5 g/d in its risk assessment for tetrachlorodibenzo-p-dioxin (TCDD) (EPA 1984b). The USDA used a value of 10 g/d in conducting exposure assessments related to the use of sludge in gardens and soils.

After reviewing the limited data available, the EPA (1990a) decided that the studies of Binder et al. (1986) and Clausing et al. (1987) appear to be the most reliable and suggested that an estimate of 0.2 g/d be used as an average value for young children (under the age of 7). An upper range of soil ingestion is 0.8 g/d. For other age groups (children older than 7 years), 0.1 g/d should be used for the soil ingestion amount. These factors account for ingestion of both outdoor soil and indoor dust. Site conditions, such as snow cover, will affect the soil ingestion rate because the cold weather will limit outdoor activities in the winter, and because snow also provides an additional cover for the contaminated soil. However, presently there is no recommended approach to correct for these influences, and any correction should be conservatively applied.

According to EPA guidance (EPA 1990b), soil ingestion should be considered separately for adults and children for a residential scenario. For the first stage (for children), 0.2 g/d is the recommended ingestion rate with an exposure duration of 6 years, and for the second stage, with an exposure duration of 24 years, the recommended ingestion rate is 0.1 g/d. In the RESRAD (onsite) and RESRAD-OFFSITE codes, only one soil ingestion rate is required, which is the yearly intake rate (g/yr) without the differentiation between contaminated soil and uncontaminated soil. The code will automatically adjust this soil ingestion rate using an area factor, an occupancy factor, and a cover-and-depth factor, so that only the contaminated-source contribution is accounted. The input value for the soil ingestion rate depends strongly on the assumed scenario. For a residential scenario with an exposure duration of 30 years, 43.8 g/yr [(36.5 g/yr × 24 yr + 73 g/yr × 6 yr)/30 yr] is an applicable input value to RESRAD, according to EPA (1990b) guidance. The current EPA-recommended exposure duration for the residential scenario is 26 years (EPA 2011, Table 16-108) and that would result in exposure duration of 20 years for adults.

The EPA (1991) has chosen 50 mg/d as the standard default value for adult soil ingestion in the workplace, based on a pilot study by Calabrese et al. (1990). After reviewing the data available in 2011, the EPA (2011) used the studies of Vermeer and Frate (1979) and Davis and Mirick (2006) as the key studies in estimating the adult soil and dust ingestion rates and reconfirmed the soil + dust ingestion value of 50 mg/d. Table 7.3.1 lists the recommended values for daily intake of soil and dust (EPA 2011).

7.3.3 Measurement Methodology

Several methods, including tracer element methodology, biokinetic model comparison methodology, and activity pattern methodology, have been used to characterize soil and dust ingestion. Tracer element methodology, sometimes also called mass balance studies, quantifies the amount of soil ingested by analyzing the quantity of tracer elements in the soil and dust samples from playgrounds, residences, and workplaces. The measured quantities are compared to the amount of tracer elements in feces or urine samples (Davis and Mirick 2006;

TABLE 7.3.1 EPA Recommended Values for Daily Intake of Soil, Dust, and Soil + Dust

	Soil (mg/day)				Dust (mg/day)		Soil + Dust (mg/day)	
Age Group	Mean/ Median	Upper Percentile	Soil-Pica	Geophagy	Mean/ Median	Upper Percentile	Mean/ Median	Upper Percentile
6 week to <1 year	30				30		60	
1 to <21 years	50	200		50,000	60		100	200
Adult	20			50,000	30		50	

Note: Soil includes soil and outdoor settled dust; dust includes indoor settled dust.

Source: EPA (2011).

Calabrese et al. 1989, 1997a,b; Barnes 1990). Davis and Mirick (2006) used the mass-balance approach to calculate soil ingestion for adults and children in the same family. Barnes (1990) also studied children's soil-pica behavior. Van Wijnen et al. (1990) measured tracer elements (titanium, aluminum, and acid soluble residue) in soil and feces.

Some methods estimate soil/dust ingestion from activity pattern data from observational studies, as well as through survey questionnaires that inquire about time spent at different places during a day and the "mouthing" behavior. The collected data from the studies are used, along with the assumptions about transfer parameters and exposure factors, to estimate the soil/dust ingestion. Lepow et al. (1975) measured hand dust by applying preweighed adhesive labels to the hands and weighing the amount of dirt that was removed. They also observed "mouthing" behavior and reported that a child would put his or her fingers into the mouth about 10 times a day. Day et al. (1975) and Duggan and Williams (1977) also measured the amount of dust on children's hands. Tulve el al. (2002) and Black et al. (2005) used video transcriptions to extract children's activity patterns.

Biokinetic model comparison methodology compares the measurement of biomarkers (e.g., levels of contaminants present in blood and urine samples) with model predictions (Hogan et al. 1998; Ozkaynak et al. 2011). Some methods combine biomarker measurements in feces and urine with their presence in environmental media to estimate soil/dust ingestion. Binder et al. (1986) studied the ingestion of soil among children 1 to 3 years of age who wear diapers. Both excreta and soil from play yards were analyzed for materials that were thought to be poorly absorbed in the gut. Clausing et al. (1987) conducted a soil ingestion study by using a tracer element method similar to that of Binder et al. (1986). They also collected fecal samples for six hospitalized, bedridden children to represent a control group.

7.3.4 Data Input Requirements

In the RESRAD (onsite) and RESRAD-OFFSITE codes, the soil ingestion rate should be entered in units of grams per year (g/yr). The default value of 36.5 g/yr is used, which accounts for an average soil intake rate of 0.1 g/d for 365 d/yr. The 0.1 g/d ingestion rate is a value recommended by the EPA for the age group 1 to <21 years (see Table 7.3.1). For the probabilistic analysis, use the distribution developed for the soil ingestion rate in NUREG/CR-6697 (Yu et al. 2000).

7.4 SEAFOOD CONSUMPTION RATE

7.4.1 Definition

This is the fresh weight of the seafood that is consumed each year by a single individual. The consumption rate includes both contaminated and uncontaminated seafood; the contaminated fraction is specified as a separate input.

7.4.2 Summary of Literature Review

National recreational catch data for coastal areas were obtained by the National Marine Fisheries Service (NMFS) in 1985. The NMFS conducted a direct survey of fishermen in the field and an independent telephone survey of households (NMFS 1986). Total fish consumption data were obtained from a one-year survey conducted by NPD Research, Inc., during 1973 and 1974 and funded by the Tuna Research Institute. Questionnaires were answered by 6,980 families representing the U.S. population.

Javitz (1980) used the data obtained by NPD Research, Inc., to calculate the mean and 95th percentile of seafood consumption for seafood consumers in the United States as 14.3 g/d (5.2 kg/yr) and 41.7 g/d (15.2 kg/yr), respectively. The mean average of 14.3 g/d (5.2 kg/yr) for seafood consumption includes 2.1 g/d (0.8 kg/yr) for nonfish seafood consumption, that is, lobsters, oysters, scallops, shrimps, squids, and so forth. Unfortunately, NPD Research, Inc.'s original survey data for seafood consumption did not distinguish between recreationally caught and purchased fish; therefore, this difference is not reflected in the calculated mean and 95th percentile values.

Ruffle et al. (1994) used the same NPD research data to generate the lognormal distributions using a nonlinear optimization method. Table 7.4.1 lists distributions developed for seafood consumption by the general population. Ten regions of the United States (the nine census regions of the U.S and the entire country) for three age groups (ages 1 through 11 years, ages 12 through 18, and ages 19 through 98 years) in three categories of seafood (SW finfish, shellfish, and FW finfish) are considered.

Puffer et al. (1982) conducted 1,059 interviews with sport fishermen in the Los Angeles Harbor area. The interviews revealed that sport fishermen keep 67 to 89% of the finfish and 97% of the shellfish that they catch. The median and 90th percentile seafood (fish plus shellfish) consumption rates of sport fishermen are 37 and 225 g/d, respectively.

Another source for the seafood consumption rate of sport fishermen is a 1981 survey conducted in Commencement Bay at Tacoma, Washington, by Pierce et al. (1981). The sample size (304 fishermen) was smaller than that of Puffer et al. (1982) and the sampling frequency was lower. It was found that over half of the fishermen caught and consumed fish weekly. Pierce et al. (1981) concluded that the mean average seafood consumption rate for the surveyed fishermen was 23 g/d (within a range of 12-54 g/d); the 90th percentile was 54 g/d.

Although the surveys conducted by Puffer et al. (1982) and Pierce et al. (1981) are limited to the West Coast, the EPA (1990a) considers these studies to be representative of actual annual consumption rates for recreational fishermen. By averaging the results of these two surveys, the EPA (1990a) has suggested that the 50th and 90th percentile seafood consumption rates of fishermen are 30 g/d (11 kg/yr) and 140 g/d (51 kg/yr), respectively.

Because sport fishermen and their families consume much more seafood than other people, the EPA recommends that consumption rates of fishermen based on the surveys by Puffer et al. (1982) and Pierce et al. (1981) be used as comparative references for any area where there is a large body of water and widespread contamination is possible.

TABLE 7.4.1 Best Fit Lognormal Distribution Parameters for Seafood Consumption from NPD Research Survey

		Saltwater Finfish		Shellfish		Freshwater Finfish	
Age Group	Region	μ	σ	μ	σ	μ	σ
Adults	All	2.311	0.72	1.37	0.858	0.334	1.183
	New England	2.57	0.572	2.01	0.794	−1.548	1.433
	Mid Atlantic	2.35	0.763	1.155	1.008	0.373	1.037
	South Atlantic	2.208	0.748	1.702	0.718	0.256	1.102
	East North Central	1.878	0.91	0.494	1.081	−0.111	1.445
	East South Central	2.437	0.541	0.28	1.294	0.662	1.109
	West North Central	1.977	0.811	−0.217	1.379	0.372	1.219
	West South Central	2.49	0.591	1.775	0.733	0.678	1.085
	Mountain	2.045	0.716	0.847	0.986	0.379	1.182
	Pacific	1.919	0.934	1.177	0.938	0.443	0.986
Teenagers	All	1.691	0.83	−0.183	1.092	0.578	0.822
	New England	1.325	1.046	−2.247	1.963	−7.698	4.033
	Mid Atlantic	1.374	1.057	−0.784	1.305	−2.584	1.1981
	South Atlantic	2.098	0.508	0.539	0.862	−0.467	1.153
	East North Central	1.874	0.661	−0.959	1.336	0.54	0.93
	East South Central	1.286	0.816	−2.356	1.948	−3.861	2.617
	West North Central	1.258	0.879	−0.239	0.919	0.421	0.889
	West South Central	1.014	1.194	-1.203	1.733	0.541	0.926
	Mountain	1.714	0.61	-1.537	1.64	-2.352	2.129
	Pacific	1.821	0.78	-0.442	1.2	-0.741	1.358
Children	All	0.881	0.97	0.854	0.73	-0.559	1.141
	New England	1.376	0.706	0.986	0.609		
	Mid Atlantic	1.049	0.917	0.724	0.688	−1.415	1.411
	South Atlantic	1.427	0.651	1.215	0.545	−1.133	1.281
	East North Central	1.347	0.658	−0.546	1.25	−1.687	1.637
	East South Central	0.678	1.041	−1.431	1.554	0.264	0.73
	West North Central	−1.396	1.884	0.029	0.745	−0.034	0.848
	West South Central	1.251	0.802	1.191	0.757	−0.089	1.027
	Mountain	1.391	0.65	0.428	0.747	−0.02	0.86
	Pacific	1.307	0.792	0.916	0.657	−0.597	1.171

Source: Ruffle et al. (1994).

The NRC (1977) used values of 2.2, 5.2, and 6.9 kg/yr for average individual fish consumption by children, teenagers, and adults, respectively. Average individual consumption rates of other seafood were 0.33, 0.75, and 1.0 kg/yr for the three different groups. For a worst-case scenario, the fish consumption rates were 6.9, 16, and 21 kg/yr for children, teenagers, and adults, respectively. For other seafood consumption rates, values of 1.7, 3.8, and 5 kg/yr were used. In its decommissioning guidance (Schmidt et al. 2006), the NRC allows use of the mean of the parameter distribution value for an average member of the critical group (Beyeler et al. 1999) for food consumption rates. The mean consumption rate for an average member of the screening group for fish is 21 kg/yr (Beyeler et al. 1999).

Some of the nationwide surveys such as the CSFII and NHANES collect data on dietary intake in the United States population. The survey data can be used to estimate the overall per capita consumption rate of different food types.

Data from 1994–1996 and 1998 CSFII surveys was used by the U.S. EPA to generate per capita intake rates for different seafood types (EPA 2002). Tables 7.4.2 and 7.4.3 list the per capita prepared and uncooked seafood consumption rates, respectively. The tables list mean, 90th percentile, 95th percentile, and 99th percentile estimates for daily per-capita seafood consumption by the U.S. population. For this estimate, quantities of consumed food reported by participants in the 50 states and the District of Columbia were used. The prepared consumption reflects the amount of seafood in prepared food, and uncooked consumption includes the amount of raw and unprocessed seafood. Tables 7.4.4 and 7.4.5 list the consumer-only prepared and uncooked seafood consumption rates, respectively. The consumer-only estimates include the individuals in the survey that consumed seafood, whereas per-capita estimates include the entire survey population.

TABLE 7.4.2 Per-capita Prepared-Seafood Consumption Rates (g/day)

Age	Sample Size	Environment	Mean	90%	95%	99%
All	20,607	Freshwater/Estuarine	4.58	6.63	29.65	91.01
		Marine	8.25	29.2	55.8	114.56
		All	12.83	48.19	78.98	153.17
18 years and older	9596	Freshwater/Estuarine	5.48	11.74	38.01	105.12
		Marine	9.79	38.57	63.76	126.32
		All	15.27	56.23	86.11	162.57
Ages 3 to 5	4391	Freshwater/Estuarine	1.47	0.14	5.13	38.72
		Marine	3.74	11.1	27.92	59.81
		All	5.22	18.94	35.31	72.22
Ages 6 to 10	1670	Freshwater/Estuarine	2.11	0	5.91	60.85
		Marine	4.18	13.12	28.72	78.6
		All	6.3	23.91	39.55	107.75
Ages 11 to 15	1005	Freshwater/Estuarine	3.01	1.41	18.2	69.51
		Marine	5.45	13.94	38.5	102.3
		All	8.46	28.14	60.33	122.23
Ages 16 to 17	363	Freshwater/Estuarine	3.44	0	13.14	81.18
		Marine	4.67	0	24.23	107.78
		All	8.11	18.62	73.81	142.34

Source: EPA 2002.

TABLE 7.4.3 Per-capita Uncooked-Seafood Consumption Rates (g/day)

Age	Sample Size	Environment	Mean	90%	95%	99%
All	20,607	Freshwater/Estuarine	6.3	11.65	41.08	123.94
		Marine	10.58	38.42	74.89	139.23
		All	16.88	63.46	102.29	198.16
18 years and older	9596	Freshwater/Estuarine	7.5	17.37	49.59	143.35
		Marine	12.41	48.92	80.68	150.77
		All	19.91	74.79	111.35	215.7
Ages 3 to 5	4391	Freshwater/Estuarine	2.19	0.05	12.17	52.46
		Marine	5.51	19.75	39.43	82.27
		All	7.7	32.56	51	100.54
Ages 6 to 10	1670	Freshwater/Estuarine	2.99	0	13.06	78.49
		Marine	5.55	18.9	38.42	99.78
		All	8.54	32.62	56.4	144.37
Ages 11 to 15	1005	Freshwater/Estuarine	4.31	2.33	25.77	94.82
		Marine	7.64	25.26	56.48	131.76
		All	11.95	43.42	87.36	170.67
Ages 16 to 17	363	Freshwater/Estuarine	4.55	0	19.32	109.18
		Marine	6.06	0	29.46	135.63
		All	10.61	29.33	83.53	192.54

Source: EPA 2002.

The per capita estimated mean and 95th percentile of uncooked seafood consumption in the United States was 16.88 g/d (6.17 kg/yr) and 102.29 g/d (37.3 kg/yr), respectively. The mean average of 16.88 g/d (6.17 kg/yr) for seafood consumption includes 4.29 g/d (1.57 kg/yr) for nonfish seafood consumption, that is, lobsters, oysters, scallops, shrimps, squids, and so forth. The per capita estimated mean and 95th percentile of prepared seafood consumption in the United States was 12.83 g/d (4.69 kg/yr) and 78.98 g/d (28.8 kg/yr), respectively. The mean average of 12.83 g/d (4.69 kg/yr) for seafood consumption includes 3.70 g/d (1.35 kg/yr) for nonfish seafood consumption.

Data from 1994–1996 and 1998 CSFII surveys were also used to estimate the consumption of different seafood species. Table 7.4.6 lists the different seafood consumption estimates in the general population and for adults (>18 years old).

The top ten favorite seafoods consumed in the United States are shrimp, tuna, salmon, pollock, tilapia, pangasius, catfish, crab, cod, and clams. Table 7.4.7 lists the average per-capita consumption rates for the top 10 seafood species from 2004 to 2012 in the United States (NOAA 2014).

TABLE 7.4.4 Consumers-Only Prepared-Seafood Consumption Rates (g/day)

Age	Sample Size	Environment	Mean	90%	95%	99%
All	2,575	Freshwater/Estuarine	56.27	145.26	188.76	332.92
	3,382	Marine	80.19	168.88	207.57	310.19
	4,391	All	92.02	184.46	249.25	379.04
18 years and older	1,633	Freshwater/Estuarine	59.15	150.15	201	338.21
	1,978	Marine	85.09	168.94	214.08	337.15
	2,634	All	97.56	191.75	253.16	399.45
Ages 3 to 5	442	Freshwater/Estuarine	27.13	72.57	95.63	158.99
	682	Marine	44.51	90.64	119.1	227.6
	834	All	50.15	103.14	133.87	259.99
Ages 6 to 10	147	Freshwater/Estuarine	43.49	121.6	186.72	260.41
	217	Marine	59.37	128.69	159.24	242.54
	270	All	70.62	154.67	218.16	280.92
Ages 11 to 15	107	Freshwater/Estuarine	48.97	126.57	149.9	307.1
	122	Marine	72.42	165.26	203.55	245.55
	172	All	79.55	167.13	208.78	285.18
Ages 16 to 17	28	Freshwater/Estuarine	75.8	158.53	167.82	371.61
	37	Marine	96.86	218.89	237.47	365.29
	52	All	104.06	200.48	241.88	450.96

Source: EPA 2002.

Data from the 2003–2006 NHANES surveys was used by the U.S. EPA to generate per capita intake rates for different seafood types (EPA 2011). Table 7.4.8 lists the EPA recommended values for uncooked seafood intakes. Not all the people in the population may consume seafood; therefore, Table 7.4.8 lists recommended seafood intake rates on a per capita basis and for the consumers only. EPA recommends the use of uncooked seafood intake rates in the dose assessment (EPA 2011). The average per-capita fish and shellfish consumption from the NHANES 2003−2006 surveys for all ages combined was estimated to be 0.22 g/kg-day (Table 7.4.8). The fish and shellfish consumption data in Table 7.4.8 is presented in g/kg-day and therefore is not directly comparable to data presented in Tables 7.4.2-7.4.5.

EPA (2011) lists the mean and 95th-percentile recreational marine seafood consumption rates for the Atlantic, Gulf, and Pacific regions for different age groups. The suggested range of mean and 95th-percentile values for the >18-year age group are 2.0–7.2 g/d and 6.8–26 g/d, respectively. The EPA (2011) has suggested the mean ranges of 5–51 g/day for FW recreational seafood consumption from the statewide surveys.

TABLE 7.4.5 Consumers-Only Uncooked-Seafood Consumption Rates (g/day)

Age	Sample Size	Environment	Mean	90%	95%	99%
All	2,575	Freshwater/estuarine	77.55	195.71	258.13	467.92
	3,382	Marine	102.93	215.32	258.28	395.27
	4,391	All	121.33	241.04	329.47	506.85
18 years and older	1,633	Freshwater/estuarine	81.08	199.62	278.91	505.65
	1,978	Marine	107.9	216.59	269.7	464.24
	2,634	All	127.44	248.37	333.76	518.69
Ages 3 to 5	442	Freshwater/estuarine	40.31	95.16	129.31	204.84
	682	Marine	65.61	125.19	164.84	315.72
	834	All	74.08	148.66	183.81	363.06
Ages 6 to 10	147	Freshwater/estuarine	61.49	156.86	247.69	385.64
	217	Marine	78.45	150.01	201.88	349.99
	270	All	95.46	199.61	312.94	387.41
Ages 11 to 15	107	Freshwater/estuarine	70.66	172.7	198.76	392.49
	122	Marine	101.93	220.07	262.34	320.07
	172	All	113.04	227.03	307.65	379.97
Ages 16 to 17	28	Freshwater/estuarine	100.11	203.14	242.13	500.59
	37	Marine	125.51	280.88	352.54	529.78
	52	All	135.54	242.38	356.9	644.93

Source: EPA 2002.

The input seafood consumption rate in the RESRAD (onsite) and RESRAD-OFFSITE codes is the yearly total consumption rate, which does not take into account the difference between the contaminated and uncontaminated portion. It is assumed that if a surface water body (a pond) is located on a site, it will provide 50% of the consumed seafood. If a user would like to use a different fraction, then the fraction of aquatic food from the site (an input parameter) needs to be modified so that the calculated dose accounts for the correct contribution of contaminated seafood to the consumption rate.

7.4.3 Data Input Requirements

In the RESRAD (onsite) and RESRAD-OFFSITE code, the seafood consumption rate should be entered in units of kilograms per year (kg/yr). The current default value for the consumption rate is 5.4 kg/yr for fish and 0.9 kg/yr for other seafood.

TABLE 7.4.6 Types of Seafood Consumed in the United States (Mean, g/day)

	Uncooked		Cooked	
Species	>18 years	All	>18 years	All
Tuna	4.18×10^{0}	3.62×10^{0}	3.05×10^{0}	2.63×10^{0}
Shrimp	2.65×10^{0}	2.21×10^{0}	1.95×10^{0}	1.63×10^{0}
Salmon (marine)	1.78×10^{0}	1.39×10^{0}	1.30×10^{0}	1.02×10^{0}
Cod	1.66×10^{0}	1.48×10^{0}	1.27×10^{0}	1.13×10^{0}
Clam (marine)	8.70×10^{-1}	6.71×10^{-1}	1.28×10^{0}	1.00×10^{0}
Flounder	6.99×10^{-1}	5.83×10^{-1}	5.50×10^{-1}	4.58×10^{-1}
Catfish (estuarine)	5.75×10^{-1}	4.89×10^{-1}	3.99×10^{-1}	3.41×10^{-1}
Catfish (freshwater)	5.75×10^{-1}	4.89×10^{-1}	3.99×10^{-1}	3.41×10^{-1}
Porgy	4.95×10^{-1}	4.01×10^{-1}	3.37×10^{-1}	2.73×10^{-1}
Fish (nonspecified as to type)	4.76×10^{-1}	6.06×10^{-1}	1.93×10^{-1}	2.30×10^{-1}
Flatfish (estuarine)	4.04×10^{-1}	3.34×10^{-1}	3.38×10^{-1}	2.79×10^{-1}
Haddock	3.74×10^{-1}	3.25×10^{-1}	2.92×10^{-1}	2.54×10^{-1}
Crab (marine)	3.40×10^{-1}	2.88×10^{-1}	2.39×10^{-1}	2.04×10^{-1}
Pollock	3.32×10^{-1}	3.29×10^{-1}	2.85×10^{-1}	2.77×10^{-1}
Whiting	3.06×10^{-1}	2.57×10^{-1}	2.39×10^{-1}	2.01×10^{-1}
Crab (estuarine)	3.00×10^{-1}	2.54×10^{-1}	2.11×10^{-1}	1.80×10^{-1}
Lobster	2.59×10^{-1}	2.13×10^{-1}	1.92×10^{-1}	1.57×10^{-1}
Trout	2.41×10^{-1}	1.99×10^{-1}	1.92×10^{-1}	1.58×10^{-1}
Scallop (marine)	2.37×10^{-1}	1.90×10^{-1}	1.86×10^{-1}	1.48×10^{-1}
Perch (estuarine)	2.13×10^{-1}	1.81×10^{-1}	1.51×10^{-1}	1.29×10^{-1}
Perch (freshwater)	2.13×10^{-1}	1.81×10^{-1}	1.51×10^{-1}	1.29×10^{-1}
Squid	2.09×10^{-1}	1.54×10^{-1}	1.64×10^{-1}	1.21×10^{-1}
Carp	1.82×10^{-1}	1.34×10^{-1}	1.3×10^{-1}	9.58×10^{-2}
Herring	1.79×10^{-1}	1.33×10^{-1}	1.27×10^{-1}	9.41×10^{-2}
Oyster	1.74×10^{-1}	1.40×10^{-1}	1.43×10^{-1}	1.16×10^{-1}
Croaker	1.68×10^{-1}	1.37×10^{-1}	1.09×10^{-1}	8.80×10^{-2}
Ocean Perch	1.57×10^{-1}	1.41×10^{-1}	1.24×10^{-1}	1.11×10^{-1}
Trout, mixed sp. (estuarine)	1.46×10^{-1}	1.19×10^{-1}	1.05×10^{-1}	8.58×10^{-2}
Trout, mixed sp. (freshwater)	1.46×10^{-1}	1.19×10^{-1}	1.05×10^{-1}	8.58×10^{-2}
Mackerel	1.46×10^{-1}	1.15×10^{-1}	1.12×10^{-1}	8.78×10^{-2}
Sardine	1.44×10^{-1}	1.06×10^{-1}	1.04×10^{-1}	7.64×10^{-2}
Swordfish	1.26×10^{-1}	1.02×10^{-1}	9.63×10^{-2}	7.79×10^{-2}
Sea bass	1.25×10^{-1}	1.29×10^{-1}	9.49×10^{-2}	9.77×10^{-2}
Pompano	1.12×10^{-1}	9.91×10^{-2}	8.11×10^{-2}	7.13×10^{-2}
Mussels	9.97×10^{-2}	7.43×10^{-2}	6.94×10^{-2}	5.18×10^{-2}
Octopus	8.82×10^{-2}	6.43×10^{-2}	6.83×10^{-2}	4.98×10^{-2}
Salmon (estuarine)	8.82×10^{-2}	6.90×10^{-2}	6.46×10^{-2}	5.06×10^{-2}
Flatfish (marine)	7.56×10^{-2}	6.25×10^{-2}	6.33×10^{-2}	5.22×10^{-2}
Anchovy	5.54×10^{-2}	4.33×10^{-2}	3.81×10^{-2}	2.98×10^{-2}
Rockfish	5.16×10^{-2}	4.45×10^{-2}	3.99×10^{-2}	3.44×10^{-2}
Mullet	4.3×10^{-2}	3.62×10^{-2}	2.93×10^{-2}	2.48×10^{-2}
Halibut	4.22×10^{-2}	3.23×10^{-2}	3.47×10^{-2}	2.65×10^{-2}
Pike	3.83×10^{-2}	3.26×10^{-2}	3.52×10^{-2}	2.96×10^{-2}
Snapper	3.62×10^{-2}	2.74×10^{-2}	3.19×10^{-2}	2.41×10^{-2}
Clam (Estuarine)	2.33×10^{-2}	1.8×10^{-2}	3.43×10^{-2}	2.69×10^{-2}

TABLE 7.4.6 (Cont.)

	Uncooked		Cooked	
Species	>18 years	All	>18 years	All
Whitefish (freshwater)	1.25×10^{-2}	9.95×10^{-3}	1.24×10^{-2}	9.88×10^{-3}
Whitefish (marine)	1.25×10^{-2}	9.95×10^{-3}	1.24×10^{-2}	9.88×10^{-3}
Crayfish	1.02×10^{-2}	7.46×10^{-3}	7.89×10^{-3}	5.75×10^{-3}
Smelts	8.38×10^{-3}	-	-	-
Smelts (estuarine)		6.11×10^{-3}	5.7×10^{-3}	4.15×10^{-3}
Smelts (marine)		6.11×10^{-3}	5.7×10^{-3}	4.15×10^{-3}
Shark	5.81×10^{-3}	4.24×10^{-3}	4.59×10^{-3}	3.55×10^{-3}
Eel	4.44×10^{-3}	3.24×10^{-3}	3.5×10^{-3}	2.55×10^{-3}
Seafood	3.94×10^{-3}	3.26×10^{-3}	2.45×10^{-3}	2.03×10^{-3}
Conch	2.84×10^{-3}	2.07×10^{-3}	2.13×10^{-3}	1.55×10^{-3}
Snails (freshwater)	2.06×10^{-3}	2.49×10^{-3}	1.64×10^{-3}	1.98×10^{-3}
Snails (marine)	2.06×10^{-3}	2.49×10^{-3}	1.64×10^{-3}	1.98×10^{-3}
Cisco	1.70×10^{-3}	2.34×10^{-3}	1.17×10^{-3}	1.60×10^{-3}
Scallop (estuarine)	1.60×10^{-3}	1.28×10^{-3}	1.25×10^{-3}	1.00×10^{-3}
Roe	1.40×10^{-3}	1.02×10^{-3}	1.11×10^{-3}	8.10×10^{-4}
Salmon (freshwater)	9.3×10^{-4}	7.30×10^{-4}	6.80×10^{-4}	5.30×10^{-4}
Smelts, rainbow (estuarine)	7.20×10^{-4}	5.20×10^{-4}	5.10×10^{-4}	3.70×10^{-4}
Smelts, rainbow	7.20×10^{-4}	5.20×10^{-4}	5.10×10^{-4}	3.70×10^{-4}
Sturgeon (estuarine)	1.70×10^{-4}	1.30×10^{-4}	1.70×10^{-4}	1.30×10^{-4}
Sturgeon (freshwater)	1.70×10^{-4}	1.30×10^{-4}	1.70×10^{-4}	1.30×10^{-4}
Total seafood (g/day)	1.99×10^{1}	1.69×10^{1}	1.53×10^{1}	1.28×10^{1}
Total seafood (kg/yr)	7.27×10^{0}	6.17×10^{0}	5.58×10^{0}	4.69×10^{0}

Source: EPA 2002.

7.5 FRUIT, VEGETABLE, AND GRAIN CONSUMPTION RATES

7.5.1 Definition

Consumption rate is the fresh weight of the fruit, vegetables, and grain consumed each year by a single individual. The consumption rate includes both contaminated and uncontaminated fruit, vegetables, and grain; the contaminated fraction is specified as a separate input.

7.5.2 Summary of Literature Review

According to the 1986–1987 survey by the National Gardening Association (1987), 38% or a total of 34 million U.S. households participated in vegetable gardening in 1986. The size of the home vegetable garden, however, has decreased from 600 ft^2 in 1982 to 325 ft^2 in 1986 (National Gardening Association 1987). The distribution of home gardens varies geographically,

TABLE 7.4.7 Average Per-capita Consumption Rates (lb/yr) in the United States for the Top 10 Seafood Species

Species	Average 2002–2010	2012	2011	2010	2009	2008	2007	2006	2005	2004
Shrimp	4.08	3.8	4.2	4	4.1	4.1	4.1	4.4	4.1	4.2
Canned tuna	2.8	2.4	2.6	2.7	2.5	2.8	2.7	2.9	3.1	3.3
Salmon	2.12	2.02	1.952	1.999	2.04	1.84	2.364	2.026	2.43	2.154
Polluck	1.48	1.167	1.312	1.192	1.454	1.34	1.73	1.639	1.468	1.277
Catfish	0.97	0.5	0.559	0.8	0.849	0.92	0.876	0.969	1.025	1.091
Tilapia	0.93	1.476	1.287	1.45	1.208	1.19	1.142	0.996	0.848	0.696
Crab	0.62	0.523	0.518	0.573	0.594	0.61	0.679	0.664	0.643	0.626
Cod	0.49	0.521	0.501	0.463	0.419	0.44	0.465	0.505	0.572	0.603
Clam	0.45	0.347	0.331	0.341	0.413	0.42	0.449	0.44	0.435	0.471
Pangasius	0.38	0.726	0.628	0.405	0.356	NA[a]	NA	NA	NA	NA
Flatfish	NA	NA	NA	NA	NA	0.43	0.305	NA	0.366	0.332
Scallops	NA	NA	NA	NA	NA	NA	NA	0.305	NA	NA
Total all species, lb/yr	16.14	14.6	15	15.8	15.8	16	16.3	16.5	16.2	16.6
Total all species, kg/yr	7.32	6.62	6.80	7.17	7.17	7.26	7.39	7.48	7.35	7.53

Note: to convert to kg, multiply the values by 0.453592.

[a] NA = value not available.

Source: National Marine Fisheries Service (NOAA 2014)

TABLE 7.4.8 EPA-Recommended Per-capita and Consumer-Only Mean Uncooked Seafood Intake Values, g/kg-day

		Per Capita			Consumer Only		
Age	Sample Size	Finfish	Shellfish	Total	Finfish	Shellfish	Total
All	16,783	0.16	0.06	0.22	0.73	0.57	0.78
Birth to 1 year	865	0.03	0	0.04	1.3	0.42	1.2
1 to 2 years	1,052	0.22	0.04	0.26	1.6	0.94	1.5
3 to 5 years	978	0.19	0.05	0.24	1.3	1	1.3
6 to 12 years	2.256	0.16	0.05	0.21	1.1	0.72	0.99
13 to 19 years	3,450	0.1	0.03	0.13	0.66	0.61	0.69
20 to 49 years	4,289	0.15	0.08	0.23	0.65	0.63	0.76
50+ years	3,893	0.2	0.05	0.25	0.68	0.41	0.71

Source: EPA (2011).

with a large percentage located in the Midwest and South, and more in rural areas than in cities and suburbs. Therefore, homegrown fruits and vegetables make up a larger portion of the average consumption rate in rural areas than in cities or suburbs.

The EPA has made recommendations on the consumption rates of homegrown fruits and vegetables on the basis of two sources: *Foods Commonly Eaten by Individuals: Amount Per Day and Per Eating Occasion* (Pao et al. 1982) and *Food Consumption: Households in the United States, Seasons and Year 1977–1978* (USDA 1983). The first source used data collected by the USDA in 1977–1978 from home interviews of 37,874 respondents who were asked to recall food consumed one day before the interview, the day of the interview, and the day after the interview, to calculate percentiles of total fruit and vegetable consumption by the U.S. population. The consumption rate of homegrown fruits and vegetables can be calculated by subtracting the data for the “bought” category for all foods from the data for the “all” category in the USDA food consumption survey. Homegrown dark green vegetables make up approximately one-third of the dark green vegetables consumed. This category includes mustard greens, kale, kohlrabi, and broccoli. Consumption of homegrown corn, cucumbers, green beans, and tomatoes makes a significant contribution to total consumption. The proportion of homegrown fruits consumed is highest for strawberries, peaches, and pears, and lowest for citrus fruits.

According to the EPA (1990a), the average consumption rate of vegetables per person is 200 g/d (73 kg/yr); homegrown products account for 25% of the total consumption rate, that is, 50 g/d (18 kg/yr). Total average daily fruit intake is 140 g/d (51 kg/yr) per individual. The total homegrown fruit consumption rate is 28 g/d (10 kg/yr), which is 20% of the total intake rate. For a reasonable worst case, it is suggested that 40% of the total intake be allocated to homegrown vegetable consumption and 30% of the total intake be allocated to homegrown fruit consumption. Table 7.5.1 summarizes the EPA’s recommendations.

TABLE 7.5.1 Vegetable and Fruit Intake Rates (All Ages)

	Average Intake				Worst-Case Intake, Homegrown	
	Total		Homegrown			
	(g/d)	(kg/yr)	(g/d)	(kg/yr)	(g/d)	(kg/yr)
Vegetables	200	73	50	18	80	29
Fruit	140	51	28	10	42	15
Total	340	124	78	28	122	44

Some nationwide surveys, such as CSFII and NHANES, collect data on dietary intake in the United States population. The survey data were used by EPA (2011) to estimate the overall per capita intake rate of different food types. Table 7.5.2 lists the estimated mean from 1994–1996 and 1998 CSFII surveys of vegetable, fruit, and grain intake in the United States in children of different age groups (EPA 2011). The average intakes of various vegetables and fruits in children of different age groups are listed in Tables 7.5.3 and 7.5.4, respectively.

Per-capita intake of fresh fruits, vegetables, and grains from 1994–1995 CSFII data was 172 g/day (62.8 kg/yr), 187 g/day (68.3 kg/yr), and 302 g/day (110.2 kg/yr), respectively (EPA 2011). This gives a total average per-capita intake of 241.3 kg/yr for fresh fruits, vegetables, and grains combined. The average per-capita intake from the NHANES 2003−2006 surveys for all ages combined was estimated to be 2.9 g/kg-day for vegetables, 1.6 g/kg-day for fruits, and 2.6 g/kg-day for grains. EPA (2011) provided intake rates in units of food consumed per kilogram of body weight per day from the NHANES surveys. Converting the intake rates into units of kg/yr by multiplying by a single average body weight is inappropriate, because intake rates were indexed to the reported body weights of the survey respondents. Average vegetable, fruit, and grain intakes varied from 2.3 to 6.7 g/kg-day, 1.7 to 6.4 g/kg-day, and 0.9 to 7.8 g/kg-day, respectively, in different age groups (EPA 2011). Table 7.5.5 lists the average intake rates on a per-kilogram-body-weight basis for each age group.

In NRC Regulatory Guide 1.109 (NRC 1977), different total consumption amounts of fruits, vegetables, and grains are suggested for different age groups. The average individual consumption for a child is 200 kg/yr, for a teenager it is 240 kg/yr, and for an adult it is 190 kg/yr. Suggested values for the maximally exposed individual in a worst-case scenario are 520, 630, and 520 kg/yr for a child, teenager, and adult, respectively. The total consumption for the maximum-exposure case consists of 22% for fruit consumption, 54% for vegetable consumption, and 24% for grain consumption. In its decommissioning guidance (Schmidt et al. 2006), the NRC allows use of the mean of the parameter distribution value for an average member of the critical group (Beyeler et al. 1999) for food consumption rates. The mean consumption rate for an average member of the screening group for homegrown fruits, vegetables, and grains is 53, 45, and 14 kg/yr, respectively (Beyeler et al. 1999).

TABLE 7.5.2 Mean Vegetable, Fruit, and Grain Intake (g/day as Consumed) in Children of Different Age Groups

Age Group	Vegetables	Fruits	Grains
Under 1	57	131	56
1 to 2	91	271	206
3 to 5	97	239	264
6 to 11	115 (male), 116 (female)	183 (male), 169 (female)	318 (male), 280 (female)
12 to 19	176 (male), 145 (female)	174 (male), 157 (female)	406 (male), 306 (female)

Source: EPA (2011).

To run the RESRAD (onsite) and RESRAD-OFFSITE codes, a yearly consumption rate for fruit, vegetables, and grain is needed that does not differentiate the contaminated fraction from the uncontaminated fraction. An area factor will automatically be calculated and used to adjust the consumption rate. It is assumed that if an area is greater than 1,000 m^2, then 50% of the plant food consumed is obtained from the site; if the area is smaller than 1,000 m^2, then the fraction of the contaminated product is the ratio of the contaminated area to an area of 2,000 m^2. The upper bound in the default adjustment for the fraction of contaminated products is 50%. If this value differs from that obtained from site-specific data, the user should adjust the yearly consumption rate and the fraction of contaminated products so that an accurate consumption rate of the contaminated product is used to derive the total dose.

7.5.3 Data Input Requirements

In the RESRAD (onsite) and RESRAD-OFFSITE codes, the consumption rate should be entered in units of kilograms per year (kg/yr). The default value for the consumption rate is 160 kg/yr for fruit, vegetables, and grains. For the probabilistic analysis, use distributions developed for the fruit, vegetable, and grain consumption rates in NUREG/CR-6697 (Yu et al. 2000).

7.6 LEAFY VEGETABLE CONSUMPTION RATE

7.6.1 Definition

The leafy vegetable consumption rate is a dietary factor for human food consumption that includes consumption of vegetables such as spinach and lettuce.

TABLE 7.5.3 Average Intake (g/day) of Different Vegetables in Children of Different Age Groups

Age Group	White Potatoes	Dark Green Vegetables	Deep Yellow Vegetables	Tomatoes	Lettuce	Green Beans	Corn, Peas, Lima Beans	Other Vegetables
Under 1	9	2	19	1	<0.5	6	5	16
1 to 2	34	5	5	13	2	5	11	16
3 to 5	38	5	5	12	3	5	11	17
6 to 11	50 (male), 46 (female)	5 (male), 5 (female)	5 (male), 4 (female)	16 (male), 15 (female)	5 (male), 7 (female)	5 (male), 5 (female)	11 (male), 12 (female)	18 (male), 20 (female)
12 to 19	85 (male), 61 (female)	6 (male), 9 (female)	6 (male), 4 (female)	28 (male), 18 (female)	12 (male), 12 (female)	3 (male), 4 (female)	10 (male), 8 (female)	25 (male), 28 (female)

Source: EPA (2011).

TABLE 7.5.4 Average Intake (g/day) of Different Fruits in Children of Different Age Groups

Age Group	Citrus Fruits and Juices	Dried Fruits	Apples	Bananas	Melons and Berries	Other Fruits	Noncitrus Juices
Under 1	4	<0.5	14	10	1	39	61
1 to 2	56	2	24	22	9	24	132
3 to 5	59	1	30	16	13	23	93
6 to 11	67 (male), 64 (female)	<0.5 (male), <0.5 (female)	28 (male), 21 (female)	11 (male), 8 (female)	16 (male), 8 (female)	19 (male), 23 (female)	40 (male), 42 (female)
12 to 19	72 (male), 70 (female)	1 (male), <0.5 (female)	13 (male), 13 (female)	8 (male), 5 (female)	11 (male), 15 (female)	10 (male), 14 (female)	29 (male), 35 (female)

Source: EPA (2011).

TABLE 7.5.5 Average Per-capita Intake of Total Fruits, Vegetables, and Grains (g/kg-day as Consumed)

Age Group (years)	Total Fruits	Total Vegetables	Total Grains
Birth to 1 year	6.2	5	3.1
1 to 2 years	7.8	6.7	6.4
3 to 5 years	4.6	5.4	6.2
6 to 12 years	2.3	3.7	4.4
13 to 19 years	0.9	2.3	2.4
20 to 49 years	0.9	2.5	2.2
50 years and older	1.4	2.6	1.7
Whole population	1.6	2.9	2.6

Source: EPA (2011).

7.6.2 Summary of Literature Review

On the basis of recommended values for the maximally exposed individual in NRC Regulatory Guide 1.109 (NRC 1977), the consumption rates of leafy vegetables for children, teenagers, and adults, respectively, are 26, 42, and 64 kg/yr. Average consumption rates used by the NRC to perform environmental dose analyses for releases of radioactive effluents from nuclear power plants into the atmosphere (Strenge 1987) are 10, 20, and 30 kg/yr for children, teenagers, and adults, respectively. In its decommissioning guidance (Schmidt et al. 2006), the NRC allows use of the mean of the parameter distribution value for an average member of the critical group (Beyeler et al. 1999) for food consumption rates. The mean consumption rate for an average member of the screening group for homegrown leafy vegetables is 21 kg/yr (Beyeler et al. 1999).

Data from the 2003–2006 National Health and Nutrition Examination Survey (NHANES) was used by the U.S. EPA to generate per-capita intake rates for individual fruits and vegetables (EPA 2011). EPA (2011) provided intake rates in units of food consumed per kilogram of body weight per day from NHANES survey. Converting the intake rates into units of kg/yr by multiplying by a single average body weight is inappropriate because intake rates were indexed to the reported body weights of the survey respondents. The average per-capita leafy vegetable intake for all ages combined was estimated to be 0.54 g/kg-day. Table 7.6.1 lists the average intake rates on a per-kilogram-body-weight basis for each age group. The average leafy vegetable consumption varied from 0.7 (birth to 1 year) to 17.5 kg/yr (50 years and older) in different age groups (EPA 2011). For adults this results in yearly average leafy vegetable consumption rate of about 16.5 kg/yr.

The RESRAD (onsite) and RESRAD-OFFSITE code input for leafy vegetable consumption rate does not differentiate the contaminated fraction from the uncontaminated fraction. As for the fruit, vegetable, and grain consumption rates covered in Section 7.5, a default adjustment is automatically performed, via the contaminated area, within the codes. If this value

TABLE 7.6.1 Average Per-capita Intake of Leafy Vegetables (g/kg-d as Consumed)

Age Group	Leafy Vegetables
Whole population	0.54
Birth to 1 year	0.22
1 to 2 year	0.71
3 to 5 year	0.61
6 to 12 year	0.43
13 to 19 years	0.35
20 to 49 years	0.55
50 years and older	0.6

Source: EPA (2011).

is different from that obtained from site-specific data, then the input consumption rate and the contaminated fraction need to be modified so that the RESRAD (onsite) or RESRAD-OFFSITE code calculates the correct dose for the contaminated product.

7.6.3 Data Input Requirements

The default value used in RESRAD (onsite) and RESRAD-OFFSITE for the leafy vegetable consumption rate is 14 kg/yr.

7.7 MEAT AND POULTRY CONSUMPTION RATE

7.7.1 Definition

The meat and poultry consumption rate is the fresh weight of the meat and poultry that is consumed each year by a single individual. The consumption rate includes both contaminated and uncontaminated meat and poultry; the contaminated fraction is specified as a separate input.

7.7.2 Summary of Literature Review

The USDA conducted a national food consumption survey in 1977–1978 (USDA 1983). The average consumption rates for beef and dairy products, as adopted by the EPA (1984b,c),are based on the results of this survey.

According to USDA studies, 44% of annual consumption is homegrown beef. This finding is based on a survey of 900 rural farm households (USDA 1966). Because the total amount of beef consumed averages approximately 100 g/d (36.5 kg/yr), the average consumption of homegrown beef is about 44 g/d (EPA 1990a), which corresponds to 16 kg/yr.

For a reasonable worst-case value, the EPA (1990a) has suggested that a consumption rate of 75 g/d (27 kg/yr) be used for homegrown beef in risk assessments until better data are available.

The average consumption rate of 36.5 kg/yr, as recommended by the EPA, accounts for beef only. The total consumption rate for meat and poultry should be much higher. According to NRC Regulatory Guide 1.109 (NRC 1977), the recommended average values for consumption of meat and poultry are 37 kg/yr for children, 59 kg/yr for teenagers, and 95 kg/yr for adults. Recommended values for use in a maximally exposed case are 41 kg/yr for children, 65 kg/yr for teenagers, and 110 kg/yr for adults. In its decommissioning guidance (Schmidt et al. 2006), the NRC allows use of the mean of the parameter distribution value for an average member of the critical group (Beyeler et al. 1999) for food consumption rates. The mean consumption rate for an average member of the screening group for homegrown beef and poultry is 40 and 25 kg/yr, respectively (Beyeler et al. 1999).

Gilbert et al. (1983) used values of 79 lb/yr (36 kg/yr) for meat, 20 lb/yr (9 kg/yr) for poultry, and 15 lb/yr (7 kg/yr) for egg consumption, with a total value of 114 lb/yr (52 kg/yr). The consumption rate used for meat is about the same as that recommended by the EPA (1990a). If the same percentage used for homegrown beef can be applied to consumption of poultry and eggs, then the average consumption of homegrown meat, poultry, and eggs would be 23 kg/yr; for a reasonable worst-case scenario, the value would be 39 kg/yr on the basis of the data of Gilbert et al. (1983).

Some of the nationwide surveys, such as CSFII or NHANES, collect data on dietary intake in the United States population. The survey data was used by EPA (2011) to estimate the overall per capita consumption rates of different food types. The per capita estimated means from the 1994 and 1995 CSFII surveys of meat intake in the United States were 195 g/d (71 kg/yr) and 202 g/d (74 kg/yr), respectively (EPA 2011). The average per-capita meat consumption from the NHANES 2003−2006 surveys for all ages combined was estimated to be 2 g/kg-day. EPA (2011) provided intake rates in units of grams of food consumed per kilogram of body weight per day based on the NHANES surveys. Converting the intake rates into units of kg/yr by multiplying by a single average body weight is inappropriate, because intake rates were indexed to the reported body weights of the survey respondents. The total average meat intake included 0.77 g/kg-day for beef, 0.39 g/kg-day for pork, and 0.77 g/kg-day for poultry (EPA 2011). The average meat intake varied from 1.2 to 4 g/kg-day in different age groups (EPA 2011). Table 7.7.1 lists the intake rates on a per-kilogram-body-weight basis for each age group.

The USDA (Putnam et al. 1999) has provided food intakes for a variety of foods (e.g., fruits and vegetables, red meat, poultry, eggs, dairy products, flour and cereals). The estimates of human intake of foods are derived by subtracting exports, farm and industrial uses, and end-of-year stocks from total supply (production, beginning stock, and imports). The

TABLE 7.7.1 Per-capita Intake of Total Meat (g/kg-day as Consumed)

Age Group (years)	Average	50th Percentile	90th Percentile	95th Percentile
Birth to 1 year	1.2	0	3.6	5.4
1 to 2 years	4	3.4	8	10
3 to 5 years	3.9	3.3	7.6	8.5
6 to 12 years	2.8	2.5	5.2	6.4
13 to 19 years	2	1.7	3.8	4.7
20 to 49 years	1.8	1.6	3.4	4.1
50 years and older	1.4	1.3	2.6	3.1
Whole population	2	1.6	3.8	4.8

Source: EPA (2011).

estimates do not consider any spoilage or waste and therefore would be an upper bound on human intakes. Table 7.7.2 lists per-capita intake rates for red meat and poultry for the years 1972–1997 from this USDA report. The average per-capita intake rates for red meat and poultry are 55.4 and 21.3 kg/yr, respectively. These rates result in a total average per-capita meat intake of 76.7 kg/yr.

In the RESRAD (onsite) and RESRAD-OFFSITE codes, it is assumed that all of the consumed meat is contaminated if the area of the contaminated zone is greater than or equal to 20,000 m^2. If the area is less than 20,000 m^2, then the fraction of the contaminated product is the ratio of the contaminated area to an area of 20,000 m^2. If site-specific data differ from the default values, the input data for consumption rate and the contaminated fraction may need to be adjusted so that the correct dose from the contaminated meat product is obtained.

7.7.3 Data Input Requirements

In the RESRAD (onsite) and RESRAD-OFFSITE codes, the consumption rate for meat and poultry should be entered in units of kilograms per year (kg/yr). The current default value for the consumption rate is 63 kg/yr. Recent studies have reported somewhat higher total meat intake rates (e.g., 71 kg/yr [1994 CSFII survey], 74 kg/yr [1995 CSFII survey], and 76.7 kg/yr [Putnam et al. 1999]).

TABLE 7.7.2 Per-capita Intake Rates for Red Meat and Poultry

Year	Red Meat (lb/yr)	Poultry (lb/yr)	Red Meat (kg/yr)	Poultry (kg/yr)
1970	131.7	33.8	59.7	15.3
1971	135.5	34.0	61.5	15.4
1972	131.8	35.4	59.8	16.1
1973	121.8	33.7	55.2	15.3
1974	130.4	33.8	59.1	15.3
1975	125.8	32.9	57.1	14.9
1976	133.0	35.5	60.3	16.1
1977	132.3	35.9	60.0	16.3
1978	127.5	37.3	57.8	16.9
1979	124.4	40.1	56.4	18.2
1980	126.4	40.8	57.3	18.5
1981	125.1	42.1	56.7	19.1
1982	119.8	42.2	54.3	19.1
1983	123.9	42.7	56.2	19.4
1984	123.7	44.0	56.1	20.0
1985	124.9	45.5	56.6	20.6
1986	122.2	47.4	55.4	21.5
1987	117.4	51.0	53.3	23.1
1988	119.5	51.9	54.2	23.5
1989	115.9	53.9	52.6	24.4
1990	112.3	58.3	50.9	26.4
1991	111.9	58.3	50.8	26.4
1992	114.1	60.8	51.8	27.6
1993	112.1	62.5	50.8	28.4
1994	114.7	63.3	52.0	28.7
1995	115.1	62.9	52.2	28.5
1996	112.8	64.4	51.2	29.2
1997	111.0	64.8	50.3	29.4
Average	122.0	47.0	55.4	21.3

Source: Putnam et al. (1999).

7.8 MILK CONSUMPTION RATE

7.8.1 Definition

Milk consumption rate is the amount of milk that is consumed each year by a single individual. The consumption rate includes both contaminated and uncontaminated milk; the contaminated fraction is specified as a separate input.

7.8.2 Summary of Literature Review

According to the EPA (1984b) and Fries (1986), the milk (fresh milk only) consumption rate can range from 254 g/d to 1,000 g/d per person, with an average rate of 305 g/d (i.e., 110 L/yr). According to the USDA (USDA 1966; Putnam 1999), 40% of the dairy products consumed in a typical farm household are from milk cows on the farm. Applying this same percentage to a typical farm scenario, 44 L/yr of the fresh milk consumed is actually from cows owned by the farmer. On the basis of EPA (1990a) suggestions for a worst-case scenario, if 75% of the fresh milk consumed is assumed to be from milk cows on the farm, the average consumption rate of fresh milk is 83 L/yr per person for a farm scenario.

In NRC Regulatory Guide 1.109 (NRC 1977), milk consumption rates for different age groups are reported. The average rates for children, teenagers, and adults are 170, 200, and 110 L/yr, respectively. Recommended values for the maximally exposed individual are 330, 330, 400, and 310 L/yr for infants, children, teenagers, and adults, respectively. In its decommissioning guidance (Schmidt et al. 2006), the NRC allows use of the mean of the parameter distribution value for an average member of the critical group (Beyeler et al. 1999) for food consumption rates. The mean consumption rate for an average member of the screening group for homegrown milk is 233 L/yr (Beyeler et al. 1999).

Some of the nationwide surveys, such as CSFII or NHANES, collect data on dietary intake in the United States population. The survey data were used by EPA (2011) to estimate the overall per capita intake rate of different food types. The per capita estimated means from the 1994 and 1995 CSFII surveys of milk intake in the United States were 229 g/d (84 L/yr) and 236 g/d (86 L/yr), respectively (EPA 2011). The estimated mean values for milk and milk products using1994–1996 and 1998 CSFII survey data were 477 g/d (174.1 L/yr) for children aged ≤5 years, 453 g/d (165.3 L/yr) for children aged ≤9 years, and 405 g/d (147.8 L/yr) for children aged ≤19 year (EPA 2011).

The average per-capita dairy consumption from the NHANES 2003−2006s survey for all ages combined was estimated to be 6.6 g/kg-day. EPA (2011) provided intake rates in units of food consumed per kilogram of body weight per day from the NHANES survey. Converting the intake rates into units of kg/yr by multiplying by a single average body weight is inappropriate, because intake rates were indexed to the reported body weights of the survey respondents. Table 7.8.1 lists the dairy intake rates on a per-kilogram-body-weight basis for each age group; the average intake varied from 3.3 to 43.2 g/kg-day in different age groups (EPA 2011).

USDA (Putnam et al. 1999) provided food intakes for a variety of foods (e.g., fruits and vegetables, red meat, poultry, eggs, dairy products, flour and cereals). The estimates of human intake of foods are derived by subtracting exports, farm and industrial uses, and end-of-year stocks from total supply (production, beginning stock, and imports). The estimates do not consider any spoilage or waste and therefore would be an upper bound on human intakes. Table 7.8.2 lists per-capita intake rates for eggs and dairy products for the years 1972–1997 from this USDA report. The average per-capita intake rates for eggs, dairy products, and milk are 15.2, 255.8 kg/yr, and 103 L/yr, respectively.

TABLE 7.8.1 Per-capita Intake of Total Dairy Products (g/kg-day as Consumed)

Age Group (years)	Average	50th Percentile	90th Percentile	95th Percentile
Birth to 1 year	10.1	6.4	19.6	43.2
1 to 2 years	43.2	39.1	84.1	94.7
3 to 5 years	24	20.7	41.9	51.1
6 to 12 years	12.9	10.8	26	31.8
13 to 19 years	5.5	4	12.3	16.4
20 to 49 years	3.5	2.4	8.1	10.3
50 years and older	3.3	2.3	7.3	9.6
Whole population	6.6	3.2	15.4	25

Source: EPA (2011).

TABLE 7.8.2 Per-capita Intake Rates for Eggs, Dairy Products, and Milk

Year	Eggs (lb/yr)	Dairy Products (lb/yr)	Milk (gallon/yr)	Eggs (kg/yr)	Dairy Products (kg/yr)	Milk (L/yr)
1970	39.5	563.8	31.3	17.9	255.7	118.5
1971	39.7	557.9	31.3	18.0	253.1	118.5
1972	38.8	559.6	31	17.6	253.8	117.3
1973	37.0	554.8	30.5	16.8	251.7	115.5
1974	36.3	535.0	29.5	16.5	242.7	111.7
1975	35.4	539.1	29.5	16.1	244.5	111.7
1976	34.8	539.7	29.3	15.8	244.8	110.9
1977	34.3	540.2	29	15.6	245.0	109.8
1978	34.9	544.3	28.6	15.8	246.9	108.3
1979	35.5	548.2	28.2	16.1	248.7	106.7
1980	34.8	543.2	27.6	15.8	246.4	104.5
1981	34.0	540.6	27.1	15.4	245.2	102.6
1982	33.9	554.6	26.4	15.4	251.6	99.9
1983	33.5	572.9	26.3	15.2	259.9	99.6
1984	33.5	581.9	26.4	15.2	263.9	99.9
1985	32.8	593.7	26.7	14.9	269.3	101.1
1986	32.6	591.5	26.5	14.8	268.3	100.3
1987	32.7	601.2	26.3	14.8	272.7	99.6
1988	31.8	562.5	25.8	14.4	255.2	97.7
1989	30.5	563.8	26	13.8	255.7	98.4
1990	30.2	568.4	25.7	13.7	257.8	97.3
1991	30.1	565.6	25.6	13.7	256.6	96.9
1992	30.3	565.9	25.3	13.7	256.7	95.8
1993	30.4	574.1	24.8	13.8	260.4	93.9
1994	30.6	586.0	24.8	13.9	265.8	93.9
1995	30.3	584.4	24.3	13.7	265.1	92.0
1996	30.6	575.5	24.4	13.9	261.0	92.4
1997	30.8	579.8	24	14.0	263.0	90.8
Average	33.6	563.9	27.2	15.2	255.8	103.0

Source: Putnam et al. (1999).

The RESRAD and RESRAD-OFFSITE codes assume that all of the consumed milk is contaminated if the area of the contaminated zone is greater than or equal to 20,000 m^2. If the area is less than 20,000 m^2, then the fraction of the contaminated product is the ratio of the contaminated area to an area of 20,000 m^2. Therefore, caution should be used in choosing the appropriate input data so that the correct site-specific dose level is obtained.

7.8.3 Data Input Requirements

In RESRAD and RESRAD-OFFSITE, the default milk consumption rate is set to 92 L/yr per person. The default milk consumption rate is comparable to the values in recent CSFII surveys. For the probabilistic analysis, use the distributions developed for milk consumption rate in NUREG/CR-6697 (Yu et al. 2000).

This page intentionally left blank.

8 SOURCE CHARACTERISTIC PARAMETERS

8.1 AREA OF CONTAMINATED ZONE

8.1.1 Definition

A contaminated zone is a compact area that contains the locations of soil samples with radionuclide concentrations clearly exceeding background levels. Background concentrations are determined from measurements in soil samples taken at several nearby off-site locations where contamination is highly unlikely. The concentration of a radionuclide is considered to clearly exceed background concentrations if it is greater than the mean background concentration plus twice the standard deviation of the background measurements. If the concentrations in the samples used for determining the background concentration are below the lower limit of detection (LLD) of the instrument used, the concentration of that radionuclide is considered to exceed background if it exceeds the LLD of the instrument. The sensitivity of the instrument used must comply with current standards for high-quality commercial instruments.

To justify the use of two or more contaminated zones, credible evidence must be provided on the basis of radiological survey data that the intervening area between any two contaminated zones is uncontaminated; otherwise, the contaminated zone should be characterized by a single compact area that contains the locations of all soil samples with above-background radionuclide concentrations.

8.1.2 Data Input Requirements

The actual area of the contaminated zone should be entered into the RESRAD (onsite) code. In the RESRAD-OFFSITE code, the area of the primary contamination is calculated from the X and Y dimensions of the primary contamination. The default values for the X and Y dimension in the RESRAD-OFFSITE code are 100 m. The area should be specified in units of square meters (m^2). A default value of 10,000 m^2 is used in both codes for the area of the contaminated zone.

8.2 THICKNESS OF CONTAMINATED ZONE

8.2.1 Definition

The thickness of the contaminated zone is the distance between the uppermost and lowermost soil samples that have radionuclide concentrations clearly above background. In determining whether the measured soil concentration is above the background level, a DOE-approved method based on a statistical analysis of site measurements in comparison to background measurements should be used (DOE 1991, Section 7). In case such an approach is not available, then as a default approach, a soil sample should be treated as clearly contaminated if the radionuclide concentration is greater than the average background radionuclide concentration plus twice the standard deviation of the background measurements.

8.2.2 Measurement Methodology

A DOE-approved statistical approach (DOE 1991, Section 7) should always be considered first when estimating averages, handling distribution analyses and estimating central tendency soil concentrations. The default approach provided below is a conservative method that may sometimes significantly overestimate the dose. To determine the thickness of a contaminated zone with an area greater than 100 m^2, the average contamination thickness of boreholes drilled to take soil samples is calculated over any integral subarea of 100 m^2. If one or more boreholes in the subarea have a contamination thickness exceeding the average thickness by a factor larger than three, then the average value is replaced by one-third the maximum contamination thickness. The thickness of the contaminated zone is then taken as the maximum average thickness calculated over a 100-m^2 subarea. For a contaminated zone with an area less than 100 m^2, the average contamination thickness of boreholes drilled to take soil samples over the contaminated zone is calculated. If. for any borehole in the subarea, the thickness is greater than three times the average value, then one-third of the maximum borehole thickness is taken as the representative value of the contaminated zone thickness. Illustrative examples for determining the thickness of the contaminated zone are shown in Figures 8.2.1 and 8.2.2.

Figure 8.2.1 shows an example for determining the thickness of a contaminated zone with an area greater than 100 $m^{2;}$ this example is for illustrative purposes only. In this example the contaminated zone area is 200 $m^{2;}$ therefore it is divided in two subareas of 100 m^2. One subarea has seven boreholes, and the other, six boreholes. The figure shows the measured contamination thickness in each borehole. The average thickness for subarea 1 is [(3 + 2 + 1 + 1 + 2 + 2 + 10) ÷ 7] 3 m. One borehole thickness, 10 m, is greater than the average thickness of 3 m; therefore the representative thickness of this subarea is (10 ÷ 3) 3.33 m. The average thickness for subarea 2 is [(2 + 2 + 3 + 3 + 2 + 3) ÷ 6] 2.5 m. None of the boreholes in this subarea exceed three times the average thickness; therefore the thickness for this subarea is 2.5 m. For determining the thickness of this contaminated zone, the highest average subarea thickness, 3.33 m, is chosen.

Figure 8.2.2 shows an example for determining the thickness of a contaminated zone with an area less than 100 m^2. This example is for illustrative purposes only. This area has ten boreholes. The figure shows the measured contamination thickness in each borehole. The average thickness for this area is [(3 + 2 + 2 + 1 + 3 + 3 + 2 + 10 + 2 + 2) ÷10] 3 m. One borehole thickness is10 m, which is greater than the average thickness of 3 m; therefore representative thickness for this area is (10 ÷ 3) 3.33 m; therefore the thickness for this contaminated zone is 3.33 m.

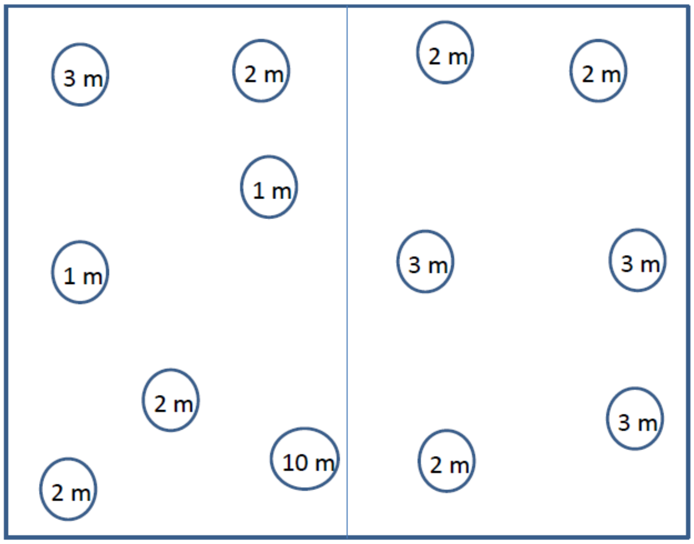


FIGURE 8.2.1 Determining the Thickness of a Contaminated Zone with an Area Greater Than 100 m^2

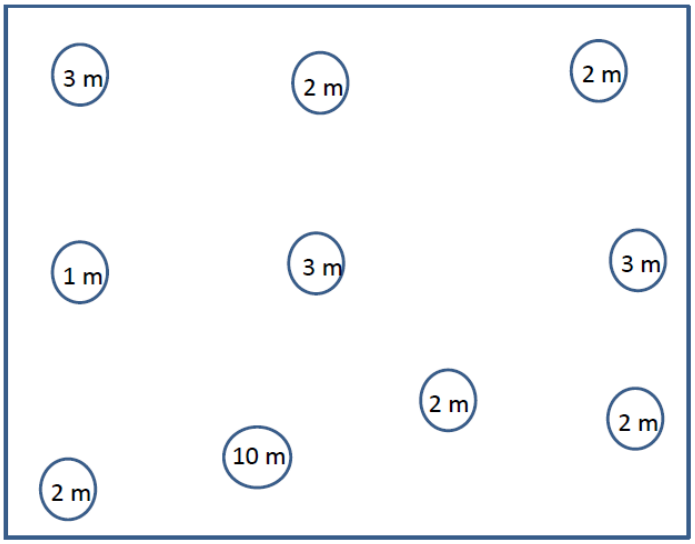


FIGURE 8.2.2 Determining the Thickness of a Contaminated Zone with an Area Less Than 100 m^2

8.2.3 Data Input Requirements

In RESRAD (onsite) and RESRAD-OFFSITE, the thickness of the contaminated zone is entered in units of meters (m). The default value is 2 m for both codes.

8.3 COVER DEPTH

8.3.1 Definition

The cover depth is the distance, in meters (m), from the ground surface to the location of the uppermost soil sample with radionuclide concentrations that are clearly above background. The background concentration of a radionuclide is defined as the mean concentration in soil samples from nearby uncontaminated regions of the same soil type, plus twice the standard deviation of the counting statistics.

8.3.2 Measurement Methodology

Because the actual radionuclide distributions in a contaminated zone are non-uniform, the cover depths measured in different sampling boreholes may not be the same. For a contaminated zone with an area greater than 100 m^2, the average cover depth over an integral subarea of 100 m^2 is calculated first. If one or more boreholes in the 100-m^2 subarea have a cover depth less than one-third of the average cover depth, then one third of the average value may be considered a conservative estimate for the cover depth. The cover depth for the entire contaminated zone is then determined to be the same as the minimum average cover depth over the subareas. For a contaminated zone with an area less than 100 m^2, the average cover depth over the contaminated zone is calculated first. If any borehole in the area has a cover depth less than one-third of the average cover depth, then one-third of the average cover depth is recommended as a conservative value of the cover depth for the contaminated zone. To determine a more realistic value, however, DOE-approved, statistically based estimates are preferred (DOE 1991). Illustrative examples for determining cover depth of the contaminated zone are shown in Figures 8.3.1 and 8.3.2.

Figure 8.3.1 shows an example for determining the cover depth of a contaminated zone with an area greater than 100 m^2. This example is for illustrative purposes only. In this example the contaminated zone area is 200 $m^{2;}$ therefore it is divided in two subareas of 100 m^2 each. One subarea has seven boreholes, and the other subarea, six boreholes. The figure shows the measured cover depth in each borehole. The average depth for subarea 1 is [(0.3 + 0.2 + 0.1 + 0.1 + 0.2 + 0.2 + 1) ÷ 7] 0.3 m. None of the boreholes' depth is less than 0.1 m; therefore the cover depth of this subarea is 0.3 m. The average depth for subarea 2 is [(0.2 + 0.2 + 0.3 + 0.3 + 0.2 + 0.3) ÷ 6] 0.25 m. None of the boreholes' depth in this subarea is less than one-third of the average depth; therefore the cover depth for this subarea is 0.25 m. To determine the cover depth for this contaminated zone, the minimum average cover depth for the subarea, 0.25 m, is chosen.

Figure 8.3.2 shows an example for determining the cover depth for a contaminated zone with an area less than 100 m^2. This example is for illustrative purposes only. This area has five boreholes. The figure shows the measured cover depth in each borehole. The average cover depth for this area is [(0.35 + 0.4 + 0.05 + 0.4 + 0.3) ÷ 5] 0.3 m. One borehole's depth in this area is less than one-third of the average depth; therefore the cover depth is (0.3 ÷ 3) 0.1 m for this contaminated zone.

FIGURE 8.3.1 Determining the Cover Depth of a Contaminated Zone with an Area Greater Than 100 m^2

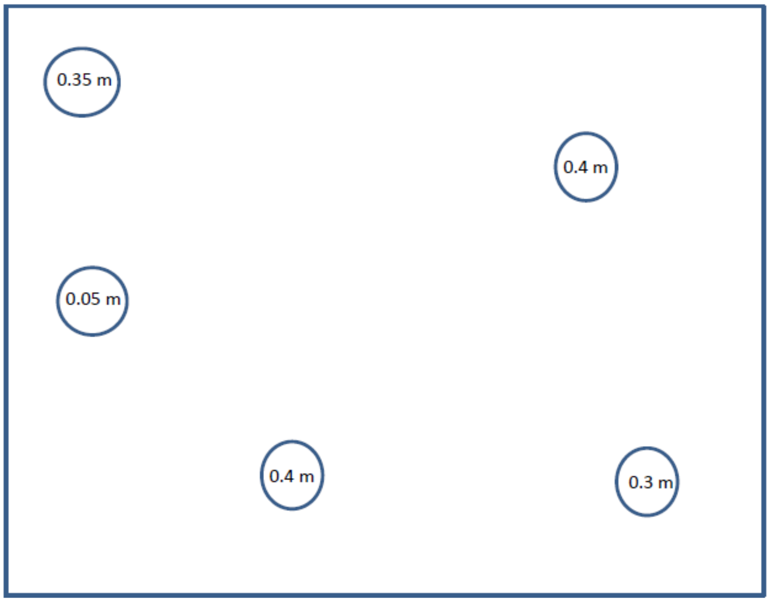


FIGURE 8.3.2 Determining the Cover Thickness of a Contaminated Zone with an Area Less Than 100 m^2

8.3.3 Data Input Requirements

In the RESRAD (onsite) and RESRAD-OFFSITE codes, the user is required to input a value for the cover depth. The default value used for the cover depth is 0 m.

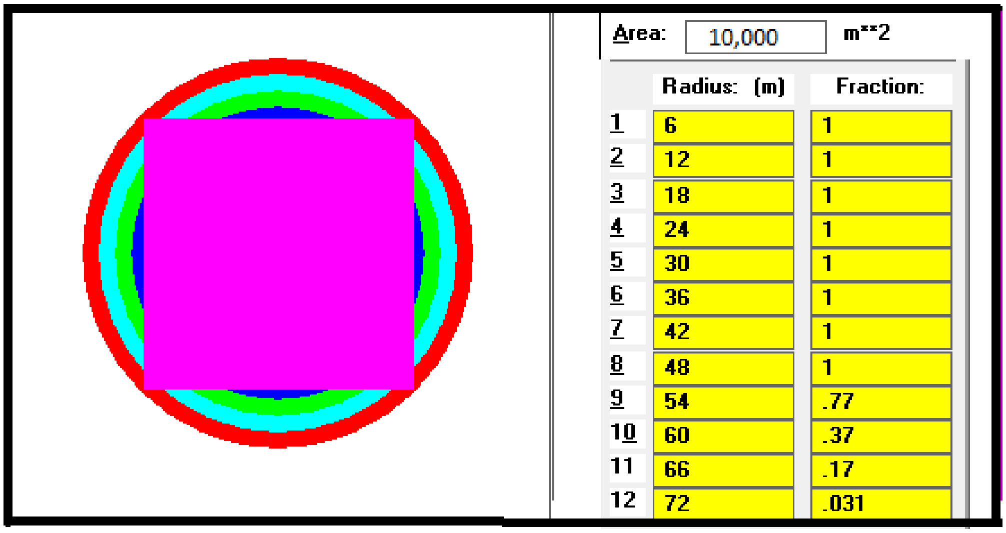
8.4 SHAPE FACTOR

8.4.1 Definition

The shape factor is calculated as the ratio of the dose estimates from a noncircular-shaped contaminated area to a reference shape. The reference shape is a fully contaminated, circularly shaped zone encompassing the given shape, centered at the receptor location. A shape factor is used to correct for a noncircular-shaped contaminated area on the basis of an ideally circular zone. The shape factor for a circular contaminated area is 1.0. For an irregularly shaped contaminated area, the shape factor is obtained by enclosing the irregularly shaped contaminated area in a circle, multiplying the area factor of each annulus by the fraction of the annulus area that is contaminated, summing the products, and dividing by the area factor of a circular contaminated zone that is equivalent in area (Kamboj et al. 1998, 2002). The area factor is the ratio of the dose from the finite geometry to the infinite slab geometry.

8.4.2 Data Input Requirements

In the RESRAD codes, the 12 annular area fields comprising the shape factor data appear if the “non-circular” box in the OCCUPANCY window is checked. The shape-factor data is obtained by drawing 2 to 12 concentric circles emanating from the center of the contaminated area. The outermost circle circumscribes the entire contaminated zone. For each annular ring, the outer radius and fraction of the ring within the contaminated zone should be entered. In the example in Figure 8.4.1, the total contaminated area is covered by 12 annular rings of increasing radius and corresponding annular area fractions. For the first eight annular rings, the total area of the contamination is within the annular rings and the fractions inside the annular rings decrease for the outer rings.



Area: 10,000 m**2

	Radius: (m)	Fraction:
1	6	1
2	12	1
3	18	1
4	24	1
5	30	1
6	36	1
7	42	1
8	48	1
9	54	.77
10	60	.37
11	66	.17
12	72	.031

FIGURE 8.4.1 Example of Shape Factor Data

9 MISCELLANEOUS PARAMETERS

9.1 RADIATION DOSE LIMIT

9.1.1 Definition

The radiation dose limit is the annual radiation dose limit used to derive site-specific soil guideline values (i.e., cleanup criteria). The calculated guidelines are linearly inversely proportional to the dose limit. If the RESRAD (onsite) and RESRAD-OFFSITE codes are used to calculate doses (rather than soil guidelines), the input value of the radiation dose limit will not affect the calculated doses.

The RESRAD (onsite) and RESRAD-OFFSITE codes provide the option to use dose conversion factors (DCFs) based on ICRP Publication 26 (ICRP 1977) or ICRP Publication 60 (ICRP 1991) methodology. The radiation dose used is the total effective dose equivalent (effective dose equivalent from external radiation plus the committed effective dose equivalent from internal radiation) if ICRP 26 methodology-based DCFs are used. If ICRP 60 methodology-based DCFs are used, radiation dose is the total effective dose (effective dose from external radiation plus the committed effective dose from internal radiation).

9.1.2 Data Input Requirements

In the RESRAD (onsite) and RESRAD-OFFSITE codes, 25 mrem/yr (0.25 mSv/yr) is used as the default value for the radiation dose limit. This value is consistent with the dose constraint specified in DOE Order 458.1 (DOE 2011) for each specific clearance of real property for any actual or likely future use of the property and is also consistent with the NRC radiological criteria for license termination (i.e., the criteria for release of sites for unrestricted use) (Smith et al. 2003).

9.2 RADIONUCLIDE CONCENTRATION IN GROUNDWATER

9.2.1 Definition

The radionuclide concentration in groundwater is a measure of the concentration of the radionuclide in a well located at the downgradient edge of the contaminated zone. The groundwater concentration and the radionuclide concentration in soil should be measured simultaneously because they are used in RESRAD (onsite) as a pair to estimate the distribution coefficient. Any natural or non-site-related sources of groundwater contamination should be considered because such sources could increase groundwater concentrations and result in a false distribution coefficient.

9.2.2 Data Input Requirements

The radionuclide concentration in groundwater should be entered in units of picocuries per liter (pCi/L). Input values for the radionuclide concentration in groundwater are required only if the value of the “elapsed time of waste placement” parameter (see Section 9.3) is greater than zero. Only radionuclides with non-zero concentrations in soils will have non-zero concentrations in groundwater. These non-zero groundwater concentration inputs will invoke the calculation of soil/water distribution coefficients, and the input distribution coefficient values will be superseded by the calculated results.

9.3 ELAPSED TIME OF WASTE PLACEMENT

9.3.1 Definition

The elapsed time of waste placement is the duration between the placement of radioactive waste on-site and the performance of a radiological survey. It is possible that on-site radioactive wastes originated from different sources and have different placement times. Under these circumstances, an average value or a best representative value should be used.

The elapsed time of waste placement is not used in the RESRAD-OFFSITE code. When using the RESRAD (onsite) code for risk assessment, the information obtained during the radiological survey is input to derive soil guidelines for cleanup criteria. This information includes soil/water distribution coefficients, soil radionuclide concentrations, and so forth. The soil/water distribution coefficients are used for calculating the breakthrough and rise times of the groundwater contamination and for predicting the future radionuclide concentration in groundwater. In this case, the elapsed time of waste placement is zero. Non-zero values of this parameter should be input only when the soil/water distribution coefficients are not available and above-background-level groundwater radionuclide concentrations are measured in a radiological survey. Under such conditions, the input radionuclide concentration in groundwater, together with the elapsed time of waste placement, would be used to derive soil/water distribution coefficients and to predict the future radionuclide concentration in groundwater.

9.3.2 Data Input Requirements

In the RESRAD (onsite) code, the elapsed time of waste placement should be entered in units of years (yr). The default value of this parameter is set at zero.

9.4 INITIAL CONCENTRATIONS OF PRINCIPAL RADIONUCLIDES

9.4.1 Definition

A principal radionuclide is a radionuclide with a half-life longer than the cutoff half-life selected by the user in the RESRAD (onsite) and RESRAD-OFFSITE codes. The user can select the cutoff half-life from four choices—180 days, 30 days, 7 days, or 1 day—or input a value of 10 minutes or more. Radionuclides with a half-life of less than the cutoff are treated as associated radionuclides. The radionuclides "associated" with a principal radionuclide consist of all decay products down to, but not including, the next principal radionuclide in the chain. It is assumed that all associated radionuclides (except radon daughters) are in secular equilibrium with their principal radionuclide in the contaminated zone and also at the location of human exposure. Only the principal radionuclides in the contaminated zone need input values of radionuclide concentrations.

The single-radionuclide soil guidelines do not depend on the radionuclide concentrations in soil. Even if the radionuclide concentrations are not known, values for these guidelines can be obtained by entering any non-zero radionuclide concentration. The calculated doses, however, depend on the radionuclide concentrations; thus, doses calculated by RESRAD (onsite) are valid only if the soil radionuclide concentrations are known. When the radionuclide concentrations in soil and groundwater are used with the elapsed time of waste placement to derive the soil/water distribution coefficient, the values of the initial concentrations of the principal radionuclide must be known to obtain accurate results.

9.4.2 Measurement Methodology

A DOE-approved statistical approach (DOE 1991, Section 7) should always be considered as the first choice regarding the estimation of the soil concentration. When such an application is impossible, then the following approach will serve as a default procedure in determining the average soil concentration; however, this approach will result in a conservative estimate of the effective dose.

For a site-specific case, the distributions of radionuclides are non-uniform. The potential annual individual dose received through a particular pathway is an average of the non-uniform residual radioactivity over an area determined by the scenario activities; for example, the area of daily activities for external radiation or the size of the garden for the plant food pathway. The effect of vertical non-uniformities is taken into account by averaging the radionuclide concentrations in a 0.15-m-thick layer over the 100-m^2 area.

The initial concentration of a principal radionuclide is determined by the following procedures: For a contaminated zone with an area greater than 100 m^2, the average radionuclide concentration for any subzone with a 100-m^2 area and 0.15-m thickness is determined. If one or more soil samples within this subzone have radionuclide concentrations greater than three times the average radionuclide concentration, then the average radionuclide concentration of this subzone is replaced by one-third the maximum measured soil radionuclide concentration. The initial concentration of a principal radionuclide in the contaminated zone is the maximum value of the average subzone radionuclide concentration. For a contaminated zone with an area less than 100 m^2, the initial concentration of a principal radionuclide is the maximum average radionuclide concentration of the 0.15-m-thick subzones.

9.4.3 Data Input Requirements

In the RESRAD (onsite) and RESRAD-OFFSITE codes, the initial concentrations of principal radionuclides in the contaminated zone are expressed in units of picocuries per gram (pCi/g). The codes treat the contaminated zone as a uniformly contaminated area with a single radionuclide concentration at every point.

9.5 FRACTION OF TIME SPENT INDOORS

9.5.1 Definition

The fraction of time spent indoors is the average fraction of time in a year during which an individual stays inside a house or a building. A typical value lies around 0.7 (EPA 2011) (dimensionless).

9.5.2 Summary of Literature Review

The EPA *Exposure Factors Handbook* (EPA 2011) contains a comprehensive review of human activity patterns, including the time spent indoors. Table 9.5.1 lists the EPA-recommended average time fractions spent indoors, indoors at a residence, outdoors, and

TABLE 9.5.1 EPA-Recommended Average Time Fraction Spent in Different Environments

Age Group	Indoors	Indoors at Residence	Outdoors	On Sand/ Gravel	On Grass	On Dirt
1 to < 2 years	0.94	0.74	0.03	0.03	0.05	0.04
2 to < 3 years	0.91	0.68	0.05	0.04	0.04	0.03
3 to < 6 years	0.89	0.66	0.07	0.04	0.05	0.04
6 to < 11 years	0.86	0.62	0.09	0.05	0.05	0.04
11 to < 16 years	0.88	0.62	0.07	0.05	0.05	0.03
16 to < 21 years	0.87	0.58	0.07	0.06	0.04	0.02
18 to < 65 years	0.80	0.66	0.20	NA[a]	0.04	NA

[a] NA = value not available.

Source: Calculated from minutes per day listed in Table 16-1, EPA (2011).

in different outdoor activities. The time spent indoors is higher compared to the average time spent indoors at a residence. Young children spend more time indoors compared to adults (age group 18 to <65 years). The average time spent indoors by an adult member of the public is 0.66. Table 9.5.2 lists the statistics for fraction of time spent indoors at a residence. The values can be used as the amount of time spent indoors at a residence for evaluating the subsistence farmer or suburban resident scenarios. Table 9.5.3 lists the statistics for fraction of time spent working in a main job. Some jobs require working indoors and others require working outdoors. The values can be used as the amount of time spent indoors at a work location for evaluating the worker scenario. Tables 9.5.2 and 9.5.3 also provide the census region data.

For short-term realistic evaluations, site- and scenario-specific data are more appropriate and should be used whenever possible. A typical local or regional value should be more representative, however, if long-term evaluations are considered.

9.5.3 Data Input Requirements

In the RESRAD (onsite) and RESRAD-OFFSITE codes, the user is required to input a value for the fraction of time spent indoors that represents conditions on-site at the primary contamination area. In the RESRAD-OFFSITE code, the user is also required to input a value for the fraction of time spent indoors at an off-site dwelling. This is a dimensionless parameter and should be entered as a decimal fraction rather than as a percentage. The sum of the fraction of time spent indoors onsite, the fraction of time spent outdoors onsite, and the fraction of time spent off-site (only required as input in RESRAD-OFFSITE) should not exceed 1.

The RESRAD (onsite) and RESRAD-OFFSITE codes can be used to analyze many potential exposure scenarios, such as subsistence farmer, suburban resident, industrial worker, and recreationist. A default value of 0.5 was adopted in the RESRAD (onsite) and RESRAD-OFFSITE codes for the fraction of time spent indoors, a value representative of a residential scenario. For RESRAD (onsite), it is assumed that the time is spent on-site, and for RESRAD-OFFSITE it is assumed that the time is spent off-site. For other scenarios, the fraction of time spent indoors should be changed according to the activity pattern associated with the scenario.

9.6 FRACTION OF TIME SPENT OUTDOORS

9.6.1 Definition

The fraction of time spent outdoors is the average fraction of time in a year during which an individual is outdoors. This is a dimensionless parameter, and the value lies around 0.25 at the 90th percentile (EPA 2011).

TABLE 9.5.2 Fraction of Time Spent Indoors at a Residence

Census Region	Age Group	N[a]	Mean	Minimum	5%	25%	50%	75%	90%	95%	99%	Maximum
Whole U.S.	All	9343	0.70	0.01	0.40	0.55	0.68	0.86	0.97	1.00	1.00	1.00
Whole U.S.	<1 year	187	0.70	0.18	0.39	0.55	0.66	0.85	1.00	1.00	1.00	1.00
Whole U.S.	1 to 4 years	498	0.84	0.19	0.55	0.74	0.88	0.98	1.00	1.00	1.00	1.00
Whole U.S.	5 to 11 years	700	0.70	0.13	0.48	0.59	0.68	0.81	0.93	0.98	1.00	1.00
Whole U.S.	12 to 17 years	588	0.67	0.07	0.41	0.56	0.66	0.80	0.91	0.98	1.00	1.00
Whole U.S.	18 to 64 years	6022	0.66	0.01	0.38	0.52	0.63	0.81	0.94	0.99	1.00	1.00
Whole U.S.	>64 years	1348	0.82	0.04	0.53	0.72	0.84	0.95	1.00	1.00	1.00	1.00
Northeast	All	2068	0.70	0.02	0.40	0.55	0.68	0.86	0.98	1.00	1.00	1.00
Midwest	All	2087	0.70	0.01	0.39	0.55	0.69	0.87	0.97	1.00	1.00	1.00
South	All	3230	0.69	0.01	0.41	0.56	0.67	0.85	0.97	1.00	1.00	1.00
West	All	1958	0.70	0.02	0.40	0.56	0.69	0.85	0.97	1.00	1.00	1.00

[a] Number of subjects in the survey.

Source: Calculated from minutes per day spent indoors listed in Table 16-16, EPA (2011)

TABLE 9.5.3 Fraction of Time Spent Working in a Main Job

Census Region	Population group	N[a]	Mean	Min	5%	25%	50%	75%	90%	95%	99%	Max
Whole U.S.	Workers	3259	0.226	0.000	0.057	0.188	0.238	0.271	0.314	0.352	0.442	0.685
Whole U.S.	Workers (18 to 64 years old)	2993	0.231	0.000	0.067	0.200	0.240	0.271	0.314	0.354	0.442	0.685
Whole U.S.	Full-time employed	2679	0.240	0.000	0.086	0.214	0.243	0.277	0.321	0.357	0.452	0.685
Northeast	Workers	721	0.226	0.000	0.057	0.193	0.235	0.271	0.318	0.352	0.452	0.685
Midwest	Workers	755	0.227	0.001	0.057	0.188	0.235	0.271	0.314	0.357	0.447	0.685
South	Workers	1142	0.227	0.000	0.050	0.193	0.240	0.271	0.314	0.350	0.428	0.685
West	Workers	641	0.224	0.002	0.057	0.186	0.238	0.271	0.313	0.347	0.419	0.514

[a] Number of subjects in the survey.

Source: Calculated from minutes per day spent working listed in Table 16-26, EPA (2011), and assuming individuals work for 250 days in one year.

9.6.2 Summary of Literature Review

The RESRAD (onsite) and RESRAD-OFFSITE codes can be used to analyze many potential exposure scenarios, such as subsistence farmer, suburban resident, industrial worker, and recreationist. Default values of 0.25 and 0.5 were adopted in the RESRAD (onsite) and RESRAD-OFFSITE codes, respectively, for the fraction of time spent outdoors. In RESRAD (onsite), it is assumed that this time is spent on-site, and in RESRAD-OFFSITE, it is assumed that this time is divided equally between four agricultural areas and at an off-site dwelling.

EPA *Exposure Factors Handbook* (EPA 2011) contains a comprehensive review of human activity patterns, including the time spent outdoors. Table 9.5.1 above lists the EPA recommended average time fractions spent indoors, indoors at a residence, outdoors, and in different outdoor activities. Young children spend less time outdoors compared to adults (age group 18 to <65 years). The average fraction of time spent outdoors by an adult member of the public is 0.20. Table 9.6.1 lists the time spent daily outdoors in recreational activities. Depending on the assumption regarding the number of days spent in recreational activities, the values can be used as the amount of time spent outdoors in a recreational scenario. Table 9.6.2 lists the fractions of time spent outdoors at a residence. The values can be used as the amount of time spent outdoors at a residence for evaluating the subsistence farmer or suburban resident scenario. Table 9.5.3 above lists the fraction of time spent working in a main job. Some jobs require working indoors and others require working outdoors. The values can be used as the amount of time spent outdoors at a work location for evaluating the worker scenario.

TABLE 9.6.1 Time Spent in Outdoor Recreational Activities (hr/day)

Census Region	Age Group	N[a]	Mean	Min	5%	25%	50%	75%	90%	95%	99%	Max
Whole U.S.	All	253	3.5	0.1	0.3	1.0	2.8	5.0	8.0	9.6	11.5	24.0
Whole U.S.	<1 year	2	5.6	3.3	3.3	3.3	5.6	8.0	8.0	8.0	8.0	8.0
Whole U.S.	1 to 4 years	13	2.8	0.3	0.3	0.5	2.2	3.0	6.2	10.5	10.5	10.5
Whole U.S.	5 to 11 years	21	3.4	0.5	1.0	1.5	2.8	4.1	6.0	9.6	9.8	9.8
Whole U.S.	12 to 17 years	27	2.6	0.1	0.1	1.0	2.3	3.8	7.0	7.0	7.8	7.8
Whole U.S.	18 to 64 years	158	3.7	0.1	0.5	1.3	2.9	5.2	8.4	9.8	11.5	24.0
Whole U.S.	>64 years	32	3.5	0.1	0.1	0.5	2.9	6.3	8.3	10.0	12.3	12.3
Northeast	All	52	3.2	0.1	0.5	1.0	2.7	3.9	6.2	9.6	11.5	11.5
Midwest	All	54	3.5	0.1	0.3	1.0	3.0	4.7	7.0	10.0	24.0	24.0
South	All	84	3.6	0.1	0.3	1.1	2.5	5.8	8.3	8.8	10.8	10.8
West	All	63	3.7	0.2	0.5	1.3	2.8	4.7	9.1	9.8	11.5	11.5

[a] Number of subjects in the survey.

Source: Calculated from time spent outdoors in recreational activities (min/day), as listed in Table 16-26 (EPA 2011).

TABLE 9.6.2 Fraction of Time Spent Outdoors (Outside the Residence)

Age Group	N[a]	Mean	Min	5%	25%	50%	75%	90%	95%	99%	Max
1 to 4 years	201	0.136	0.002	0.021	0.052	0.094	0.188	0.299	0.372	0.485	0.497
5 to 11 years	353	0.130	0.003	0.014	0.056	0.104	0.184	0.253	0.333	0.500	0.868
12 to 17 years	219	0.094	0.001	0.003	0.024	0.069	0.132	0.208	0.314	0.424	0.500
18 to 64 years	1809	0.100	0.001	0.003	0.021	0.063	0.138	0.250	0.326	0.497	0.750
>64 years	502	0.109	0.001	0.003	0.025	0.076	0.146	0.260	0.337	0.510	0.896

[a] Number of subjects in the survey.

Source: Calculated from time spent outdoors (min/day), as listed in Table 16-20 (EPA 2011).

A default value of 0.25 was adopted in RESRAD (onsite) for the fraction of time spent outdoors on-site. For realistic short-term evaluations, site-specific data are more appropriate and should be used whenever possible. A typical local or regional value should be more representative, however, if long-term evaluations are considered.

9.6.3 Data Input Requirements

In the RESRAD (onsite) code, the user is required to input a value for the fraction of time spent outdoors that represents conditions on-site at the primary contamination. In the RESRAD-OFFSITE code, the user is also required to input a value for the fraction of time spent outdoors on-site at the primary contamination, at an off-site dwelling, and in four agricultural areas. The agricultural areas can be on-site at the primary contamination or off-site. The fraction of time spent outdoors is a dimensionless parameter and should be entered as a decimal fraction rather than as a percentage. The sum of the fraction of time spent indoors on-site, the fraction of time spent outdoors on-site, and the fraction of time spent off-site (only required as input in RESRAD-OFFSITE) should not exceed 1.

10 REFERENCES

Abumurad, K.M., and M. Al-Tamimi, 2001, "Emanation Power of Radon and Its Concentration in Soil and Rocks," *Radiation Measurements* 34:423–426.

Agneessens, J.P., et al., 1978, "Modelisation de la Migration d'Elements dans les Sols. II. Determination du Coefficient de Dispersion et de la Porosite Efficace," *Pedologie* 27:373–388.

AHS (American Housing Survey), 2013*, Version 1.2 of Public Use File released on April 1, 2013: 2011 American Housing Survey for the United States—Complete Set of Tables*, http://www.huduser.org/portal/datasets/ahs.html.

AIHA (American Industrial Hygiene Association), 1971, "Ergonomics Guides: Ergonomics Guide to Assessment of Metabolic and Cardiac Costs of Physical Work," *American Industrial Hygiene Association Journal* 32:560–564.

Aldenkamp, F.J., et al., 1992, "An Assessment of in situ Radon Exhalation Measurements and the Relation between Free and Bound Exhalation Rates," *Radiation Protection Dosimetry* 45(1–4):449–453.

Allen, R.G., et al., 1998, *Crop Evapotranspiration—Guidelines for Computing Crop Water Requirements—FAO Irrigation and Drainage Paper 56*, Food and Agriculture Organization of the United Nations, Rome, http://www.fao.org/docrep/X0490E/x0490e00.htm. Accessed Oct. 29, 2014.

Amoozegar, A., and A.W. Warrick, 1986. "Hydraulic Conductivity of Saturated Soils: Field Methods", in A. Klute (editor), *Methods of Soil Analysis, Part 1: Physical and Mineralogical Methods*, 2nd ed., pp. 735–770, American Society of Agronomy, Inc., and Soil Science Society of America, Madison, WI.

Alpert, D.J., D.I. Chanin, and L.T. Ritchie, 1986, *Relative Importance of Individual Elements to Reactor Accident Consequences Assuming Equal Release Fractions,* NUREG/CR-4467, SAND-85–2575, Sandia National Laboratory, Albuquerque, NM.

Alzona, J., et al., 1979, "Indoor-Outdoor Relationships for Airborne Particulate Matter of Outdoor Origin," *Atmospheric Environment* 13:55–60.

Ames, L.L., and D. Rai, 1978, *Radionuclide Interaction with Rock and Soil Media, Volume I: Processes Influencing Radionuclide Mobility and Retention, Element Chemistry and Geochemistry;Conclusions and Evaluation*, EPA520/6-78-007-a, U.S. Environmental Protection Agency, Office of Radiation Programs, Las Vegas, NV.

Arcus-Arth, A., and R.J. Blaisdell, 2007, "Statistical Distributions of Daily Breathing Rates for Narrow Age Groups of Infants and Children," *Risk Analysis* 27(1):97–110.

Aris, R., 1958, “On the Dispersion of Linear Kinematic Waves,” *Proceedings of the Royal Society of London* Series A 245:268–277.

ASTM (American Society for Testing and Materials), 1992a, “Standard Test Method for Specific Gravity of Soils (D 854–91),” in *1992 Annual Book of ASTM Standards,* Sec. 4: Construction, Vol. 8: Soil and Rock; Dimension Stone; Geosynthetics, ASTM, Philadelphia.

ASTM, 1992b, “Standard Test Method for Density and Unit Weight of Soil in Place by the Sand-Cone Method (D 1556–90),” in *1992 Annual Book of ASTM Standards*, Sec. 4: Construction, Vol. 8: Soil and Rock; Dimension Stone; Geosynthetics, ASTM, Philadelphia.

ASTM, 1992c, “Standard Practice for the Thin-Walled Tube Sampling of Soils (D 1587–83),” in *1992 Annual Book of ASTM Standards*, Sec. 4: Construction, Vol. 8: Soil and Rock; Dimension Stone; Geosynthetics, ASTM, Philadelphia.

ASTM, 1992d, “Standard Test Method for Density and Unit Weight of Soil in Place by the Rubber Balloon Method (D 2167–84),” in *1992 Annual Book of ASTM Standards*, Sec. 4: Construction, Vol. 8: Soil and Rock; Dimension Stone; Geosynthetics, ASTM, Philadelphia.

ASTM, 1992e, “Standard Test Method for Laboratory Determination of Water (Moisture) Content of Soil and Rock (D 2216–90),” in *1992 Annual Book of ASTM Standards*, Sec. 4: Construction, Vol. 8: Soil and Rock; Dimension Stone; Geosynthetics, ASTM, Philadelphia.

ASTM, 1992f, “Standard Test Method for Permeability of Granular Soils (Constant Head) (D 2434–68),” in *1992 Annual Book of ASTM Standards*, Sec. 4: Construction, Vol. 8: Soil and Rock; Dimension Stone; Geosynthetics, ASTM, Philadelphia.

ASTM, 1992g, “Standard Test Methods for Density of Soil and Soil-Aggregate in Place by Nuclear Methods (Shallow Depth) (D 2922–91),” in *1992 Annual Book of ASTM Standards*, Sec. 4: Construction, Vol. 8: Soil and Rock; Dimension Stone; Geosynthetics, ASTM, Philadelphia.

ASTM, 1992h, “Standard Test Method for Density of Soil in Place by the Drive-Cylinder Method (D 2937–83),” in *1992 Annual Book of ASTM Standards,* Sec. 4: Construction, Vol. 8: Soil and Rock; Dimension Stone; Geosynthetics, ASTM, Philadelphia.

ASTM, 1992i, “Standard Test Method for Infiltration Rate of Soils in Field Using Double-Ring Infiltrometers (D 3385–88),” in *1992 Annual Book of ASTM Standards,* Sec. 4: Construction, Vol. 8: Soil and Rock; Dimension Stone; Geosynthetics, ASTM, Philadelphia.

ASTM, 1992j, “Standard Test Method for Distribution Ratios by the Short-Term Batch Method (D 4319–83),” in *1992 Annual Book of ASTM Standards,* Sec. 4: Construction, Vol. 8: Soil and Rock; Dimension Stone; Geosynthetics, ASTM, Philadelphia.

ASTM, 1992k, “Standard Test Method for Density of Soil in Place by the Sleeve Method (D 4564–86),” in *1992 Annual Book of ASTM Standards*, Sec. 4: Construction, Vol. 8: Soil and Rock; Dimension Stone; Geosynthetics, ASTM, Philadelphia.

ASTM, 1992l, “Standard Guide for Soil Sampling from the Vadose Zone (D 4700–91),” in *1992 Annual Book of ASTM Standards*, Sec. 4: Construction, Vol. 8: Soil and Rock; Dimension Stone; Geosynthetics; ASTM, Philadelphia.

ASTM, 1992m, “Standard Test Method for Measurement of Hydraulic Conductivity of Saturated Porous Materials Using a Flexible Wall Permeameter (D 5084–90),” in *1992 Annual Book of ASTM Standards*, Sec. 4: Construction, Vol. 8: Soil and Rock; Dimension Stone; Geosynthetics, ASTM, Philadelphia.

ASTM, 1992n, “Standard Guide for Comparison of Field Methods for Determining Hydraulic Conductivity in the Vadose Zone (D 5126–90),” in *1992 Annual Book of ASTM Standards*, Sec. 4: Construction, Vol. 8: Soil and Rock; Dimension Stone; Geosynthetics, ASTM, Philadelphia.

ASTM, 1992o, “Standard Test Method for Density of Soil and Rock In-Place at Depths Below the Surface by Nuclear Methods (D 5195–91),” in *1992 Annual Book of ASTM Standards*, Sec. 4: Construction, Vol. 8: Soil and Rock; Dimension Stone; Geosynthetics, ASTM, Philadelphia.

ASTM, 2007, “Standard Test Method for Density and Unit Weight of Soil in Place by the Sand-Cone Method (ASTM D1556–07).”

ASTM, 2008a, “Standard Test Method for Density and Unit Weight of Soil in Place by the Rubber Balloon Method (ASTM D2167–08).”

ASTM, 2008b, “Standard Test Method for Density and Unit Weight of Soil in Place by the Sleeve Method (ASTM D4564–08e1).”

ASTM, 2008c, “Standard Test Method for Density of Soil and Rock In-Place at Depths Below the Surface by Nuclear Methods (ASTM D5195–08).”

ASTM, 2009, “Standard Test Methods for Laboratory Determination of Density (Unit Weight) of Soil Specimens (ASTM D7263–09).”

ASTM, 2010a, “Standard Test Methods for Specific Gravity of Soil Solids by Water Pycnometer (ASTM D 854–10).”

ASTM, 2010b, “Standard Test Method for Density of Soil in Place by the Drive-Cylinder Method (ASTM D 2937–10).”

ASTM, 2010c, “Standard Test Methods for In-Place Density and Water Content of Soil and Soil-Aggregate by Nuclear Methods (Shallow Depth) (ASTM D6838–10).”

ASTM, 2010d, “Standard Test Method for Distribution Coefficients of Inorganic Species by the Batch Method (ASTM C1733–10)”

ASTM, 2010e, “Standard Test Method for Laboratory Determination of Water (Moisture) Content of Soil and Rock by Mass (ASTM D 2216–10)” (Published August 2010), doi:10.1520/D2216–10.

ATSDR (Agency for Toxic Substances and Disease Registry), 2001, *Summary report for the ATSDR soil-pica workshop*, U.S. Department of Health and Human Services, Atlanta, GA. March 20, 2001.

Baes, C.F., 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(1):17–28.

Baes, C.F., III, et al., 1984*, A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides through Agriculture*, ORNL-5786, Oak Ridge National Laboratory, Oak Ridge, Tenn.

Baixeras, C., et al., 2001, “Radon Emanation from Soil Samples,” *Radiation Measurements* 34:441–443.

Barillon, R., et al., 2005, “Direct Recoil Radon Emanation from Crystalline Phase. Influence of Moisture Content,” *Geochimica et Cosmochimica Acta* 69:2735–2744.

Barltrop, D., 1966, “The Prevalence of Pica,” *American Journal of Diseases in Children* 112:116–123.

Barnes, R.M., 1990, “Childhood Soil Ingestion: How Much Dirt Do Kids Eat?” *Analytical Approach* 62(19): 023A – 1033A.

Barney, G.S., and G.E. Brown, 1980, “The Kinetics and Reversibility of Radionuclide Sorption Reaction with Rocks, Progress Report for Fiscal Year 1979, “in J.F. Relyea (editor), *Task 4, Third Contractor Information Meeting*, Vol. II, pp. 261–308, PNL-SA-8571, Pacific Northwest Laboratory.

Barr, C.,, D. Schmidt, and S. Sherbini, 2010, *Development of Site-Specific Shielding Factors for Use in Radiological Risk Assessments,* Waste Management Conference, Phoenix, AZ, March 7–11.

Barretto, P.M.C., 1973, *Emanation Characteristics of Terrestrial and Lunar Materials and the* ^{222}*Rn Loss Effect on the U–Pb System Discordance,* Ph.D. Thesis, Rice University, Houston, TX.

Barton, T.P., and P.L. Ziemer, 1986, “The Effects of Particle Size and Moisture Content on the Emanation of Rn from Coal Ash.” *Health Physics* 50(5):581–588.

Bear, J., 1972, *Dynamics of Fluids in Porous Media*, American Elsevier Publishing Company, Inc., New York (reprinted by Dover Publications, Inc., 1988).

Bechtel SAIC, 2006, *Soil-Related Input Parameters for the Biosphere Model*, ANL-NBS-MD-000009, REV 3, September.

Bedient, P.B., and W.C. Huber, 1988, *Hydrology and Floodplain Analysis*, Addison-Wesley Publishing Company, Reading, Mass.

Behrman, L.E., and V.C. Vaughan, III, 1983, *Textbook of Pediatrics*, W.B. Saunders Company, Philadelphia.

Bellinger, D., et al., 1986, "Correlates of Low-Level Lead Exposure in Urban Children at 2 Years of Age," *Pediatrics* 77:826–833.

Beyeler, W.E., W.A. Hareland, F.A. Duran, T.J. Brown, E. Kalinina, D.P. Gallegos, and P.A. Davis, 1999, *Residual Radioactive Contamination From Decommissioning–Parameter Analysis*, NUREG/CR-5512, Vol. 3, SAND99-2146, Oct.

Bicknell J., 1974, "Lead Poisoning in Childhood," *Update* (England) 9:653–659.

Binder, S., et al., 1986, "Estimating Soil Ingestion: The Use of Tracer Elements in Estimating the Amount of Soil Ingested by Young Children," *Archives of Environmental Health* 41:341–345.

Biwer, B.M., S. Kamboj, J. Arnish, C. Yu, and S.Y. Chen, 2002, unpublished information, Argonne National Laboratory.

Black, K., et al., 2005, "Children's mouthing and food-handling behavior in an agricultural community on the US/Mexico border," *Journalof Exposure Science and Environmental Epidemiology* 15:244–251.

Blake, G.R., and K.H. Hartge, 1986a, "Bulk Density," in A. Klute (editor), *Methods of Soil Analysis, Part 1: Physical and Mineralogical Methods*, 2nd ed., pp. 363-376, American Society of Agronomy, Inc., and Soil Science Society of America, Madison, WI.

Blake, G.R., and K.H. Hartge, 1986b, "Particle Density," in A. Klute (editor), *Methods of Soil Analysis, Part 1: Physical and Mineralogical Methods*, 2nd ed., pp. 377-382American Society of Agronomy, Inc., and Soil Science Society of America, Madison, WI.

Bond, W.J., and D.E. Smiles, 1988, "Predicting the Average Movement of Reactive Solutes in Soils," *Soil Use Management* 4:115–120.

Bossew, P., 2003. "The Radon Emanation Power of Building Materials, Soils and Rocks," *Applied Radiation and Isotopes* 59:389–392.

Brady, N.C., 1984, *The Nature and Properties of Soils*, 9th ed., MacMillan Publishing Company, New York.

Breitner, D., et al., 2008, “Multidisciplinary Analysis of Finnish Esker Sediment in Radon Source Identification,” *Science of the Total Environment* 405:129–139.

Breitner, D., H. Arvela, K. Hellmuth, and T. Renvall, 2010, “Effect of Moisture Content on Emanation at Different Grain Size Fractions—A Pilot Study on Granitic Esker Sand Sample,” *Journal of Environmental Radioactivity* 101:1002–1006, 2010.

Brochu, P., et al., 2006, “Physiological Daily Inhalation Rates for Free-Living Individuals Aged 1 Month to 96 Years, Using Data from Doubly Labeled Water Measurements: a Proposal for Air Quality Criteria, Standard Calculations and Health Risk Assessment,” *Human Ecological Risk Assessment* 12:675–701.

Calabrese, E.J., et al., 1989, “How Much Soil Do Young Children Ingest: an Epidemiologic Study,” *Regulatory Toxicology and Pharmacology* 10(2):123−137.

Calabrese, E.J., et al., 1990, “Preliminary Adult Soil Ingestion Estimates: Results of a Pilot Study,” *Regulatory Toxicology and Pharmacology* 12:88–95.

Calabrese, E.J., et al., 1997a, “Soil Ingestion Estimates for Children Residing on a Superfund Site,” *Ecotoxicology Environmental Safety* 36(3):258−268.

Calabrese, E. J., et al., 1997b, “Soil Ingestion Rates in Children Identified by Parental Observation as Likely High Soil Ingesters,” *Journal of Soil Contamination* 6(3):271−279.

Campbell, G.S., 1974, “A Simple Method for Determining Unsaturated Conductivity from Moisture Retention Data,” *Soil Science* 117:311–314.

Canadell et al., 1996, “Maximum Rooting Depth of Vegetable Types at the Global Scale”, Oecologia 108:583–595.

Cantor, K.P., et al., 1987, “Bladder Cancer, Drinking Water Source, and Tap Water Consumption: A Case-Control Study,” *Journal of the National Cancer Institute* 79:1269–1279.

Carmody, J., J. Christian, and K. Labs, 1991, *Builder’s Foundation Handbook*, ORNL/CON-295, prepared for Oak Ridge National Laboratory, Oak Ridge, TN.

Charney, E., et al., 1980, “Increased Lead Absorption in Inner City Children: Where Does the Lead Come From?,” *Pediatrics* 65:226–231.

Chow, V.T., 1964, *Handbook of Applied Hydrology: A Compendium of Water-Resources Technology*. McGraw-Hill. New York

Clapp, R.B., and G.M. Hornberger, 1978. “Empirical Equations for Some Soil Hydraulic Properties,” *Water Resource Research* 14: 601–604.

Clausing, P., et al., 1987, “A Method of Estimating Soil Ingestion by Children,” *International Archives of Occupational and Environmental Health* (W. Germany) 59:73–82.Cohen, A.F., and B.L. Cohen, 1979. *Infiltration of Particulate Matter into Buildings*, NUREG/CR-1151, prepared by Sandia National Laboratories for U.S. Nuclear Regulatory Commission, Office of Nuclear Regulatory Research, Washington, DC., November.

Colombo, P., Doty, R., Dougherty, D., Fuhrmann, M., and Sanborn, Y., 1985, *Leaching Mechanisms of Solid Low-level Waste: the Literature Survey*. Technical Report, Low-Level Radioactive Waste Management – TIC-4500, BNL 51899/UC-708.

Cooper, M., 1957, *Pica*, Charles C. Thomas, Springfield, Ill.

Cosma, C., et al., 2001, “Determination of 222Rn Emanation Fraction and Diffusion Coefficient in Concrete Using Accumulation Chambers and the Influence of Humidity and Radium Distribution,” *Applied Radiation and Isotopes* 54:467–473.

Coughtrey, P.J., et al., 1985*, Radionuclide Distribution and Transport in Terrestrial and Aquatic Ecosystems, A Compendium of Data*, A.A. Balkema, Rotterdam, The Netherlands.

Culot, M.V.J., 1976, “Effective Diffusion Coefficient of Radon in Concrete: Theory and Method for Field Measurements,” *Health Physics* 30:263–270.

Currie, J.A., 1960a, “Gaseous Diffusion in Porous Media. Part 1. A Non-Steady State Method,” *British Journal of Applied Physics* 11:314–317.

Currie, J.A., 1960b, “Gaseous Diffusion in Porous Media. Part 2. Dry Granular Materials,” *British Journal of Applied Physics* 11:318–324.

Curwen, D., and L.R. Massie, 1994, *Irrigation Management in Wisconsin - the Wisconsin Irrigation Scheduling Program (WISP)*, A3600, I-01-94-2M-90-MSC.

Damkjær, A., and U. Korsbech, 1985, “Measurement of the Emanation of Radon-222 from Danish Soils,” *Science of the Total Environment* 45:343–350.

Danckwerts, P.V., 1953, “Continuous Flow Systems,” *Chemical Engineering Science* 2:1–13.

Danford, D.E., et al., 1982, “Pica and Mineral Status in the Mentally Retarded,” *American Journal of Clinical Nutrition* 35:958–967.

Danielson, R.E., and P.L. Sutherland, 1986, “Porosity,” in A. Klute (editor), *Methods of Soil Analysis, Part 1: Physical and Mineralogical Methods*, 2nd ed., pp. 443–462, American Society of Agronomy, Inc., and Soil Science Society of America, Madison, WI.

Danilk, W.R., 1981, *Laboratory Studies of Radionuclide Media, Oct. 1, 1979–Sept. 30, 1980,* LA-8586-PR, Los Alamos Scientific Laboratory, Los Alamos, NM.

Davis, S., and D. Mirick, 2006, “Soil Ingestion in Children and Adults in the Same Family.“ *Journal of Exposure Science and Environmental Epidemiology* 16:63−75.

Day, J.P., et al., 1975, “Lead in Urban Street Dust,” *Nature* 253:343–345.

Dickson, E.D., 2013, *Experimental Shielding Evaluation of the Radiation Protection Provided by Residential Structures,* Dissertation, Oregon State University.

Dickson, E.D., and D.M. Hamby, 2014, “External Shielding Evaluation of the Radiation Protection Provided by the Structurally Significant Components of Residential Structures,” *Journal of Radiological Protection* 34:201–221.

Dietz, R.N., et al., 1986, “Detailed Description and Performance of a Passive Perfluorocarbon Tracer System for Building Ventilation and Air Exchange Measurements,” in H.R. Trechsel and P.L. Lagus (editors), *Measured Air Leakage of Buildings: A Symposium*, ASTM STP 904, pp. 203–264, American Society for Testing and Materials, Philadelphia.

DOA (U.S. Department of the Army), 1970, *Engineering and Design: Laboratory Soils Testing*, EM 1110–2–1906, U.S. Army Corps of Engineers, Washington, DC.

DOE (U.S. Department of Energy), 1991, *Environmental Regulatory Guide for Radiological Effluent Monitoring and Environmental Surveillance*, DOE/EH-0173T, U.S. Department of Energy, Washington, DC.

DOE, 2010, *Final Environmental Impact Statement for Decommissioning and/or Long-Term Stewardship at the West Valley Demonstration Project and Western New York Nuclear Service Center,* DOE/EIS-0226. U.S. Department of Energy, Washington, DC.

DOE, 2011, *Radiation Protection of the Public and the Environment,* DOE O 458.1, U.S. Department of Energy, Office of Health, Safety and Security, Washington DC, February.

DOI (U.S. Department of the Interior), 1990a, “Procedure for Performing Field Permeability Testing by the Well Permeameter Method (USBR 7300–89),” in *Earth Manual, Part 2, A Water Resources Technical Publication*, 3rd ed., Bureau of Reclamation, Denver, CO.

DOI, 1990b, “Procedure for Constant-Head Hydraulic Conductivity tests in Single Drill Holes (USBR 7310–89).” In *Earth Manual, Part 2, A Water Resources Technical Publication,* 3rd ed., Bureau of Reclamation, Denver, CO.

Domenico, P.A., and F.W. Schwartz, 1990, *Physical and Chemical Hydrogeology,* John Wiley & Sons, Inc., New York.

Duggan, M.J., and S. Williams, 1977, "Lead-in-Dust in City Streets," *Science of the Total Environment* 7:91–97.

Ebbert, J.C., and Roe, R.D., 1998, Soil Erosion in the Palouse River Basin: Indications of Improvement, U.S. Geological Survey Fact Sheet FS-069-98, online at URL http://wa.water.usgs.gov/pubs/fs/fs069-98/. Accessed June 28, 2015.

EIA (*U.S. Energy Information Administration), 2014, 2009 Residential Energy Consumption Survey data; Housing characteristics tables; Structural and geographic characteristics by type of housing unit*, http://www.eia.gov/consumption/residential/data/2009/.

EPA (U.S. Environmental Protection Agency), 1977, *Proposed Guidance on Dose Limits for Persons Exposed to Transuranic Elements in the General Environment*, Office of Radiation Programs, Criteria and Standards Division, Washington, DC.

EPA, 1984a, *An Estimation of the Daily Food Intake Based on Data from the 1977–1978 USDA Nationwide Food Consumption Survey*, EPA-520/1-84-021, Office of Radiation Programs, Washington, DC.

EPA, 1984b, *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*, EPA-520/1-84-02, Washington, DC.

EPA, 1984c, *Tolerance Assessment System: Crop to Food Map*, draft report, August (data analyzed were compiled in the *USDA Nationwide Food Consumption Survey, 1977–1978*), Washington, DC.

EPA, 1985, *Development of Statistical Distributions or Ranges of Standard Factors Used in Exposure Assessments*, EPA-600/8-85-010, Washington, DC.

EPA, 1986, "Method 9100: Saturated Hydraulic Conductivity, Saturated Leachate Conductivity, and Intrinsic Permeability," in *Test Methods for Evaluating Solid Waste, Volume IC: Laboratory Manual Physical/Chemical Methods*, EPA SW-846, Part 3 of 4, 3rd ed., Washington, DC.

EPA, 1990a, *Exposure Factors Handbook*, EPA 600/8-89/043, Office of Health and Environmental Assessment, Washington, DC.

EPA, 1990b, *Risk Assessment Guidance for Superfund, Vol. I—Human Health Evaluation Manual, Part A*, OSWER Directive 9285.6-01, Washington, DC.

EPA, 1991, *Risk Assessment Guidance for Superfund, Vol. I: Human Health Evaluation Manual, Supplemental Guidance—Standard Default Exposure Factors*, OSWER Directive 9285.6-03, Washington, DC.

EPA, 1996, *Documenting Ground-Water Modeling at Sites Contaminated with Radioactive Substances*, EPA 540-R-96-003, Cooperative effort by U.S. EPA, Office of Solid Waste and Emergency Response, Office of Radiation and Indoor Air, Washington, DC.; U.S. Department of Energy, Office of Environmental Restoration, Washington, DC.; and U.S. Nuclear Regulatory Commission, Office of Nuclear Material Safety and Safeguards, Washington, DC, January.

EPA, 2002, *Estimated Per Capita Fish Consumption in the United States*, EPA-821-C-02-003, Office of Science and Technology, Washington, DC.

EPA, 2004, *Air Quality Criteria for Particulate Matter, Vols. I and II*, EPA/600/P-99/002aF and EPA/600/P-99/002bF, National Center for Environmental Assessment-RTP Office, Office of Research and Development, Research Triangle Park, NC.

EPA, 2009, *Metabolically Derived Human Ventilation Rates: A Revised Approach Based upon Oxygen Consumption Rates*, EPA 600/R-06/129F, Office of Research and Development, Washington, DC, May.

EPA, 2011, *Exposure Factors Handbook: 2011 Edition*, EPA/600/R-090/052F, Office of Research and Development, Washington, DC, September.

EPA, 2014, *Office of Air and Radiation,* home page for the AirData Web site, http://www.epa.gov/airdata.

Evans R. et al., 1996, *Soil, Water, and Crop Characteristics Important to Irrigation Scheduling*, AG 452–1, North Carolina Cooperative Extension Service, June, http://www.bae.ncsu.edu/programs/extension/evans/ag452-1.html.

Facciola, N.A., 2006, *Identification, Classification, and Correlation of Ultrafine Indoor Airborne Particulate Matter with Outdoor Values*, Master of Science Thesis, Department of Mechanical Engineering, University of Colorado, Boulder, CO.

Feldman M.D., 1986, "Pica: Current Perspectives," *Psychosomatics* (USA) 27:519–523.

Ferry, C., et al., 2002. "An Experimental Method for Measuring the Radon-222 Emanation Factor in Rocks," *Radiation Measurements* 35:579–583.

Fogh, C. L., et al., 1997, "Size Specific Indoor Aerosol Deposition Measurements and Derived I/O Concentration Ratios," *Atmospheric Environment* 31:2193–2203.

Forfar, J.O., and G.C. Arneil (editors), 1984, *Textbook of Pediatrics*, 3rd ed., Churchill Livingstone, London.

Foster, G.R., 1979, "Sediment Yield from Farm Fields: The Universal Soil Loss Equation and Onfarm 208 Plan Implementation," in *Universal Soil Loss Equation: Past, Present, and Future,* Soil Science Society of America Special Publication No. 8, pp. 17–24.

Freeze, R.A., and J.A. Cherry, 1979, *Groundwater*, Prentice Hall, Inc., Englewood Cliffs, NJ.

Fries, G.F., 1986, "Assessment of Potential Residues in Foods Derived from Animals Exposed to TCDD-Contaminated Soil," presented at *Dioxin 87: 6th International Symposium on Chlorinated Dioxin and Related Compounds*, Fukuoka, Japan, Sept.

Funtua, I.I., et al., 1997, "Radon Emanation Study of Uranium Ore Samples from N.E. Nigeria," *Applied Radiation and Isotopes* 48:867–869.

Gardner, W.H., 1986, "Water Content," in A. Klute (editor), *Methods of Soil Analysis, Part 1: Physical and Mineralogical Methods*, 2nd ed., pp. 493–544, American Society of Agronomy, Inc., and Soil Science Society of America, Madison, WI.

Garver, E., and M. Baskaran, 2004, "Effects of Heating on the Emanation Rates of Radon-222 from a Suite of Natural Minerals," *Applied Radiation and Isotopes* 61:1477–1485.

Gavrelis, N., et al., 2011, "An Analysis of the Proportion of the U.S. Population That Ingests Soil or Other Non-food Substances," *Human and Ecological Risk Assessment: An International Journal* 17(4):996–1012.

Gee, G.W., et al., 1980, *Interaction of Uranium Mill Tailings Leachate with Soils and Clay Liners*, NUREG/CR-1494, PNL-3381, prepared by Pacific Northwest Laboratory, Richland, Wash., for Office of Nuclear Regulatory Research, U.S. Nuclear Regulatory Commission, Washington, DC.

Gee, G.W., et al., 1983, "Mobility of Radionuclides in Soil," in *Chemical Mobility and Reactivity in Soil Systems*, Soil Science Society of America Special Publication No. 11:203–227.

Georgeson, C.C., and Payne, J.E., 1897, *Investigation of the Root Development of Some Forage Plants*, Experimental Station of the Kansas State Agricultural College, Manhatton, Bulletin No. 75, August.

Geraghty, J.J., et al., 1973, *Water Atlas of the United States*, Water Information Center, Port Washington, NY.

Gibb, J.P., M.J. Barcelona, J.D. Ritchey, and M.H. LeFaivre, 1984, *Effective Porosity of Geologic Materials: First Annual Report*, SWS Contract Report 351, Illinois Department of Energy and Natural Resources, State Water Survey Division, Ground Water Section, Champaign, IL, September.

Gilbert, T.L., et al., 1983, unpublished information, Argonne National Laboratory.

Gilbert, T.L., et al., 1989, *A Manual for Implementing Residual Radioactive Material Guidelines*, ANL/ES-160, DOE/CH/8901, prepared by Argonne National Laboratory, Argonne, IL, for U.S. Department of Energy, Washington, DC, June.

Gil-Garcia, C., A. Rigol, and M. Vidal, 2009a, “New Best Estimates for Radionuclide Solid–Liquid Distribution Coefficients in Soils, Part 1: Radio Strontium and Radio Cesium,” *Journal of Environmental Radioactivity* 100:690–696.

Gil-Garcia, C., K. Tagami, S. Uchida, A. Rigol, and M. Vidalm, 2009b, “New Best Estimates for Radionuclide Solid–Liquid Distribution Coefficients in Soils, Part 3: Miscellany of Radionuclides (Cd, Co, Ni, Zn, I, Se, Sb, Pu, Am, and Others),” *Journal of Environmental Radioactivit,* 100:704–715.

Gillies, M.E., and H.V. Paulin, 1983, “Variability of Mineral Intakes from Drinking Water: A Possible Explanation for the Controversy over the Relationship of Water Quality to Cardiovascular Disease,” *International Journal of Epidemiology* 12:45–50.

Glickman, L.T., 1981, “Pica Patterns, Toxocariasis, and Elevated Blood Lead in Children,” *American Journal of Tropical Medicine and Hygiene* 30:77–80.

Gnanapragasam, E.K., and C. Yu, 1997, *Application of the RESRAD Computer Code to VAMP Scenario S*, ANL/EAD/TM-70, Argonne National Laboratory, Argonne, IL, March.

Greeman, D.J., and A.W. Rose, 1996, “Factors Controlling the Emanation of Radon and Thoron in Soils of the Eastern U.S.A.,” *Chemical Geology* 129:1–14.

Grimsrud, D.T., et al., 1983, *Calculating Infiltration: Implications for a Construction Quality Standard*, LBL-9416, Lawrence Berkeley Laboratory, Berkeley, CA.

Grogan, H.A., 1985, *Concentration Ratios for BIOPATH: Selection of the Soil-to-Plant Concentration Ratio Database,* Report EIR-Bericht Nr, 575, Swiss Federal Institute for Reactor Research, Geneva, Switzerland.

Gutiérrez, J.L., et al., 2004, “Radon Emanation Measurements Using Silicon Photodiode Detectors,” *Applied Radiation and Isotopes* 60:583–587.

Hauth, J.J., 1967, “Vibrational Compaction of Nuclear Fuels,” , in H.H. Hausner, K.H. Roll, and P.K. Johnson (editors), *Vibratory Compacting*, pp. 253–276, Plenum Press, New York.

Hawley, J.K., 1985, “Assessment of Health Risk from Exposure to Contaminated Soil,” *Risk Analysis* 5:289–302.

Healy, J.W., and J.C. Rodgers, 1979, *Limits for the Burial of the Department of Energy Transuranic Waste*, LA-UR-79-100, Los Alamos Scientific Laboratory, Los Alamos, NM.

Hillel, D., 1980a, *Applications of Soil Physics*, Academic Press, Inc., New York.
Hillel, D., 1980b, *Fundamentals of Soil Physics*, Academic Press, Inc., New York.

Hoeffner, S.L., 1985, *Radionuclide Sorption on Savannah River Plant Burial Ground Soil—A Summary and Interpretation of Laboratory Data*, DP-1702, Savannah River Laboratory, E.I. du Pont de Nemours and Co., Aiken, SC.

Hoffmann, F.O., R.H. Gardner, and K.F. Eckerman, 1982, *Variability in Dose Estimates Associated with the Food Chain Transport and Ingestion of Selected Radionuclides*, NUREG/CR-2612, ORNL/TM-8099, Oak Ridge National Laboratory, Oak Ridge, TN.

Hogan, K., et al., 1998, "Integrated Exposure Uptake Biokinetic Model for Lead in Children: Empirical Comparisons with Epidemiologic Data," *Environmental Health Perspectives* 106(Supp 6):1557−1567.

Holub, R.F., and B.T. Brady, 1981, "The Effect of Stress on Radon Emanation from Rock," *Journal of Geophysical Research* 86:1776–1784.

Howard, A.J., et al., 1995. "Measurements of Rn-220 Emanation from Rocks," *Health Physics* 69:936–943.

HUD (U.S. Department of Housing and Urban Development), 1999, *Concrete Masonry Homes: Recommended Practices*, Prepared by National Association of Home Builders Research Center, Contract H-21134CA, September.

IAEA (International Atomic Energy Agency), 1982, *Generic Models and Parameters for Assessing the Environmental Transfer of Radionuclides from Routine Releases; Exposure of Critical Groups*, Safety Series No. 57, Vienna, Austria.

IAEA, 2008, *Field Estimation of Soil Water Content: A Practical Guide to Methods, Instrumentation and Sensor Technology*, IAEA-TCS-30, February.

IAEA, 2009, *Quantification of Radionuclide Transfer in Terrestrial and Freshwater Environments for Radiological Assessments*, Technical Document No. 1616, IAEA, Vienna, Austria, May.

IAEA, 2010a, *Handbook of Parameter Values for the Prediction of Radionuclide Transfer in Terrestrial and Freshwater Environments*, Technical Report Series no. 472, STI/DOC/010/472, IAEA, Vienna, Austria, January

IAEA, 2010b, *Analytical Methodology for the Determination of Radium Isotopes in Environmental Samples*, Analytical Quality in Nuclear Applications Series No. 19, IAEA, Vienna, Austria.

IAEA, 2013. *Measurement and Calculation of Radon Releases from NORM Residues*, Technical Report Series No. 474, IAEA, Vienna, Austria.

IAEA 2014. *Handbook of Parameter Values for the Prediction of Radionuclide Transfer to Wildlife*, Technical Reports Series 479, STI/DOC/010/479, IAEA, Vienna, Austria, ICRP

(International Commission on Radiological Protection), 1975, *Report on the Task Group on Reference Man*, ICRP Publication 23, Pergamon Press, New York.

ICRP, 1977, *Recommendations of the International Commission on Radiological Protection,* ICRP Publication 26, Annals of the ICRP, Vol. 1(3), Pergamon Press, New York.

ICRP, 1991, *1990 Recommendations of the International Commission on Radiological Protection,* ICRP Publication 60, Volume 21, No. 1–3, Pergamon Press, Oxford, England.

ICRP, 2002, *Basic Anatomical and Physiological Data for Use in Radiological Protection*, ICRP Publication 89, Pergamon Press, New York.

Isherwood, D., 1981, *Geoscience Data Base Handbook for Modeling a Nuclear Waste Repository*, NUREG/CR-0912, Vols. 1 and 2, U.S. Nuclear Regulatory Commission, Washington, DC.

Jackson, R.D., et al., 1962, "A Mechanized Device for Packing Soil Columns," *U.S. Department of Agriculture Bulletin* 41–42.

Javitz, H., 1980, *Seafood Consumption Data Analysis*, prepared by SRI International, Stanford, CA, for U.S. Environmental Protection Agency Office of Water Regulations and Standards, EPA Contract 68–01–3887.

Johnson, M.G., D.T. Tingey, D.L. Philips, and M.J. Storm, 2001, "Advancing fine root research with minirhizotrons," *Environmental and Experimental Botany* 45, 263–289.

Jury, W.A., and G. Sposito, 1985, "Field Calibration and Validation of Solute Transport Models for the Unsaturated Zone," *Soil Science Society of America Journal* 49:1331–1341.

Kalkwarf, D.R., P.O. Jackson, and J.C. Kutt, 1985, "Emanation Coefficients for Rn in Sized Coal Fly Ash," *Health Physics* 48:429–436.

Kamboj, S., D. LePoire, and C. Yu, 2002, External Exposure Model in the RESRAD Computer Code, *Health Physics* 82(6):831–839.

Kamboj, S., et al., 2000, *Probabilistic Dose Analysis Using Parameter Distributions Developed for RESRAD and RESRAD-BUILD Codes*, NUREG/CR-6676, ANL/EAD/TM-89, July.

Kamboj, S., C. Yu, and D.J. LePoire, 1998, *External Exposure Model Used in the RESRAD Code for Various Geometries of Contaminated Soil,* ANL/EAD/TM-84, Argonne National Laboratory, Argonne, IL.

Keller, G., B. Hoffmann, and T. Feigenspan, 2001, "Radon Permeability and Radon Exhalation of Building Materials," *The Science of the Total Environment* 272:85–89.

Kemble, J.K., Sanders, D.C., 2000, *Basics of Vegetable Crop Irrigation*, ANR-1169, April, http://www.aces.edu/pubs/docs/A/ANR-1169/index2.tmpl.

Kim, A.G., 2002, "CCB Leaching Summary: Survey of Methods and Results," in *Proceedings of Coal Combustion By-Products and Western Coal Mines: A Technical Interactive Forum*, April 1618, 2002, Golden, CO, U.S. Department of Interior, Office of Surface Mining, Alton, Illinois and Coal Research Center, Southern Illinois University, Carbondale.

Kirkham, D., and W.L. Powers, 1972, *Advanced Soil Physics*, Wiley-Interscience, New York.

Klute, A., and C. Dirksen, 1986, "Hydraulic Conductivity and Diffusivity: Laboratory Methods," in A. Klute (editor), *Methods of Soil Analysis, Part 1: Physical and Mineralogical Methods*, 2nd ed., pp. 687–734, American Society of Agronomy, Inc., and Soil Science Society of America, Madison, WI.

Klysz, G., and J.-P. Balayssac, 2007, "Determination of Volumetric Water Content of Concrete Using Ground-Penetrating Radar," *Cement and Concrete Research* 37:1164–1171.

Knight, M.J., 1983, unpublished information, Argonne National Laboratory.

Koontz, M.D., and H.E. Rector, 1995, *Estimation of Distributions for Residential Air Exchange Rates, Final Report*, prepared for Office of Pollution and Toxics, U.S. Environmental Protection Agency, Washington, DC, March.

Krishnaswami, S., and D.E. Seidemann, 1988, "Comparative Study of ^{222}Rn, ^{40}Ar, ^{39}Ar and ^{37}Ar Leakage from Rocks and Minerals: Implications for the Role of Nanopores in Gas Transport through Natural Silicates," *Geochimica et Cosmochimica Acta* 52:655–658.

Landa, E.R., 1987a, "Radium-226 Contents and Rn Emanation Coefficients of Particle-size Fractions of Alkaline, Acid and Mixed U Mill Tailings," *Health Physics* 52:303–310.

Landa, E.R., 1987b. "Influence of Ore Type and Milling Process on ^{222}Rn Emanation Coefficients of U Mill Tailings," *Health Physics* 53:679–683.

Lapidus, L., and N.R. Amundson, 1952, "Mathematics of Adsorption in Beds. IV. The Effect of Longitudinal Diffusion in Ion Exchange Chromatographic Columns," *Journal of Physical Chemistry* 56:984–988.

Lepow M.L., et al., 1975, "Investigations into Sources of Lead in the Environment of Urban Children," *Environmental Research* 10:415–426.

Linsley, R.K., et al., 1982, *Hydrology for Engineers*, 3rd ed., McGraw-Hill, New York.

López-Coto, I., et al., 2009, "A Short-time Method to Measure the Radon Potential of Porous Materials,"*Applied Radiation and Isotopes* 67:133–138.

Lysandrou, M., A. Charalambides, and I. Pashalidis, 2007, “Radon Emanation from Phosphogypsum and Related Mineral Samples in Cyprus,” *Radiation Measurements* 42:1583–1585.

Maraziotis, E.A., 1987, “Theoretical Evaluation of the ^{222}Rn Emanation Coefficient for Coal Fly Ash,” *Health Physics* 52:297–302.

Markkanen, M., and H. Arvela, 1992, “Radon Emanation from Soils,” *Health Physics* 45:269–272.

Marshall, T.J., and J.W. Holmes, 1979, *Soil Physics*, Cambridge University Press, New York.

Marsily, G. de, 1986, *Quantitative Hydrogeology*, Academic Press, Orlando, FL.

McWorter, D.B., and D.K. Sunada, 1977, *Groundwater Hydrology and Hydraulics*, Water Resources Publications, Fort Collins, CO.

Megumi, K., and T. Mamuro, 1974. “Emanation and Exhalation of Radon and Thoron Gases from Soil Particles,” *Journal of Geophysical Research* 79:3357–3360.

Morris, D.A., and A.I. Johnson, 1967, *Summary of Hydrologic and Physical Properties of Rock and Soil Materials as Analyzed by the Hydrologic Laboratory of the U.S. Geological Survey 1948–1960*, Contributions to the Hydrology of the United States, United States Department of the Interior.

Napier, B.A., R.J. Fellows, and L.D. Minc, 2014, *Transfer Factors for Contaminant Uptake by Fruit and Nut Trees*, NUREG/CR-7174, PNNL-22975, Pacific Northwest Laboratory, Richland, WA, July.

NAS (National Academy of Sciences), 1977, *Drinking Water and Health, Volume 1*, National Academy Press, Washington, DC.

National Gardening Association, Inc., 1987, *National Gardening Survey: 1986–1987*, National Gardening Association, Inc., Burlington, VT

Nazaroff, W.W., 1992, “Radon Transport from Soil to Air,” *Review of Geophysics* 30(2):137–160. Published online June 14, 2010.

Nazaroff, W.W., et al., 1988, “Soil as a Source of Indoor Radon: Generation, Migration, and Entry,” in W.W. Nazaroff and A.V. Nero (editors), *Radon and Its Decay Products in Indoor Air*, John Wiley & Sons, New York.

NCRP (National Council on Radiation Protection and Measurements), 1996, *Screening Models for Releases of Radionuclides to Atmosphere, Surface Water, and Ground*, Report No. 123, NCRP, Bethesda, MD, January.

NCRP, 1999, *Recommended Screening Limits for Contaminated Surface Soil and Review of Factors Relevant to Site-Specific Studies*, Report No. 129, NCRP, Bethesda, MD.

Nero, A.V., 1988, "Radon and Its Decay Products in Indoor Air: An Overview," in W.W. Nazaroff and A.V. Nero (editors), *Radon and Its Decay Products in Indoor Air*, John Wiley & Sons, New York.

Newell, C. J., L.P. Hopkins, and P.B. Bedient, 1990, "A Hydrogeologic Database for Ground-Water Modeling." *Groundwater* 28:703–714. doi: 10.1111/j.1745-6584.1990.tb01986.x

Ng, Y.C., et al., 1982a, *Soil-to-Plant Concentration Factors for Radiological Assessments,* report NUREG/CR-2975, UCID-19463, Lawrence Livermore National Laboratory, Livermore, CA.

Nielson, K.K., et al., 1982, *Comparison of Radon Diffusion Coefficients Measured by Transient-Diffusion and Steady-State Laboratory Methods*, NUREG/CR-2875, U.S. Nuclear Regulatory Commission, Washington, DC.

Nikula, A., 1982, *Sorption in Typical Finnish Soils and Some Fracture Filling of Bedrock* (in Finnish), Imatra Power Company, Helsinki, Finland.

NMFS (National Marine Fisheries Service), 1986, *Fisheries of the United States, 1985,* Current Fisheries Statistics No. 8368, U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Washington, DC.

NOAA (National Oceanic and Atmospheric Administration), 1982a, *Evaporation Atlas for the Contiguous 48 United States*, NOAA Technical Report NWS 33, U.S. Department of Commerce, Washington, DC.

NOAA, 1982b, *Mean Monthly, Seasonal, and Annual Pan Evaporation for the United States*, NOAA Technical Report NWS 34, U.S. Department of Commerce, Washington, DC.

NOAA, 2014, *Fish Watch—The Surprising Sources of Your Favorite Seafood: The Top 10 List*, http://www.fishwatch.gov/features/top10seafoods_and_sources_10_10_12.html. Accessed January 14, 2014.

NRC (U.S. Nuclear Regulatory Commission), 1977, *Calculation of Annual Doses to Man from Routine Releases of Reactor Effluents for the Purpose of Evaluating Compliance with 10 CFR 50, Appendix 2,* Regulatory Guide 1.109, Office of Standards Development, Rev. 1, Washington, DC, Oct.

NRC, 1983, *Radiological Assessment: A Textbook on Environmental Dose Analysis,* Report NUREG/CR-3332 (ORNL-5968), prepared by Oak Ridge National Laboratory, Oak Ridge, TN, for U.S. Nuclear Regulatory Commission, Washington, DC.

Özkaynak, H., Xue, J., Zartarian, V. G., Glen, G., and Smith, L., 2011, "Modeled Estimates of Soil and Dust Ingestion Rates for Children," *Risk Analysis* 31(4):592−608.

Oztunali, O.I., et al., 1981, *Data Base for Radioactive Waste Management, Impacts Analyses Methodology Report*, NUREG/CR-1759, Vol. 3, prepared by Dames and Moore, White Plains, NY, for Office of Nuclear Material Safety and Safeguards, U.S. Nuclear Regulatory Commission, Washington, DC.

Pao, E.M., et al., 1982, *Foods Commonly Eaten by Individuals: Amount Per Day and Per Eating Occasion,* Home Economics Report No. 44, U.S. Department of Agriculture, Washington, DC.

Paustenbach, D.J., et al., 1986, "A Critical Examination of Assumptions Used in Risk Assessments of Dioxin Contaminated Soil," *Regulatory Toxicology and Pharmacology* 6:284–307.

Pennington, J.A.T., 1983, "Revision of the Total Diet Study Food List and Diets," *Journal of the American Dietetic Association* 82:166–173.

Peterson, A.E., and J.B. Swan (editors), 1979, *Universal Soil Loss Equation: Past, Present, and Future,* SSSA Special Publication Number 8, Soil Science Society of America, Madison, WI.

Pierce, R.S., et al., 1981, *Commencement Bay Seafood Consumption Report*, preliminary report, Tacoma-Pierce County Health Department, Tacoma, WA.

Poffijn, A., et al., 1988, "On the Exhalation and Diffusion Characteristics of Concrete," *Radiation Protection Dosimetry* 24:203–206.

Poston, T. M., and D. C. Klopfer, 1986, *A Literature Review of the Concentration Ratios of Selected Radionuclides in Freshwater and Marine Fish*, PNL-5484, Pacific Northwest Laboratory, Richland, WA, September.

Pryor, S.C., R.J. Barthelmie, D.T. Young, E S. Takle, R W. Arritt, D. Flory, W J. Gutowski Jr., A. Nunes, and J. Roads, 2009, "Wind Speed Trends over the Contiguous United States," J. Geophys. Res., 114, D14105, doi: 10.1029/2008JD011416.

Puffer, H., et al., 1982, *Consumption Rates of Potentially Hazardous Marine Fish Caught in the Metropolitan Los Angeles Area*, EPA-600/3-82-070, prepared by the U.S. Environmental Protection Agency for Corvallis Environmental Research Laboratory, Corvallis, OR.

Putnam, J.J., et al., 1999, *Food Consumption, Prices, and Expenditure, 1970–1997,* U.S. Department of Agriculture, Statistical Bulletin No. 965, Washington, DC.

Rama, W.S. Moore, 1984. "Mechanism of Transport of U–Th Series Radioisotopes from Solids into Ground Water," *Geochimica Cosmochimica Acta* 48:395–399.

Rama, W.S.M., 1990, "Micro-crystallinity in Radioactive Minerals," *Nuclear Geophysics* 4:475–478.

Rasby, R.J. and T.M. Walz, 2011, *Water Requirements for Beef Cattle*, G 2060, NebGuide, University of Nebraska, March.

Renard, K.G., G.R. Foster, G.A. Weesies, D.K. McCool, and D.C. Yoder 1997, *Predicting Soil Erosion by Water: A Guide to Conservation Planning with the Revised Universal Soil Loss Equation (RUSLE),* Agriculture Handbook No. 703, U.S. Department of Agriculture, Argicultural Research Service.

Rifai, M.N.E., et al., 1956, *Dispersion Phenomena in Laminar Flow through Porous Media*, Engineering Research Series No. 93(2), Sanitary Engineering Research Laboratory, University of California, Berkeley, CA.

Ripple, C.D., et al., 1974, "Packing-Induced Radial Particle-Size Segregation: Influence on Hydrodynamic Dispersion and Water Transfer Measurement," *Soil Science Society of America Proceedings* 38:219–222.

Robertson, D.E., D.A. Cataldo, B.A. Napier, K.M. Krupa, and L.B. Sasser, 2003, *Literature Review and Assessment of Plant and Animal Transfer Factors Used in Performance Assessment Modeling*, NUREG/CR-6825, PNNL-14321, Pacific Northwest National Laboratory, Richland, WA, August.

Robischon, P., 1971, "Pica Practice and Other Hand-Mouth Behavior and Children's Developmental Level," *Nursing Research* 20:4–16.

Rogers, V.C., K.K., Nielson, and G.B. Merrel, 1983, *Radon Attenuation with Earthen Covers, 1983 Annual Report of Rogers and Associates Engineering Corporation,* Rep. RAE363, Salt Lake City, UT.

Rogers, V.C., et al., 1984, *Radon Attenuation Handbook for Uranium Mill Tailings Cover Design*, NUREG/CR-3533, U.S. Nuclear Regulatory Commission, Washington DC.

Rogers, V.C., and K.K. Nielson, 1991, "Multiphase Radon Generation and Transport in Porous Materials," *Health Physics* 60:807–815.

Rogers, V.C., et al., 1994, "Radon Diffusion Coefficients for Residential Concretes," *Health Physics* 67(3):261–265.

Rolston, D.E., 1986, "Gas Diffusivity," in A. Klute (editor), *Methods of Soil Analysis, Part 1: Physical and Mineralogical Methods*, 2nd ed., pp. 1089–1102, American Society of Agronomy, Inc., and Soil Science Society of America, Madison, WI.

Rosas J., K.Z. Jadoon, and T.M. Missimer, 2015," New Empirical Relationship between Grain Size Distribution and Hydraulic Conductivity for Ephemeral Streambed Sediments," *Environmental Earth Science* 73:1303–1315, doi: 10.1007/s12665-014-3484-2.

Rouston, R.C., et al., 1984, “Radionuclide Sorption on Low-Exchange Capacity Hanford Site Soils,” *Communication in Soil Science and Plant Analysis* 15:375–400.

Rovey II, C.W., and W.L. Niemann, 2001, “Wellskins and Slug Tests: Where’s the Bias?” *Journal of Hydrology* 243:120–132.

Ruffle, B., et al., 1994, Lognormal Distributions for Fish Consumption by the General U.S. Population, *Risk Analysis*, 14(4):395–404.

Sahoo, B.K., et al., 2010, “Radon Exhalation Studies in an Indian Uranium Tailings Pile,” *Radiation Measurement* 45:237–241.

Sakoda, A., et al., 2008. “Radioactivity and Radon Emanation Fraction of the Granites Sampled at Misasa and Badgastein,” *Applied Radiation and Isotopes* 66:648–652.

Sakoda, A., et al., 2010a, “Differences of Natural Radioactivity and Radon Emanation Fraction Among Constituent Minerals of Rock or Soil,” *Applied Radiation and Isotopes* 68(6):1180–1184.

Sakoda, A., et al., 2010b,“Erratum to ‘Differences of Natural Radioactivity and Radon Emanation Fraction between Constituent Minerals of Rock or Soil’ [Applied Radiation and Isotopes, 68(6):1180–1184, 2010],” *Applied Radiation and Isotopes* 68(12):2452.

Sakoda, A., et al., 2011, “A Comprehensive Review of Radon Emanation Measurements for Mineral, Rock, Soil, Mill Tailing and Fly Ash,” *Applied Radiation and Isotopes* 69:1422–1435.

Salinas, I.C.P., et al., 2006, “Gamma Shielding Factor for Typical Houses in Brazil,” *Radiation Protection Dosimetry* 121:420–424.

Sanford, W.E., and D.L. Selnick, 2012. “Estimation of Evapotranspiration cross the Conterminous United States Using a Regression with Climate and Land-Cover Data,” *Journal of the American Water Resources Association* 49(1):217230, doi: 10.1111⁄jawr.12010.

Sappington, F.C., and M.A. Phifer, 2005, *Moisture Content and Porosity of Concrete Rubble Study*, WSRC-TR-2005-00054, Revision 0, Savannah River National Laboratory, October.

Saxena, S.K., et al., 1974, “Experimental Evaluation of Chemical Transport in Water-Saturated Porous Media: 1. Nonsorbing Media,” *Soil Science* 118:120–126.

Saxton, K.E., and W.J. Rawls, 2006, Soil Water Characteristics Estimates by Texture and Organic Matter for Hydrologic Solutions, *Soil Science Society of America Journal* 70:1569–1578.

Sayre, J.W., et al., 1974, “House and Hand Dust as a Potential Source of Childhood Lead Exposure,” *American Journal of Diseases in Children* 127:167–170.

Schery, S.D., D.H. Gaeddert, and M.H. Wilkening, 1982, “Transport of Radon from Fractured Rock,” *Journal of Geophysical Research* 87:2969–2976.

Schery, S.D., D.H. Gaeddert, and M.H. Wilkening, 1984, “Factors Affecting Exhalation of Radon from a Gravelly Sandy Loam,” *Journal of Geophysical Research* 89:7299–7309.

Schmidt, D.W., K.L. Banovac, J.T. Buckley, D.W. Esh, R.L. Johnson, J.J. Kottan, C.A. McKenney, T.G. McLaughlin, and S. Schneider, 2006, *Consolidated Decommissioning Guidance–Characterization, Survey, and Determination of Radiological Criteria*, NUREG-1757, Vol. 2, Rev. 1, U.S. Nuclear Regulatory Commission, Office of Nuclear Material Safety and Safeguards, Division of Waste Management, Washington, DC, Sept.

Serne, R.J., and J.F. Relyea, 1981, “The Status of Radionuclide Sorption-Desorption Studies Performed by the WRIT Program,” in P.L. Hoffman (editor), *The Technology of High-Level Nuclear Waste Disposal*, Vol. I, DOE/TIC-4621, National Technical Information Service, U.S. Department of Commerce, Springfield, VA,

Sheppard, M.I., and D.H. Thibault, 1990, “Default Soil Solid/Liquid Partition Coefficients, K_ds, for Four Major Soil Types: A Compendium,” *Health Physics* 59:471–482.

Sheppard, M.I., and D.H. Thibault, 1991, “A Four-Year Mobility Study of Selected Trace Elements and Heavy Metals,” *Journal of Environmental Quality* 20:101–114.

Sheppard, M.I., et al., 1983, “Technetium and Uranium: Sorption by and Plant Uptake from Peat and Sand,” *Health Physics* 44:635–643.

Sheppard, M.I., et al., 1987, “Element Leaching and Capillary Rise in Sandy Soil Cores: Experimental Results,” *Journal of Environmental Quality* 16:273–284.

Sheppard, S.C., 2011, “Robust Prediction of K_d from Soil Properties for Environmental Assessment,” *Human and Ecological Risk Assessment: An International Journal* 17(1):263–279.

Sheppard, S.C., and M.I. Sheppard, 1989, “Impact of Correlations on Stochastic Estimates of Soil Contamination and Plant Uptake,” *Health Physics* 57:653–657.

Sherman, M.H. (editor), 1990, *Air Change Rate and Airtightness in Buildings*, American Society for Testing and Materials, Philadelphia.

Shinn, Jospeh H., 1998, *Post-Accident Inhalation Exposure and Experience with Plutonium,* UCRL-JC-131173, Lawrence Livermore National Laboratory, June.

Silker, W.B., 1981, *A Radon Attenuation Test Facility*, NUREG/CR-2243, U.S. Nuclear Regulatory Commission, Washington, DC.

Silker, W.B., and D.R. Kalkwarf, 1983, “Radon Diffusion in Candidate Soils for Covering Uranium Mill Tailings,” NUREG/CR-2924, PNL-4434, prepared by Pacific Northwest Laboratory for U.S. Nuclear Regulatory Commission, Washington, DC.

Simpson, D.B., and B.L. McGill, 1980, *User’s Manual for LADTAP II—A Computer Program for Calculating Radiation Exposure to Man from Routine Release of Nuclear Reactor Liquid Effluents*, NUREG/CR-1276, ORNL/NUREG/TDMC-1, Oak Ridge National Laboratory, Oak Ridge, TN, March.

Sisigina, T.I., 1974, “Assessment of Radon Emanation from the Surface of Extensive Territories,” *Nuclear Meteorology*, Israeli Program of Scientific Translations, Jerusalem, p. 239.

Skopp, J., 1985, “Analysis of Solute Movement in Structured Soils,” in J. Bouma and P.A.C. Raats (editors), *Proceedings of International Society of Soil Science Symposium on Water and Solute Movement in Heavy Clay Soil*, International Institute for Land Reclamation and Improvement, Wagemingen, The Netherlands, pp. 220–228.

Smith, J.A., 1992, “Precipitation,” Chapter 3 in D.R. Maidment (editor in chief), *Handbook of Hydrology*, McGraw-Hill, Inc.

Smith, T.B., et al., 2003, *Consolidated NMSS Decommissioning Guidance: Decommissioning Process for Materials Licensees*, NUREG-1757, Vol. 1, Rev. 1, U.S. Nuclear Regulatory Commission, Office of Nuclear Material Safety and Safeguards, Division of Waste Management, Washington, DC, Sept.

Staven, L.H., B.A. Napier, K. Rhoads, and D.L. Strenge, 2003, *A Compendium of Transfer Factors for Agricultural and Animal Products*, PNNL-13421, Pacific Northwest National Laboratory, Richland, WA, June.

Steingrobe, B., et al., 2001, “The Use of the Ingrowth Core Method for Measuring Root Production of Arable Crops – Influence of Soil and Root Disturbance during Installation of the Bags on Root Ingrowth into the Cores,” *European Journal of Agronomy* 15:143–151.

Stifelman, M., 2007, “Using Doubling-Labeled Water Measurements of Human Energy Expenditure to Estimate Inhalation Rates,” *Science of the Total Environment* 373:585–590.

Stoulos, S., M. Manolopoulou, and C. Papastefanou, 2004, “Measurement of Radon Emanation Factor from Granular Samples: Effects of Additives in Cement,” *Applied Radiation and Isotopes* 60:49–54.

Stranden, E., 1988, “Building Materials as a Source of Indoor Radon,” in W.W. Nazaroff and A.V. Nero (editors), *Radon and Its Decay Products in Indoor Air*, John Wiley & Sons, New York,

Stranden, E., A.K. Kolstad, and B. Lind, 1984. “The Influence of Moisture and Temperature on Radon Exhalation,” *Radiation Protection Dosimetry* 7:55–58.

Strenge, D.L., et al., 1987, *GASPAR II — Technical Reference and Users Guide*, NUREG/CR-4653, PNL-5907, Pacific Northwest Laboratory, prepared for U.S. Nuclear Regulatory Commission, Washington, DC.

Strong, K.P., and D.M. Levins, 1982, “Effect of Moisture Content on Radon Emanation from Uranium Ore and Tailings,” *Health Physics* 42:27–32.

Strong, K.P., et al., 1981, “Radon Diffusion through Uranium Tailings and Earth Cover,” in M. Gomez (editor), *Radiation Hazards in Mining: Control, Measurement, and Medical Aspects*, pp. 713‒719, Society of Mining Engineers, New York.

Tanner, A.B., 1964, “Radon Migration in the Ground: A Review,” in J.A.S. Adams and W.M. Lowder (editors), *Natural Radiation Environment*, University of Chicago Press, Chicago, pp. 161‒190.

Tucker, G.E., 2011, *CHILD Users' Guide*, Cooperative Institute for Research in Environmental Sciences (CIRES) and Department of Geological Sciences University of Colorado, http://csdms.colorado.edu/mediawiki/images/Child_users_guide.pdf.

Tulve, N.S., et al., 2002, “Frequency of Mouthing Behavior in Young Children.” *Journal of Exposure Science andEnvironmental Epidemiology* 12:259–264.

Turk, B.H., et al., 1987, *Indoor Air Quality and Ventilation Measurements in 38 Pacific Northwest Commercial Buildings: Volume I: Measurement Results and Interpretation: Final Report*, LBL22315-1/2, Lawrence Berkeley National Laboratory, Berkeley, CA. Available at http://www.ntis.gov/search/product.aspx?A BBR=DE88014064.

Uchida, S., and H. Kamada, 1987, “Sorption of Strontium on Soils in Layered and Aerated Zones,” *Hoken Butsuri* 22:179–187.

UDFCD (Urban Drainage and Flood Control District), 2001, *Urban Storm Drainage Criteria Manual*, 2008 revision, Denver, CO, http://www.udfcd.org/downloads/down_critmanual.htm.

USDA (U.S. Department of Agriculture), 1966, *Household Food Consumption Survey, 1965–1966, Report 12, Food Consumption of Households in the United States—Seasons and Year, 1965–1966*, U.S. Government Printing Office, Washington, DC.

USDA, 1983, *Food Consumption: Households in the United States, Seasons, and Year 1977–1978*, U.S. Government Printing Office, Washington, DC.

USDA, 2009, *Soil Survey Field and Laboratory Methods Manual*, Soil Survey Investigations Report No. 51, Version 1.0, National Soil Survey Center, Natural Resources Conservation Service., Lincoln, NE.

USDA Natural Resources Conservation Service (NRCS), 2008, *Soil Quality Indicators*, June.

USDA NRCS and CSSM (Center for Survey Statistics and Methodology), 2015, *Summary Report: 2012 National Resources Inventory*, Washington, DC, and Iowa State University, Ames, Iowa, http://www.nrcs.usda.gov/technical/nri/12summary. Accessed October 6, 2015.

Valocchi, A.J., 1985, “Validity of the Local Equilibrium Assumption for Modeling Sorbing Solute Transport through Homogeneous Soils,” *Water Resources Research* 21:808–820.

Van der Lugt, G., and L.C. Scholten, 1985, “Radon Emanation from Concrete and the Influence of Using Fly Ash in Cement,” *Science of Total Environment* 45:143–150.

Van der Spoel, W.H., et al., 1997, “Diffusive Transport of Radon in a Homogeneous Column of Dry Sand,” *Health Physics* 72(5):766–778.

Van der Spoel, W.H., et al., 1999, “Diffusive Transport of Radon in a Column of Moisturized Sand,” *Health Physics* 77:163–177.

Van Genuchten, M.Th.V., and P.J. Wierenga, 1986, “Solute Dispersion Coefficients and Retardation Factors,” in A. Klute (editor), *Methods of Soil Analysis, Part I: Physical and Mineralogical Methods*, 2nd ed., American Society of Agronomy, Inc., and Soil Science Society of America, Madison, WI, 1025–1054.

Van Wïjnen, J.H., Clausing, P., Brunekreff, B., 1990, “Estimated Soil Ingestion by Children,” *Environmental Research* 51(2):147−162.

Vandenhove, H., C. Gil-Garcia, A. Rigol, and M. Vidal, 2009, “New Best Estimates for Radionuclide Solid–Liquid Distribution Coefficients in Soils, Part 2: Naturally Occurring Radionuclides,” *Journal of Environmental Radioactivity* 100:697–703.

Veihmeyer, F.J., 1964, “Evapotranspiration,” in *Handbook of Applied Hydrology*, V.T. Chow (editor), McGraw-Hill, New York, 11.1 – 11.38.

Vermeer, D.E., and D.A. Frate, 1979, “Geophagia in Rural Mississippi: Environmental and Cultural Contexts and Nutritional Implications,” *American Journal of Clinical Nutrition* 32:2129–2135.

Walter, S.D., et al., 1980, “Age-Specific Risk Factors for Lead Absorption in Children,” *Archives of Environmental Health* 35:53–58.

Wang, T-H., M-H. Li, and S-P. Teng, 2009, “Bridging the Gap between Batch and Column Experiments: A Case Study of Cesium Adsorption on Granite,” *Journal of Hazardous Materials* 161:409–415.

Wang, Y-Y., B.M. Biwer, and C. Yu, 1993, *A Compilation of Radionuclide Transfer Factors for the Plant, Meat, Milk, and Aquatic Food Pathways and the Suggested Default Values for the RESRAD Code*, ANL/EAIS/TM-103, Argonne National Laboratory, Argonne, IL.

Ward, D. and K. McKague, 2007, *Water Requirements of Livestock*, Fact Sheet, Order No. 07–023, AGDEX 716/400, Ministry of Agriculture, Food and Rural Affairs, Ontario, Canada, May.

Washington State Department of Ecology, 2003, *An Assessment of Laboratory Leaching Tests for Predicting the Impacts of Fill Material on Ground Water and Surface Water Quality*, Publication No. 03-09-107.

Weaver, J.E., 1926, *Root Development of Field Crops*, 1st ed., McGraw-Hill, New York, http://www.soilandhealth.org/01aglibrary/010139fieldcroproots/010139toc.html.

Weaver, J.E., and Bruner, W.E., 1927, *Root Development of Vegetable Crops*, 1st ed., McGraw-Hill, New York, http://www.soilandhealth.org/01aglibrary/010137veg.roots/010137toc.html.

Willgoose, G. 2005, User Manual for SIBERIA (Version 8.30), Telluric Research.

White, G.J., and A.S. Rood, 2001, "Radon Emanation from NORM-contaminated Pipe Scale and Soil at Petroleum Industry Sites," *Journal of Environmental Radioactivity* 54:401–413.

Wilkening, M.H., 1974, "Radon-222 from the Island of Hawaii: Deep Soils Are More Important than Lava Fields or Volcanoes," *Science* 183:413–415.

Wischmeier, W.H., and D.D. Smith, 1978, *Predicting Rainfall Erosion Losses—A Guide to Conservation Planning*, Agricultural Handbook No. 537, prepared for U.S. Department of Agriculture by Science and Education Administration, Washington, DC.

Yu, C., 1999, "RESRAD Family of Codes and Comparison with Other Codes for Decontamination and Restoration of Nuclear Facilities,: Chapter 11 in M.J. Slobodien (editor), *Decommissioning and Restoration of Nuclear Facilities*, pp. 207–231. Health Physics Society 1999 Summer School Textbook.

Yu, C., 2006, "Modeling Radionuclide Transport in the Environment and Assessing Risks to Humans, Flora, and Fauna: The RESRAD Family of Codes," Chapter 5 in *Applied Modeling and Computations in Nuclear Science*, ACS Symposium Series 945, American Chemical Society, Washington, DC.

Yu, C., 2007, unpublished information, Argonne National Laboratory.

Yu, C., A.J. Zielen, J-J. Cheng, D.J. LePoire, E. Gnanapragasam, S. Kamboj, J. Arnish, A. Wallo III, W.A. Williams, and H. Peterson, 2001, *User's Manual for RESRAD* Version 6, ANL/EAD-4, Argonne National Laboratory, Argonne, IL, July.

Yu, C., et al., 1984, "Simultaneous Determination of Dispersion Coefficients and Retardation Factors for a Low-Level Radioactive Waste Burial Site," *Radioactive Waste Management Nuclear Fuel Cycle* 4:401–420.

Yu, C., et al., 1993, *Data Collection Handbook to Support Modeling the Impacts of Radioactive Material in Soil*, ANL/EAIS-8, Argonne National Laboratory, Argonne, IL, April.

Yu, C., W.A. Williams, C. Corredor, 2013a, "An Update on the Development and Application of the RESRAD Family of Codes," invited presentation at the HPS 58th Annual Conference, Madison, Wisconsin, July 8–11.

Yu, C., E.K. Gnanapragasam, J.-J. Cheng, S. Kamboj, and S.Y. Chen, 2013b, *New Source Term Model for the RESRAD-OFFSITE Code Version 3*, NUREG/CR-7127, ANL/EVS/TM/11-5, prepared by Argonne National Laboratory for U.S. Nuclear Regulatory Commission, December.

Yu, C., D. LePoire, E. Gnanapragasam, J. Arnish, S. Kamboj, B.M. Biwer, J-J. Cheng, A. Zielen, and S.Y. Chen, 2000, *Development of Probabilistic RESRAD 6.0 and RESRAD-BUILD 3.0 Computer Codes*, NUREG/CR-6697, ANL/EAD/TM-98, Argonne National Laboratory, Argonne, IL, December.

Yu, C., D.J. LePoire, J-J. Cheng, E. Gnanapragasam, S. Kamboj, J. Arnish, B.M. Biwer, A.J. Zielen, W.A. Williams, A. Wallo III, and H.T. Peterson, Jr., 2003, *User's Manual for RESRAD-BUILD Version 3*, ANL/EAD/03-1, June.

Yu, C., E. Gnanapragasam, B.M. Biwer, S. Kamboj, J-J. Cheng, T. Klett, D. LePoire, A. Zielen, S.Y. Chen, W.A. Williams, A. Wallo, S. Domotor, T. Mo, and A. Schwartzman, 2007, *User's Manual for RESRAD-OFFSITE, Version 2*, NUREG/CR-6937, ANL/EVS/TM-07–1, DOE/HS-0005, Argonne National Laboratory, Argonne, IL, June.

Zapalac, G.H., 1983, "Time-Dependent Method for Characterizing the Diffusion of Rn-222 in Concrete," *Health Physics* 45:377–383.

Zhuo, W., et al., 2006, "Modeling Radon Flux Density from the Earth's Surface," *Journal of Nuclear Science and Technology*, 43(4):479–482.

Ziai, M., 1983, *Bedside Pediatrics*, Little, Brown and Company, Boston,

Zotarelli, L., M.D. Dukes, and K.T. Morgan, 2010, Interpretation of Soil Moisture Content to Determine Soil Field Capacity and Avoid Over-Irrigating Sandy Soils Using Soil Moisture Sensors, AE460, University of Florida, IFAS Extension, Gainesville, FL.

Environmental Science Division
Argonne National Laboratory
9700 South Cass Avenue, Bldg. 240
Argonne, IL 60439-4847

www.anl.gov