FINAL REPORT

DOSES TO THE PUBLIC FROM ATMOSPHERIC RELEASES OF RADIONUCLIDES FROM THE IDAHO CHEMICAL PROCESSING PLANT AT THE IDAHO NATIONAL ENGINEERING LABORATORY (1957-1959)

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Centers for Disease Control and Prevention
Atlanta, Georgia 30335

SC&A, Inc.
6858 Old Dominion Drive, Suite 301
McLean, Virginia 22101

SENES Oak Ridge, Inc.
102 Donner Drive
Oak Ridge, Tennessee 37830

Authors:
A. Iulian Apostoaei / *SENES* Oak Ridge, Inc.
Brian A. Thomas / *SENES* Oak Ridge, Inc.
David C. Kocher / *SENES* Oak Ridge, Inc.
F. Owen Hoffman / *SENES* Oak Ridge, Inc.

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1.0 INTRODUCTION

The Idaho National Engineering Laboratory (INEL)[1] was established in 1949 as the National Reactor Testing Station, a site for building, testing and operation of various kinds of nuclear reactors and support facilities. The name was changed to INEL in 1974 to reflect a broader mission. The first reactor on the INEL site was the Experimental Breeder Reactor (EBR) No. 1, which achieved initial criticality in 1951, followed by the Material Testing Reactor (MTR) in 1952. The Idaho Chemical Processing Plant (ICPP), which was designed to recover uranium and other radionuclides from spent nuclear fuel, began operations in 1953. In all, more than 50 nuclear reactors were built and tested at INEL. A large number of research programs also were carried out in INEL facilities or on the INEL site.

The first releases of radionuclides to the atmosphere can be traced back to MTR operations in 1952. Many facilities within the INEL complex released radionuclides into the environment during different periods of time, raising concerns about radiation exposures of members of the public living around the INEL site.

In 1991 the Department of Energy (DOE) published a historical dose evaluation for the INEL (DOE 1991). The DOE review committee recommended a more detailed analysis using source documents and with public involvement. The Governor of Idaho asked the Centers for Disease Control and Prevention (CDC) to perform such an analysis. In Phase I of the analysis, which was completed in 1994, CDC developed a database of documents at INEL relevant to an environmental dose reconstruction. The Risk Assessment Corporation (RAC), under contract to CDC, added more documents to the database and published a report listing the major radionuclides released from INEL (RAC 2000). The present study represents Phase II of the analysis and was assigned by CDC to S. Cohen and Associates (SC&A, Inc., McLean, Virginia) and their subcontractors (*SENES* Oak Ridge, Inc., Oak Ridge, Tennessee, and *SENES* Consultants Limited, Ontario, Canada).

The present study estimates doses to members of the public from exposure to radionuclides released to the atmosphere from the ICPP at INEL. Previous investigations of releases throughout the operating history of INEL (DOE 1991; RAC 2000) indicated that airborne emissions from Radioactive Lanthanum (RaLa) process operations at the ICPP during the years 1957-1959 resulted in the greatest potential for offsite exposures of the public. During this period, releases of radionuclides to the atmosphere occurred during normal RaLa process operations and as a result of a criticality accident that took place on October 16, 1959.

As part of the present study, Wichner et al. (2005a, 2005b) estimated the activities of more than 130 radionuclides that were released to the atmosphere from RaLa process operations at the ICPP during the years 1957-1959, based on information obtained from historical Stack Monitoring Datasheets, official RaLa project reports, progress reports, operational logs, calculation sheets, and contemporary project letters. A summary of estimated releases is presented in Section 2 of this report. On the basis of a screening analysis summarized in Section 3, I-131 was determined to be by far the most important radionuclide released to the atmosphere

[1] In this report, we refer to the site by its historical name, rather than its current name, which is the Idaho National Engineering and Environmental Laboratory (INEEL).

from the ICPP. Iodine-131 and several additional radionuclides of potential concern were selected for inclusion in a dose reconstruction (Kocher 2005a, 2005b). For the selected radionuclides, concentrations in air were estimated at various locations in the region around INEL using the CALPUFF atmospheric transport modeling system (Radonjic et al. 2005), which is summarized in Section 5.

For the purpose of analyzing impacts of atmospheric releases from the ICPP, CDC requested the development of at least five scenarios for exposure of members of the public; at least one scenario should address onsite exposures and the others should address offsite exposures. A total of six basic exposure scenarios were developed to provide reasonably realistic representations of exposures of members of the public (Apostoaei and Reed 2005), including three scenarios describing offsite exposures (a rural resident, an urban resident, and a migrant farm worker) and three scenarios describing onsite exposures (an onsite rancher, a hunter of onsite game, and an onsite visitor). As described in Section 4, numerous exposure situations can be analyzed using these basic exposure scenarios (e.g., various ages at exposure, various types and amounts of milk consumed).

On the basis of screening calculations performed to select radionuclides of concern (Kocher 2005a and 2005b) and previous studies of emissions from INEL (DOE 1991), doses from exposure to I-131 are expected to be considerably higher than doses from the other radionuclides that were selected by the screening process. Thus the main focus of this report is estimation of doses from ingestion or inhalation of I-131.

The methodology used in this report to estimate doses from exposure to I-131 in an assumed scenario on the basis of estimated concentrations of radionuclides in air at various receptor locations is briefly discussed in Section 6, and details of the equations, assumptions and parameter values are presented in Appendix A. Appendix B provides estimates of concentrations of I-131 in food products that were calculated using the models and parameters in Appendix A. A detailed analysis of the estimated doses from offsite exposures to I-131 is summarized and discussed in Section 7.1.

A preliminary assessment was performed to estimate upper bounds of offsite doses from exposure to radionuclides other than I-131 and upper bounds doses from exposure within the INEL site boundary (Appendix C). Such bounding estimates are used to discuss the potential importance of offsite doses from radionuclides other than I-131 and exposures of the public on the INEL site (Section 7.2 and 7.3).

Releases of radionuclides into surface or ground water from the ICPP operations are not considered in this analysis. Most of INEL lies in a closed topographical depression, and surface water flows toward the Big and Little Lost River Sinks located in the northwest portion of the INEL. Surface water infiltrates the Big Lost River channel bottom and sinks, recharging the Upper Snake River Plain Aquifer, which flows beneath INEL in a southwesterly direction. Thus, people living offsite have no access to surface water sources that originate from INEL.

Most of the radioactivity in the Upper Snake River Plain Aquifer below the site originated from injections into special wells and seepage from liquid-waste disposal ponds that contain low levels of radioactivity. Use of injection wells started as early as 1952 and was discontinued in

1983 (Bowman et al. 1984). However, extensive monitoring of the aquifer has revealed no important levels of radioactivity in ground water outside the INEL boundary.

In addition to releases of radionuclides from RaLa process operations at the ICPP, CDC has determined that episodic atmospheric releases from initial engine tests (IETs) in the Aircraft Nuclear Propulsion (ANP) Program could have resulted in significant offsite exposures. As part of a dose reconstruction at INEL, emissions from three selected tests (IET 3, 4 and 10) out of the total of 26 tests in the ANP Program, were analyzed (Behling and Mauro 2005). Those three tests were responsible for most of the releases during the ANP Program. A rigorous analysis of releases from the selected IETs was hampered by the absence of original logbooks and other primary sampling data, which either no longer exist or have not yet been declassified for public use. Thus, for the selected IETs, only point estimates of total releases for each radionuclide were obtained on the basis of information retrieved from historical summary reports, and the intent of the analysis by Behling and Mauro (2005) was to provide source terms that did not underestimate actual releases. A time-dependence of releases during a particular IET could not be determined. Due to the lack of detailed information, potential impacts of releases during the selected IETs in the ANP Program have been investigated only by performing screening calculations (Kocher 2005b). A detailed reconstruction of doses resulting from releases during the ANP Program has not been performed at this time.

2.0 ATMOSPHERIC RELEASES OF RADIONUCLIDES FROM THE IDAHO CHEMICAL PROCESSING PLANT

The Radioactive Lanthanum (RaLa) process was designed to extract Ba-140 (half-life of 12.9 days), which decays to La-140 (40.2 hours), from irradiated reactor fuel. The latter isotope is an intense source of high-energy gamma rays that was used to evaluate the implosion process of a nuclear weapon. The short half-life of Ba-140 required the design of a process in which fuel elements irradiated in a nuclear reactor were dissolved shortly after irradiation.

The RaLa process at the Idaho Chemical Processing Plant (ICPP) used fuel elements irradiated in the Material Testing Reactor (MTR) located onsite at INEL. A total of 36 RaLa runs took place at the ICPP between February 1957 and December 1959. Additional releases of radionuclides occurred as a result of a criticality accident on October 16, 1959, when an air sparging operation inadvertently transferred a highly enriched uranium solution to a waste tank that was not designed for that purpose.

Essentially all releases from the ICPP during the years 1957-1959 were due to RaLa process operations (Wichner et al. 2005a and 2005b). Gases and aerosol particles generated during RaLa dissolution and extraction processes were captured by an off-gas system and then passed through a series of scrubbers and charcoal beds to a temporary 10,000-ft^3 storage tank. The number and type of filters changed with time during the period of operations. Before gases and aerosols were released to the atmosphere through the 250-ft ICPP stack, they were diluted by mixing with large amounts of the air from the ICPP building ventilation system, which created a total output flow rate of about 100,000 ft^3 per minute.

RaLa process operations and the RaLa off-gas system were complex and are difficult to model for the purpose of estimating atmospheric releases of radionuclides. However, radionuclide emissions through the ICPP stack were monitored continuously during 1957-1959 by sampling of air that went out the stack. Daily samples of air were analyzed for iodine content, and gross-beta and gross-alpha measurements of the samples were made after iodine was removed. Information obtained from Stack Monitoring Datasheets provides the basis for estimated releases of iodine and other radionuclides. Additional sources of information include official RaLa project reports, progress reports, operational logs, calculation sheets, and contemporary project letters.

The following sections summarize estimated releases of potentially important radionuclides during RaLa process operations at the ICPP. Estimated releases of isotopes of iodine, bromine, krypton, and xenon are given by Wichner et al. (2005a), and estimated releases of radionuclides attached to aerosols are given by Wichner et al. (2005b).

2.1 Releases of Iodine from Idaho Chemical Processing Plant during 1957-1959

Two isotopes of iodine were released in sufficiently large quantities to be of concern in regard to potential offsite exposures of the public: I-131 (half-life of 8.04 days) and I-133 (20.8 hours). Iodine-132 (2.3 hours) was not identified as a radionuclide of concern in a screening analysis (see Section 3), but the pattern and magnitude of I-132 releases are discussed in this section for the purpose of comparing them with releases of I-131 and I-133. On the basis of a screening analysis described by Kocher (2005a, b) and summarized in Section 3 of this report and previous studies of emissions from INEL (DOE 1991), I-131 is believed to be the most important radionuclide in regard to potential radiation doses to members of the public who resided near INEL during the years 1957-1959.

Releases of radioactive isotopes of iodine were estimated using measurements of I-131 in samples of air from the ICPP stack, as reported in Stack Monitoring Datasheets. This method is considered more reliable than theoretical modeling of the RaLa dissolving, extraction, storage, and off-gas systems. Iodine in stack air was collected during a 24-hour period (midnight to midnight) in a one-liter scrubber liquid sampler, which was later analyzed using a NaI crystal scintillation counter set to record the principal I-131 emissions. The sampled liquid also was analyzed to estimate the activity of I-132 starting in May 1958. After removal of iodine from the liquid, a gross-beta (i.e., beta minus iodine; β-I) and a gross-alpha (α) reading were taken and recorded.

The activity of I-131 released to the atmosphere was estimated directly from I-131 readings reported in the Stack Monitor Datasheets, adjusted for the collection efficiency of the liquid sampler. Releases of I-132 from May 1958 to December 1959 also were estimated from measurements reported in the Stack Monitoring Datasheets. Prior to May 1958, releases of I-132 were estimated on the basis of I-131 readings and an empirical relationship between the activities of I-131 and I-132 observed after 1958 and recorded in the Stack Monitoring Datasheets. The stack monitor system did not detect I-133. In contrast to I-132, which is produced directly in fission and by decay of its longer-lived Te-132 precursor, activities of I-131 and I-133 in stack emissions were not influenced by the presence of precursor radionuclides. Thus, the activity of I-133 was calculated using the measured activity of I-131 multiplied by the fission inventory ratio I-133/I-131 for each day after irradiation.

Similar methods were used to estimate releases that occurred as a result of the criticality accident on October 16, 1959, since the stack monitoring system operated correctly during and after the accident and the number of fissions (4×10^{19}) was determined with reasonable accuracy. Atmospheric releases from the criticality event began on October 16 and continued until a new RaLa run started on November 5. However, most of the iodine releases attributable to the criticality accident occurred within the first 8 days (October 16 – 23, 1959).

Since estimated releases following the criticality accident are similar to releases during a normal RaLa run and they occurred over several days, the criticality accident can be analyzed in the same way as a routine RaLa run from the point of view of estimating doses to the public. In this report, results are presented for all releases from the ICPP during the years 1957-1959, including

releases following the criticality accident, and a separate accounting of doses due to the criticality accident is not given.

An evaluation of the reliability of the stack monitoring system (Wichner et al. 2005a) indicated that the main sources of uncertainty were the efficiency of the liquid sampler, losses of iodine due to deposition in the sampling line (the pipe carrying air from the stack to the sampler), and, for short-lived I-132, decay during the 24-hour collection time.

As summarized in Table 2.1, about 3,200 Ci[2] (95% C.I.[3] = 2,400 – 5,100 Ci) of I-131 were released from the ICPP as a result of normal RaLa operations during 1957-1959 and the 1959 criticality accident (Wichner et al. 2005a). Similarly, about 37,000 Ci (95% C.I. = 24,800 – 58,000 Ci) of I-132 and 470 Ci (95% C.I. = 340 – 730 Ci) of I-133 were released to the atmosphere during the same time period. Releases of I-131 and I-132 following the criticality accident were less than 0.5% of the total releases of these isotopes during 1957-1959. However, about 30% of all I-133 was released following the criticality accident. Even though a much larger activity of I-132 was released to the atmosphere during 1957-1959, doses to members of the public who lived near INEL were much smaller than doses from I-131 or I-133, due to the much shorter half-life of I-132 (2.3 hours).

Detailed dose calculations presented in Section 7.1 were performed using estimated daily releases of iodine from February 1, 1957, to December 31, 1959. An example of estimated daily releases of iodine from the ICPP during January 1958 is given in Figure 2.1. Estimated daily releases following the criticality accident are shown in Figure 2.2. As noted previously, data in the two figures indicate that releases of I-131 following the criticality accident were similar to those during a typical RaLa run. In contrast, releases of I-132 were lower and releases of I-133 were much higher than the respective releases during a typical RaLa run. Even though the half-life of I-132 (2.3 hours) is shorter than the half-life of I-133 (20.8 hours), I-132 was released for many days after a RaLa run (or the criticality accident), because it was continuously produced by decay of its longer-lived Te-132 precursor (78.2 hours).

[2] 1 Ci = 3.7×10^{10} Bq (Becquerel). Table 2.1 presents estimated releases in Ci and Bq.

[3] C.I. = Confidence Interval. As used in this report, the confidence interval is actually a "credibility" interval, meaning that there is a high degree of confidence (in this case a subjective degree of belief of 95%) that the true activity released is inside this interval.

Table 2.1 Estimated atmospheric releases of radioactive iodine from the Idaho Chemical Processing Plant during 1957-1959

Routine Releases* (including October 16, 1959, criticality accident)			
Isotope	**95% Confidence Interval (C.I.)**		
	Lower bound	**Central Estimate**	**Upper Bound**
I-131	2,400 (87)	3,200 (120)	5,100 (190)
I-132	24,800 (920)	37,000 (1400)	58,000 (2100)
I-133	340 (13)	470 (17)	730 (27)
October 16, 1959, criticality accident			
I-131	9.4 (0.35)	13 (0.48)	20 (0.75)
I-132	130 (4.9)	230 (8.6)	450 (17)
I-133	110 (3.9)	140 (5.3)	230 (8.4)

* Releases are given in Curies (Ci) and, in parenthesis, in terabecquerel (TBq).
1 Ci = 3.7×10^{10} Bq = 0.037 TBq; 1 TBq = 10^{12} Bq.

Source: Wichner et al. 2005a

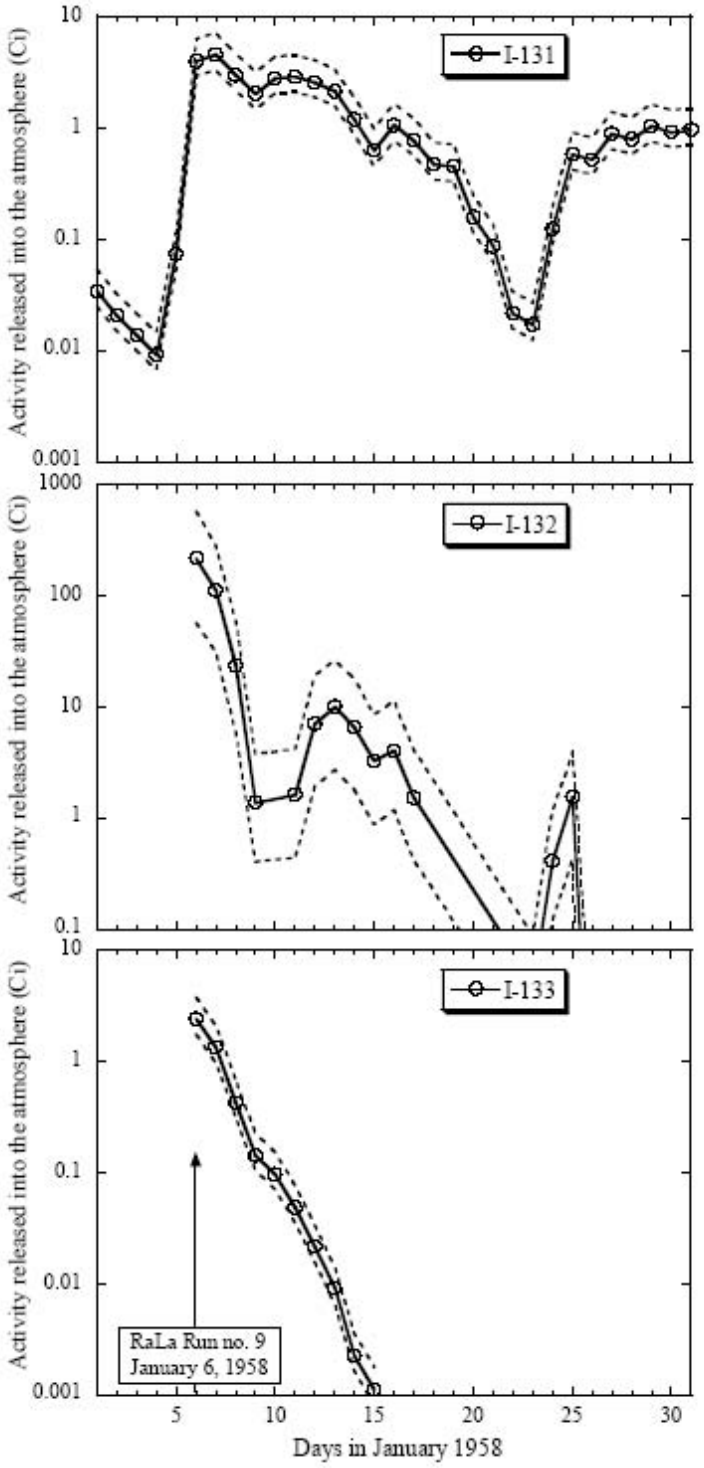

Figure 2.1 Estimated activity of iodine-131, iodine-132 and iodine-133 released daily into the atmosphere during January 1958

Dashed curves give upper and lower bounds of 95% confidence intervals.
Source: Wichner et al. 2005a.

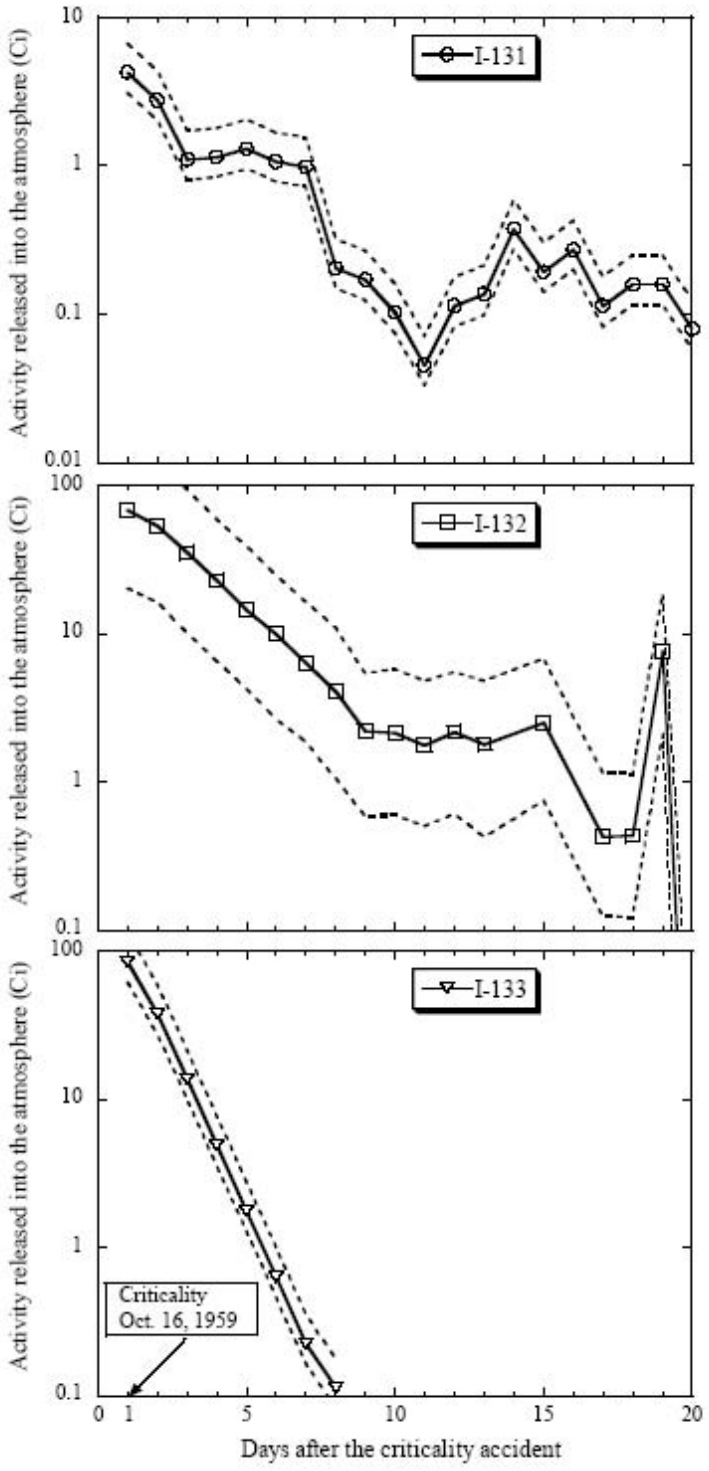

Figure 2.2 Estimated activity of iodine-131, iodine-132 and iodine-133 released into the atmosphere due to the October 16, 1959, criticality accident

Dashed curves give upper and lower bounds of 95% confidence intervals.
Source: Wichner et al. 2005a.

2.2 Releases of Radionuclides Attached to Aerosols from the Idaho Chemical Processing Plant during 1957-1959

Daily records of total beta minus iodine (β-I) activity and alpha activity provided by ICPP Stack Monitor Datasheets indicate that non-gaseous radionuclides were emitted from ICPP operations. Those radionuclides were attached to very small particles (aerosols), which were transported through the off-gas system and released to the atmosphere. Possible mechanisms of aerosol formation and release are discussed by Wichner et al. 2005b.

The ICPP off-gas flow combined four processing lines, one of which was off-gas from RaLa operations; the other three lines were off-gases from other operations at the ICPP. While it is clear that iodine came from RaLa process operations (which used short-cooled fuel), it is possible that the other three lines contributed to aerosol releases. An analysis of correlations between the β-I and α peak releases and peak releases of I-131 indicated that the main source of aerosol releases was RaLa process solutions, except during the period following the criticality accident.

Daily records contain only measurements of gross beta and gross alpha activity. Even though gamma spectrometry was becoming available during late 1950s, no record was found to indicate the radionuclide compositions of β-I readings. In the absence of direct measurements, the radionuclide composition that resulted in a β-I reading during a given day of a RaLa run was assumed to be similar to the radionuclide composition of a fuel element that was cooled to the specified day. Daily releases of 115 radionuclides attached to aerosols were estimated on the basis of the β-I measurements and the assumed radionuclide compositions. Estimated releases accounted for the efficiency of the stack-sampling device and for radioactive decay during the 24-hour sampling period and the waiting time before the sample was counted. Similarly, releases of alpha-emitting radionuclides were estimated using the gross α measurements and an assumed efficiency of the stack-sampling device. Since alpha-emitting radionuclides are long-lived, radioactive decay during the sampling period and waiting time was neglected.

The main sources of uncertainty in estimated aerosol releases were the efficiency of the liquid sampler and the reduction in activity due to deposition in the sampling line (Wichner et al. 2005b).

Of the 115 radionuclides attached to aerosols that were assumed to be released to the atmosphere from the ICPP, the screening analysis summarized in Section 3 resulted in selection of 10 β/γ-emitting radionuclides (Sr-89, Sr-90, Y-91, Zr-95, Nb-95, Ru-103, Ba-140, Ce-141, Ce-144, and Pr-143) and one α-emitting radionuclide (Pu-238) for inclusion in a dose reconstruction. Estimated releases of these radionuclides during the years 1957-1959 are presented in Table 2.2. As in the case of releases of iodine discussed previously, releases of radionuclides in aerosol form following the criticality accident are similar to releases during a normal RaLa run, and the two types of releases are combined in estimating doses to the public.

Table 2.2 Estimated atmospheric releases of radionuclides attached to aerosols from the Idaho Chemical Processing Plant during 1957-1959

Isotope	Routine Releases* (including October 16, 1959, criticality accident)		
	95% Confidence Interval (C.I.)		
	Lower bound	Central Estimate	Upper Bound
Pr-143	174 (6.4)	344 (12.7)	886 (33)
Ce-141	171 (6.3)	339 (12.5)	873 (32)
Ba-140	165 (6.1)	327 (12.1)	841 (31)
Zr-95	147 (5.5)	292 (10.8)	751 (28)
Y-91	142 (5.2)	281 (10.4)	723 (27)
Sr-89	120 (4.4)	237 (8.8)	611 (23)
Nb-95	98 (3.6)	195 (7.2)	502 (19)
Ru-103	84 (3.1)	166 (6.2)	428 (16)
Ce-144	41 (1.5)	81 (3.)	208 (7.7)
Sr-90	1.3 (0.048)	2.6 (0.095)	6.6 (0.24)
Pu-238	0.50 (0.018)	0.98 (0.036)	2.5 (0.093)
October 16, 1959, criticality accident			
Ba-140	4.9 (0.18)	9.7 (0.36)	25 (0.93)
Pr-143	3.9 (0.14)	7.7 (0.28)	20 (0.73)
Ce-141	2.3 (0.085)	4.5 (0.17)	12 (0.43)
Zr-95	1.4 (0.051)	2.7 (0.10)	7.0 (0.26)
Y-91	1.3 (0.048)	2.6 (0.10)	6.6 (0.25)
Sr-89	1.2 (0.046)	2.4 (0.091)	6.3 (0.23)
Ru-103	1.0 (0.037)	2.0 (0.073)	5.1 (0.19)
Ce-144	0.28 (0.010)	0.55 (0.020)	1.4 (0.052)
Nb-95	0.18 (0.0066)	0.35 (0.013)	0.91 (0.034)
Sr-90	0.0043 (0.00016)	0.0085 (0.00032)	0.022 (0.00081)
Pu-238†	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)

* Releases are given in Curies (Ci), and, in parenthesis, in terabecquerel (TBq). 1 Ci = 3.7×10^{10} Bq = 0.037 TBq; 1 TBq = 10^{12} Bq.

† No releases of alpha-emitters were recorded during criticality accident, until a new RaLa run started.

Source: Wichner et al. 2005b.

As in the case of releases of I-131, atmospheric dispersion calculations (Section 5) are based on the estimated daily releases of radionuclides in aerosol form. For example, Figure 2.3 shows the estimated activities of Ba-140 and Sr-90 released daily during January 1958 (RaLa Run 9). Of the 10 β/γ-emitting radionuclides of potential concern, Ba-140 has the shortest half-life (12.7 days), while Sr-90 has the longest half-life (29.2 yrs). The one alpha-emitter of potential concern (Pu-238) is long-lived (87.7 years).

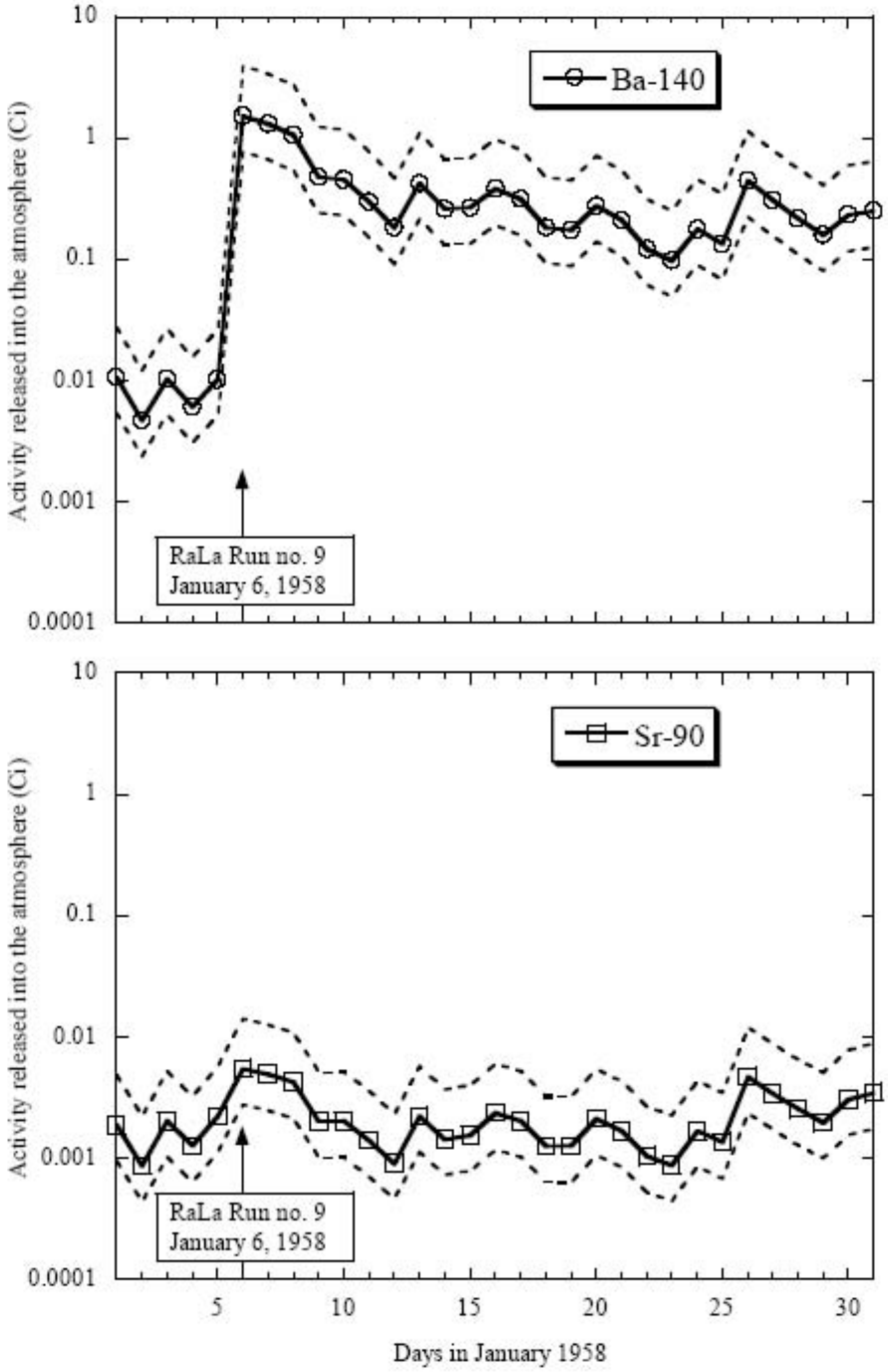

Figure 2.3 Estimated activity of barium-140 and strontium-90 released daily into the atmosphere during January 1958

Dashed curves give upper and lower bounds of 95% confidence intervals.
Source: Wichner et al. 2005b

3.0 DETERMINATION OF POTENTIALLY SIGNIFICANT RADIONUCLIDES

More than 130 radionuclides were released to the atmosphere during operations at the Idaho Chemical Processing Plant (ICPP) from 1957-1959 during the peak Radioactive Lanthanum (RaLa) operations (Wichner et al. 2005a and 2005b). However, when consideration is given to estimated releases of each radionuclide, the resulting concentrations in air at possible receptor locations, and doses per unit concentration in air for each radionuclide, it is evident that most radionuclides released from the ICPP would not have contributed significantly to doses received by people who lived near the INEL site. To promote efficient use of resources allocated to a detailed dose reconstruction for releases from the ICPP, only those radionuclides that could have contributed significantly to doses and risks to the public are included in the analysis.

In a previous report (Kocher 2005a), a simple method to screen radionuclides released to the atmosphere from the ICPP was developed. The screening methodology provides estimates of lifetime risks of cancer incidence per unit activity of radionuclides released. Risks per unit release of radionuclides are calculated on the basis of assumptions about atmospheric transport between the source and a receptor location on the INEL site boundary and assumptions about an exposure scenario, exposure pathways, and parameters for estimating dose and risk that are intended to result in substantial overestimates of actual doses and risks to people who lived near the site. By multiplying the calculated cancer risk per unit activity of a radionuclide released and an upper confidence limit of an estimated release of that radionuclide, a calculated screening risk is obtained. That risk then is compared with an assumed screening criterion, which is a lifetime risk of cancer incidence of 10^{-5}. If the calculated risk corresponding to an estimated release of a radionuclide equals or exceeds the screening criterion, that radionuclide is selected for inclusion in a dose reconstruction. Since the assumed screening criterion is a risk that is generally considered negligible and calculated screening risks should be considerable overestimates of actual risks to the public, assurance is provided that doses and risks from radionuclides that are eliminated by screening would not affect the overall results of a detailed dose reconstruction.

In a companion report (Kocher 2005b), the method of screening described above was applied to upper confidence limits of estimated releases from the ICPP developed by Wichner et al. (2005a and 2005b) and summarized in Section 2 (see Tables 2.1 and 2.2). The radionuclides listed in Table 3.1 were selected for inclusion in a dose reconstruction. Table 3.1 also indicates the importance of different exposure pathways to the calculated screening risk for each radionuclide in an assumed scenario for a largely self-sufficient homesteader used in the screening methodology (IAEA 2001).

It is important to understand the limitations of the rankings of different exposure pathways for each radionuclide in Table 3.1. Most importantly, since those rankings apply only to the different pathways of exposure to a given radionuclide, they do not provide an indication of the overall importance of a particular radionuclide and its associated pathways in a dose reconstruction that takes all radionuclides of concern into account. A particular pathway can have high importance for a given radionuclide, but that radionuclide and pathway can be unimportant to an estimated dose from all radionuclides combined. In addition, the rankings from different pathways do not necessarily indicate their relative importance in a detailed dose reconstruction for a given radionuclide released from the ICPP, because a more realistic analysis

might include parameter values that differ substantially from those assumed in the screening methodology, and some scenarios for exposure of the public do not include all exposure pathways that were assumed in screening.

Table 3.1 Radionuclides selected by screening for inclusion in dose reconstruction for releases from the Idaho Chemical Processing Plant and importance of different exposure pathways to calculated screening risks

Nuclide	Half-life	Importance of exposure pathway*				
			Milk	Meat	External‡	Inhalation
	50.5 d	High	High	Medium	Low	Low
Sr-90	29.1 y	Medium	High	Medium	Low	Low
Y-91	58.5 d	High	Low	Medium	Low	Low
Zr-95	64.0 d	High	Low	Low	High	Low
Nb-95	35.1 d	High	Low	Low	High	Low
Ru-103	39.3 d	High	Low	High	Medium	Low
I-131	8.04 d	Medium	High	Low	Low	Low
I-133	20.8 h	Low	High	Low	Low	Low
Ba-140	12.74 d	High	High	Low	Low	Low
Ce-141	32.5 d	High	Low	Low	Low	Low
Ce-144	284 d	High	Low	Low	Low	Low
Pr-143	13.6 d	High	Low	Low	Low	Low
Pu-238	87.7 y	High	Low	Low	Low	High

* Importance of different exposure pathways is evaluated on the basis of pathway models and parameters used in screening methodology to select radionuclides of concern (IAEA 2001); "High" indicates contribution to total dose from all pathways combined of greater than 40%, "Medium" indicates contribution of 10% to 40%, and "Low" indicates contribution of less than 10%. Ranking of pathways for a given radionuclide does not indicate importance of that radionuclide and associated pathways in a dose reconstruction that takes all radionuclides of concern into account.

† Includes ingestion of contaminated garden vegetables and direct ingestion of contaminated soil.

‡ Includes external exposure to atmospheric cloud and contaminated ground surface.

Source: Kocher 2005b

In general, the rankings of different exposure pathways for each radionuclide indicate that when an individual consumed substantial quantities of foods, especially garden crops and milk, that were produced near the INEL site boundary, doses from food-chain pathways should be considerably more important than doses from external exposure and inhalation. However, when an individual did not consume contaminated foods that were produced at such locations, the dose should be much lower.

In the screening analysis to select radionuclides of potential importance in releases from the ICPP during the years 1957-1959, the results indicate that I-131 is by far the most important radionuclide in exposure scenarios that include consumption of locally produced milk and other foods (Kocher 2005b). For this reason, initial efforts in the detailed dose reconstruction for the ICPP have focused on releases of I-131. Only a preliminary analysis (Section 7.2) was performed for the other radionuclides listed in Table 3-1 to allow evaluation of the merit of a more detailed analysis.

4.0 ASSUMED EXPOSURE SCENARIOS FOR MEMBERS OF THE PUBLIC

The main exposure pathways for atmospheric releases of radionuclides include inhalation, external exposure, ingestion of food items (e.g., garden crops, milk and meat), and incidental ingestion of soil (e.g., by individuals working in dusty environments). Iodine-131 contamination of vegetation is assumed to occur via direct deposition (root uptake of I-131 from contaminated soil can be neglected). Ingestion of soil by animals is included in all scenarios in which consumption of animal products is assumed.

Exposure scenarios are defined for representative individuals in the population (i.e., hypothetical individuals chosen to provide reasonably realistic representations of exposures of the public). For the purpose of analyzing the effects of emissions from the Idaho Chemical Processing Plant (ICPP), six basic exposure scenarios were defined (Apostoaei and Reed 2005; Table 4.1) on the basis of site-specific information collected from detailed documents about the INEL site and surrounding regions (USDA 1999; IDFG 2002; Bowman et al. 1984; Stacy 2000; INEL 1998, 2000, 2002a, b, c), from individuals living in the area (Shay 2002), and from members of the Idaho Health Effects Subcommittee (IHES) (Garcia 2003). The basic scenarios include scenarios for exposure beyond the INEL site boundary or exposure at onsite locations.

The assumed exposure scenarios are described briefly below, and their main characteristics are summarized in Table 4.1.

Offsite scenarios

- ***Rural Resident*** – a homesteader who produced much of his or her own food products or had access to locally produced food products. Separate analyses are performed according to the source of milk (backyard cow or backyard goat).

- ***Urban Resident*** - an individual who purchased most of his or her food from grocery stores, which obtained food from local producers and producers in the extended INEL region. Milk produced in the extended INEL region is obtained from multiple large dairies, where milk from many cows is mixed.

- ***Migrant Farm Worker*** - a farm worker who participated in seasonal farming activities and thus was present at the site for only part of the year. This scenario applies to male and female adults and their children who traveled with them at the work site. Migrant workers did not have their own animals or gardens, but their food came from local sources. For instance, any milk they consumed was most likely obtained from a local dairy farm or from a store selling locally produced foods. In some cases, the employer provided food (including milk) from his or her own garden (or backyard cow).

Table 4.1 Exposure scenarios assumed in dose reconstruction for releases from the Idaho Chemical Processing Plant

	Name	Ingestion pathway	Inhalation and external exposure	Age and gender	Location
	Offsite Exposure Scenarios				
1a	Rural resident* (backyard cow milk diet)	Backyard cow milk diet and locally produced foods†	Large fraction of time spent outside	Males and females; all ages at exposure	Selected communities
1b	Rural resident* (goat milk diet)	Goat milk diet and locally produced foods†	Large fraction of time spent outside	Males and females; all ages at exposure	Selected communities
2	Urban resident* (commercial milk diet)	Commercial milk diet and commercially available foods‡	Limited time spent outside	Males and females; all ages at exposure	Selected communities
3	Migrant farm worker	Cow milk from a local dairy farm and locally produced foods†	Very large fraction of time spent outside	Males and females; all ages at exposure	Selected communities
	Onsite Exposure Scenarios				
4	Onsite rancher	Meat from cattle or sheep grazing within INEL site boundary§	Large fraction of time spent outside	Adult male (age 20 in 1957)	Big Lost River sink area
5	Hunter	Meat from deer or pronghorn grazing within INEL site boundary§, ¶	Limited time spent outside	Adult male	INEL area
6a	One-time visitor	Not applicable	Limited time spent in the plume	Males or females; teenagers or adults	Central Facilities Area (CFA)
6b	Regular visitor	Not applicable	Limited time spent in the plume	Adult male (age 20 in 1957)	CFA

* These terms refer to a lifestyle (i.e., source of food), not to a place of residence in a rural or urban area.

† Includes locally grown vegetables and locally produced meats.

‡ Doses are calculated for both local and regional food products.

§ This person could have been exposed as a rural resident at one of the locations around the INEL site. The doses from the exposure as a rural resident are calculated separately.

¶ This scenario assumes that hunting took place outside the INEL boundary, because hunting was not permitted within the INEL site boundary during 1957-1959, but the killed animals grazed on the INEL property before roaming into areas where hunting was permitted.

An important aspect of the rural and urban offsite exposure scenarios is that these scenarios are representations of a lifestyle, not of a place of residence (i.e., rural or urban location). For example, consider the case of a farmer who cultivated only wheat, but had no vegetable garden and did not own cows. Because such a person relied on the commercial food products, doses from I-131 would be estimated in accordance with the exposure scenario for an urban resident.

Conversely, a person who lived in Idaho Falls but had a vegetable garden and owned a dairy cow is considered to have had a rural lifestyle, even though he or she lived in a city.

A detailed analysis of the doses to the public from I-131 was performed for each offsite exposure scenario at the location of many cities and towns around INEL (Table 4.2; Figure 4.1). That is, doses from I-131 were estimated for representative individuals living in each city or town, assuming a rural, urban, or migrant worker lifestyle.

The importance of radionuclides other than I-131 in the offsite exposure scenarios is assessed in Appendix C.2 and summarized in Section 7.2. A detailed evaluation of doses from exposure to other radionuclides was not performed, but upper-bound estimates of doses are obtained to allow an evaluation of the merit of a more detailed analysis.

Onsite scenarios

- ***Onsite Rancher*** – a rancher who is assumed to raise beef cattle or sheep in the Big and Little Lost River sink area, which is located 16-24 km (10-15 miles) north-northeast of the ICPP (see Figure 4.2). This area within the INEL site has been open to controlled grazing. Such a person is assumed to come onsite to take care of livestock for a limited amount of time each week (e.g., about 2 days per week), and to consume meat obtained from animals grazing onsite.

- ***Hunter*** - an adult who consumed the whole edible tissue on an animal that grazed within the INEL site boundary. The hunter scenario can be considered representative of Native Americans who relied more heavily on hunting as a food source in the 1950s. Even though the INEL site was a grazing area for deer and prong-horned antelope, hunting was not permitted inside the INEL site boundary during 1957-1959. This scenario assumes that hunting took place outside the INEL boundary, and that the killed animals grazed on the INEL property before roaming into areas where hunting was permitted.

- ***Visitor*** - occasional visitors (e.g., a group of students taking a site tour) or outside workers who visit one of the INEL facilities regularly as part of their job (e.g., a delivery person). An occasional (one-time) visitor is assumed to take a 1-day trip to the Central Facilities Area (CFA) where the average air concentrations of radionuclides are the highest. An outside worker is assumed to be a regular visitor present for a few hours each week at the CFA. The relevant exposure pathways are inhalation and external exposure to atmospheric plume and contaminated ground surface.

Individuals who are included in the onsite exposure scenarios also could have been exposed as rural or urban residents beyond the site boundary (as described in the offsite exposure scenarios). However, estimated doses for onsite scenarios do not include additional exposures that could have occurred at locations beyond the site boundary. Thus, the total dose to an individual who was exposed onsite could be the sum of the doses estimated in the two different scenarios.

Table 4.2 Locations at which doses to members of the public are estimated

	Community or town*	Population in 2000	Locations included in the DOE (1991) report†
	Offsite Locations		
1	Aberdeen Junction	1840	
2	Arco	1026	Arco
3	Atomic City	25	Atomic City
4	Basalt	419	
5	Blackfoot	10,419	Blackfoot
6	Butte City	76	Butte City
7	Dubois	647	
8	Firth	408	
9	Fort Hall	3193	
10	Grandview	470	
11	Hamer	12	
12	Howe	20	Howe
13	Idaho Falls	50,730	Idaho Falls
14	Lewisville	467	
15	Lost River	26	
16	Mackay	566	
17	Menan	707	
18	Moore	196	
19	Monteview	10	Monteview
20	Mud Lake	270	Mud Lake
21	Roberts	647	Roberts
22	Shelley	3813	
23	Spencer	38	
24	Terreton	1537	
	Onsite Locations		
25	CFA @ 250-ft Met Tower location		
26	Big and Little Lost River sink area - 12 km (8 miles) east of Howe.		
27	Average over the INEL area		

* Selected locations are within a 50-mile radius of the ICPP (Figure 4.1).

† Locations are listed to emphasize differences between this study and DOE (1991) study.

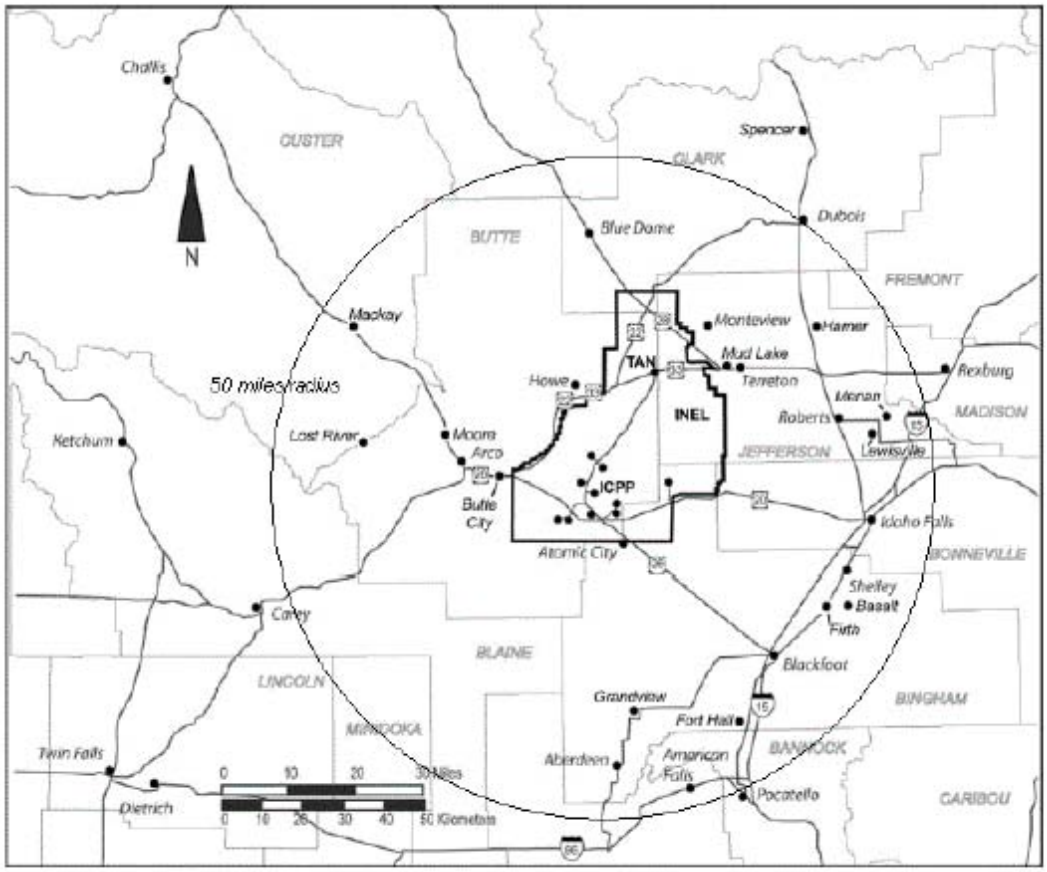

Figure 4.1 Region surrounding Idaho National Engineering Laboratory site
(Adapted from INEL 2002c.)

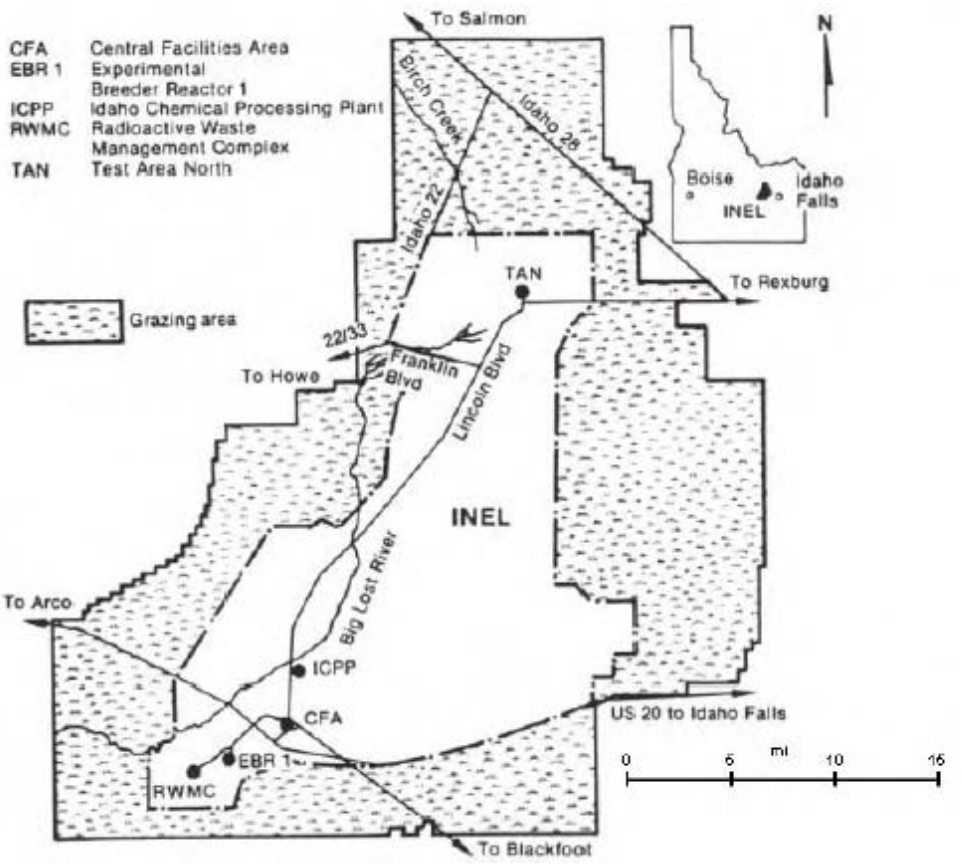

Figure 4.2 Permit grazing area for beef cattle and sheep at the Idaho National Engineering Laboratory

The location with the largest number of animals was Big and Little Lost River sink area, near Howe.
Source: Bowman et al. 1984.

The onsite exposure scenarios are presumably less important than scenarios for offsite exposure even though airborne concentrations of radionuclides were higher at onsite locations, because they include fewer exposure pathways and exposure durations are lower. The importance of the assumed onsite exposure scenarios is assessed in Appendix C.3 and summarized in Section 7.3. A detailed evaluation of doses from these exposure scenarios was not performed, but upper-bound estimates of doses were obtained to allow evaluation of the merit of a more detailed analysis.

5.0 ATMOSPHERIC TRANSPORT OF RELEASED RADIONUCLIDES

For radionuclides released from the Idaho Chemical Processing Plant (ICPP), concentrations in air at the selected offsite and onsite locations around the INEL were estimated using the CALPUFF modeling system (Radonjic et al. 2005). Meteorological data collected within the INEL boundary during the years of release were used as an input to the calculations.

This section describes the CALPUFF modeling system, the available meteorological data, and a validation exercise performed to assess the reliability of the CALPUFF model. A summary of concentrations in air estimated using CALPUFF is given in Section 5.3.

5.1 CALPUFF Modeling System

CALPUFF is a non-steady-state air quality modeling system developed by Sigma Research Corporation (now part of Earth Tech, Inc., of Concord, Massachusetts; Scire et al. 1999). The original development of the CALMET/CALPUFF models was sponsored by the California Air Resources Board and is now supported by the U.S. Environmental Protection Agency. The CALPUFF modeling system consists of the following three sub-systems:

(1) A meteorological modeling package (CALMET) with both diagnostic and prognostic wind field generators

(2) A Gaussian puff dispersion model (CALPUFF) that incorporates chemical removal, wet and dry deposition, complex terrain algorithms, building downwash, plume fumigation, and other effects

(3) Post-processing programs (CALPOST) to generate output fields of meteorological data, airborne concentrations, and deposition fluxes

The CALMET model includes a diagnostic wind field generator that contains parameterized treatments of slope flows, kinematic terrain effects, terrain blocking effects, a divergence minimization procedure, and a micrometeorological model for overland and over-water boundary layers. CALMET develops hourly wind and temperature fields on a two- or three-dimensional grid domain, and it incorporates the effects of terrain on wind flow. The wind field can be developed in CALMET using observations from several meteorological monitoring stations in the vicinity of the source of emissions.

CALPUFF is a dispersion model that transports a discrete number of packets of pollutant material (i.e., “puffs”) as a representation of a continuous plume, and simulates dispersion and transformation processes along the way. Temporal and spatial variations in the meteorological fields derived from CALMET are explicitly incorporated in the resulting distribution of puffs throughout the simulation period.

5.2 Approach to Atmospheric Dispersion Modeling

5.2.1 Meteorological Data

Kirk L. Clawson, Deputy Director for the National Oceanic and Atmospheric Administration (NOAA) in Idaho Falls (Clawson 2002), provided meteorological data for 1957-1959 obtained from two weather stations located within the INEL site boundary. Hourly data are reported for wind speed, wind direction, and temperature. The more complete data are from the weather station at the Central Facilities Area (CFA), which is near the ICPP (known as the South station). The second (North) station is for Test Area North (TAN), which is the site of the initial engine tests in the Aircraft Nuclear Propulsion Program. The data were collected at heights above ground of 20 ft and 250 ft at the South station and 20 ft and 150 ft at the North station. In addition to these two stations, meteorological data for the years of interest were obtained at Pocatello and Idaho Falls, and upper atmosphere data were obtained from stations at Boise, Lander, and Salt Lake City.

More detailed meteorological data were found for the night of October 16, 1959, when the criticality accident occurred. Data at the South station are reported by Ginkel et al. 1960 (Table IV, page 40), and they include 10-minute average wind speed and direction at 20 ft and 250 ft for 4 hours after the accident.

A complete set of meteorological data for 1999 was obtained from NOAA (Clawson 2002) for 25 stations in and around the INEL site. The data contain hourly observations (when available) of mean wind speed, mean (vector) wind direction, mean temperature, and total precipitation. These data were used to calibrate and validate the CALPUFF/CALMET system before it was applied to model atmospheric transport during 1957-1959, as described in the next section. A comparison of the wind roses in 1999 and 1958 at the Central Facilities Area (CFA) meteorological station is shown in Figure 5.1.

5.2.2 Modeling Details

A 320 × 320 km modeling domain with a grid spacing of 4 km was chosen for this analysis. Airborne concentrations of radionuclides were calculated at each node of the grid and each discrete location listed in Table 4.1.

The CALMET model was used to develop two sets of hourly wind fields for the year 1999. One wind field was obtained using data from all stations in the dense network of stations available in 1999. The second wind field for 1999 was obtained using meteorological data only for the same stations that were in existence in the late 1950s. Using CALPUFF, two sets of air concentrations (per unit release) were estimated for the two sets of wind fields for 1999.

The differences between the two sets of air concentrations for 1999 were used to develop a multiplicative uncertain bias correction factor, which was applied to single-valued air concentrations that were predicted for the 1957-1959 period on the basis of contemporary meteorological data from the limited number of stations (Radonjic et al. 2005).

In addition to uncertainties in the wind field, estimated air concentrations accounted for the following sources of uncertainty:

(a) Uncertainty in emission rates of radionuclides (see Tables 2.1 and 2.2)

(b) Model uncertainty related to calculating
- vertical wind profiles (in CALMET),
- terrain adjustment methods (in CALPUFF)
- atmospheric dispersion coefficients (in CALPUFF)

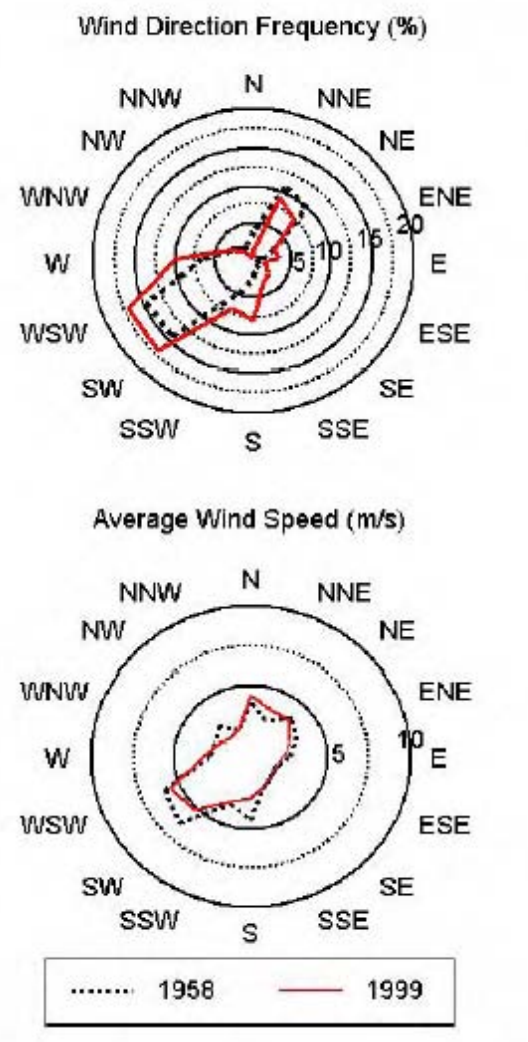

Figure 5.1 Wind speed and wind frequency recorded at Central Facilities Area Meteorological Station in 1958 and 1999
Source: Clawson 2002.

5.2.3 Validation of CALPUFF Modeling System

To test the validity of estimates of airborne concentrations of radionuclides, CALPUFF was evaluated using data collected in 1999 as part of an atmospheric tracer experiment. In that experiment, sulfur hexafluoride (SF_6) was released from a 21-m stack in the INEL site in April and May 1999 during six 4-hour tests and one 2-hour test. Two mobile real-time SF_6 detectors were placed in vans and deployed during each test. The SF_6 air concentrations were measured using detectors in the vans and detectors at fixed locations along three sampling arcs located 15 to 50 km northeast of the release (i.e., downwind). The number of sampling points per test varied from 520 to 12,478, and 54 sets of measurements were generated. The complete set of data, including release information, meteorological information, and measured air concentrations, was provided by Dr. Kirk Clawson of NOAA, one of the organizers of the experiment (Clawson 2003).

The CALPUFF model was used to predict air concentrations summarized on an hourly basis, using the known release rates and release conditions (stack height, exit velocities, etc.), as well as available meteorological information (Radonjic et al. 2005). Out of the 54 sets of hourly predictions, 60% were within a factor of 2 of the observed concentrations, with an overall correlation coefficient between the measurements and the hourly predictions of 0.77.

In this study, doses are estimated on the basis of monthly average air concentrations (Section 6), not hourly average air concentrations. This means that it is more important to understand the reliability of CALPUFF in predicting long-term average air concentrations. In the validation exercise (Radonjic et al. 2005), the CALPUFF prediction of the average for all hours of the SF_6 tests was within 20% of the measured average for all hours. These results indicate that the CALPUFF system is a reasonably reliable predictor of the time-averaged air concentrations for the INEL stie.

5.3 Summary of Estimated Concentrations of Iodine-131 in Air

The highest annual-average concentration of I-131 was estimated for 1957, reflecting the higher level of releases that occurred in that year. In 1958 and 1959, releases were progressively lower, as shown by the annual-average air concentrations (Figure 5.2). The highest average concentrations of I-131 in air over the period of release occurred in the Mud Lake/Terreton area located northeast of the ICPP (Figure 5.3, Figure 4.1, Table 5.1). Even though these locations are not the closest to the ICPP, the wind frequency is the highest in that direction. The highest upper bound of the estimated airborne concentration of I-131 was obtained in Atomic City, which is the location closest to the ICPP (19 km = 11.8 miles southeast of the ICPP).

Concentrations of I-131 in air at each of the selected locations were estimated on an hourly basis. In performing the detailed dose reconstruction, hourly air concentrations have been averaged on a monthly basis, which is a period short enough to observe seasonal variations, but long enough to assume equilibrium in I-131 transfers between different environmental media. An example of monthly air concentrations in Atomic City is presented in Figure 5.4. The temporal pattern of the air concentration expressed on a monthly basis is consistent with the dates of individual RaLa runs.

When released from the ICPP stack, iodine is almost entirely in elemental form (i.e., I_2). This form of iodine is highly reactive chemically. During transport through the atmosphere, elemental iodine attaches to small atmospheric particles and also interacts with other chemical elements in air to form more stable organic compounds. Ludwick (1964, 1967) determined experimentally that beyond a distance of about 3 km downwind, the proportions of elemental, particulate, and organic iodine are roughly equal. That is, the concentration of I-131 in air is comprised of about one-third elemental iodine, one-third particulate iodine, and one-third organic iodine. The air concentrations presented in Table 5.1 and Figures 5.2 through 5.4 represent the activity of all physico-chemical forms of I-131. However, in estimating doses from I-131, the concentration of each physico-chemical form of iodine is calculated at each location of interest. It is important to account for the different physico-chemical forms of iodine, because they have markedly different deposition rates to vegetation (Appendix A).

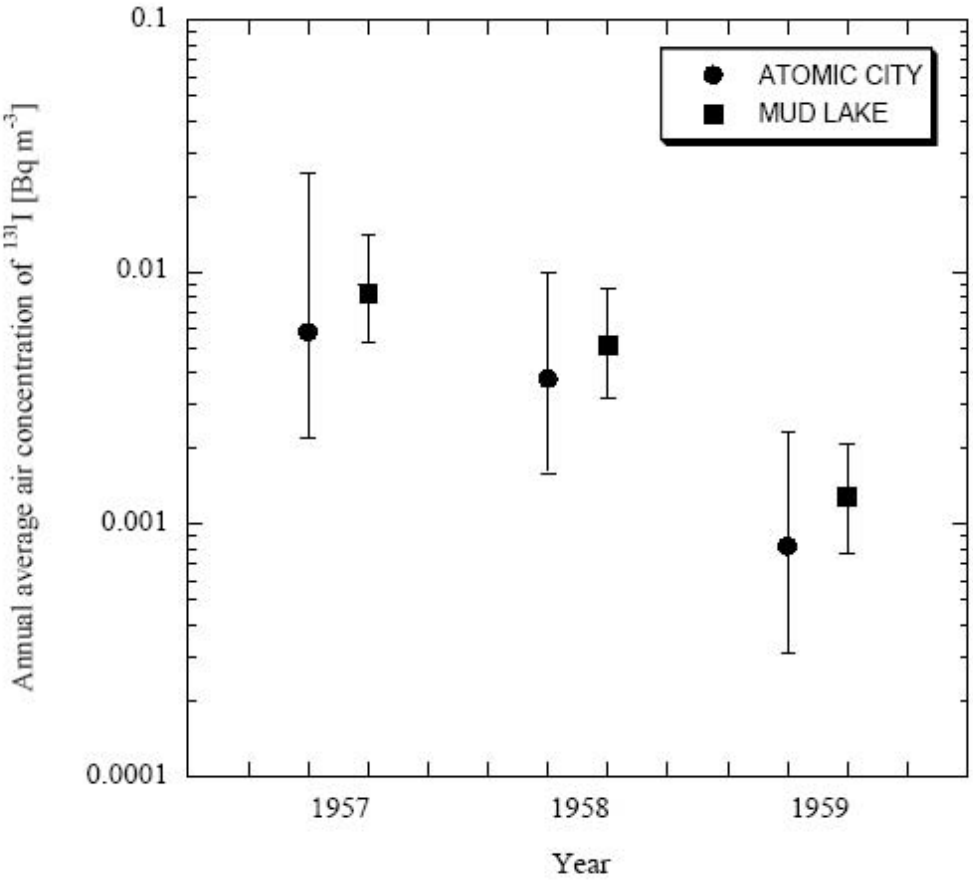

Figure 5.2 Estimated annual average concentrations of iodine-131 at two selected locations
The vertical lines represent 95% confidence intervals.

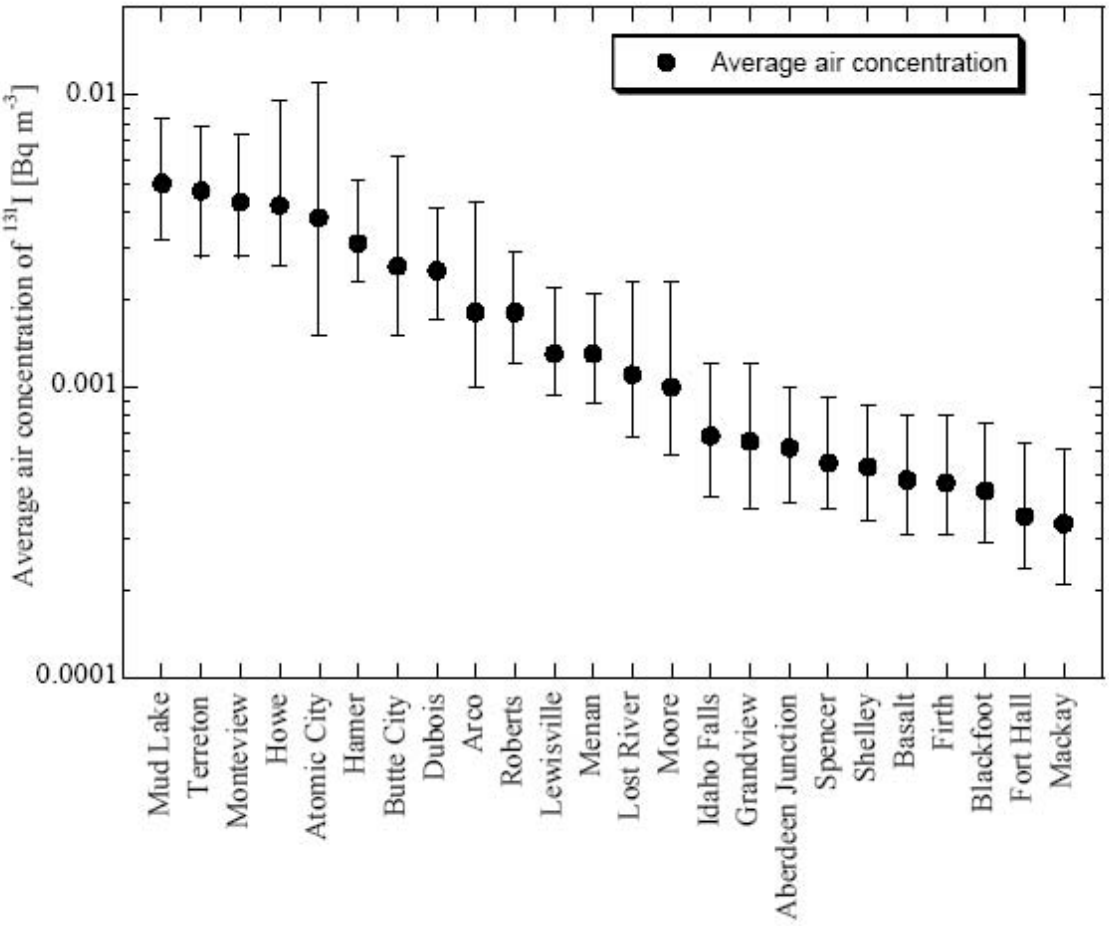

Figure 5.3 Estimated concentrations of iodine-131 in air averaged over the period of release (February 1957 – December 1959) at different offsite locations

The vertical lines represent 95% confidence intervals.

Table 5.1 Estimated concentrations of iodine-131 in air averaged over the period of release (February 1957 – December 1959)

Offsite Location	I-131concentration in air [Bq m^{-3}] 95% confidence interval		
	Lower bound	**Central estimate***	**Upper bound**
Aberdeen Junction	4.0×10^{-4}	6.2×10^{-4}	1.0×10^{-3}
Arco	1.0×10^{-3}	1.8×10^{-3}	4.3×10^{-3}
Atomic City	1.5×10^{-3}	3.8×10^{-3}	1.1×10^{-2}
Basalt	3.1×10^{-4}	4.8×10^{-4}	8.0×10^{-4}
Blackfoot	2.9×10^{-4}	4.4×10^{-4}	7.5×10^{-4}
Butte City	1.5×10^{-3}	2.6×10^{-3}	6.2×10^{-3}
Dubois	1.7×10^{-3}	2.5×10^{-3}	4.1×10^{-3}
Firth	3.1×10^{-4}	4.7×10^{-4}	8.0×10^{-4}
Fort Hall	2.4×10^{-4}	3.6×10^{-4}	6.4×10^{-4}
Grandview	3.8×10^{-4}	6.5×10^{-4}	1.2×10^{-3}
Hamer	2.3×10^{-3}	3.1×10^{-3}	5.1×10^{-3}
Howe	2.6×10^{-3}	4.2×10^{-3}	9.6×10^{-3}
Idaho Falls	4.2×10^{-4}	6.8×10^{-4}	1.2×10^{-3}
Lewisville	9.4×10^{-4}	1.3×10^{-3}	2.2×10^{-3}
Lost River	6.8×10^{-4}	1.1×10^{-3}	2.3×10^{-3}
Mackay	2.1×10^{-4}	3.4×10^{-4}	6.2×10^{-4}
Menan	8.8×10^{-4}	1.3×10^{-3}	2.1×10^{-3}
Moore	5.8×10^{-4}	1.0×10^{-3}	2.3×10^{-3}
Monteview	2.8×10^{-3}	4.3×10^{-3}	7.4×10^{-3}
Mud Lake	3.2×10^{-3}	5.0×10^{-3}	8.3×10^{-3}
Roberts	1.2×10^{-3}	1.8×10^{-3}	2.9×10^{-3}
Shelley	3.5×10^{-4}	5.3×10^{-4}	8.6×10^{-4}
Spencer	3.8×10^{-4}	5.5×10^{-4}	9.2×10^{-4}
Terreton	2.8×10^{-3}	4.7×10^{-3}	7.8×10^{-3}
Onsite Location	**Lower bound**	**Central estimate***	**Upper bound**
CFA @ 250-ft Met Tower	1.6×10^{-2}	2.6×10^{-2}	4.9×10^{-2}
Big Lost River Sink Area	4.3×10^{-3}	7.1×10^{-3}	1.5×10^{-2}
Average over INEL area	3.7×10^{-3}	7.5×10^{-3}	1.3×10^{-2}
Average over milk production area	4.7×10^{-4}	6.7×10^{-4}	1.1×10^{-3}

* 50^{th} percentile of probability distribution function describing the uncertainty in estimated concentrations of I-131 in air.

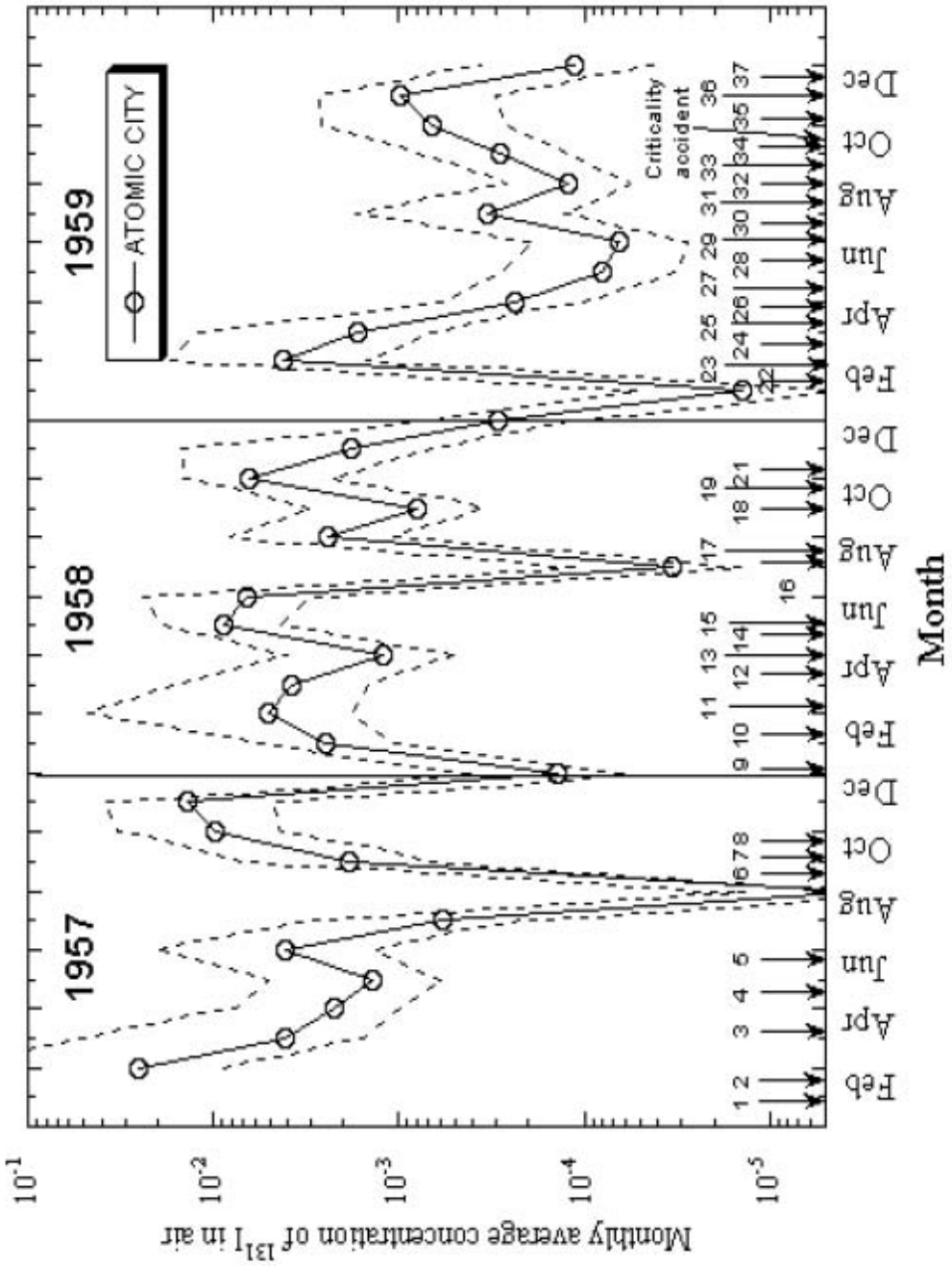

Figure 5.4 Estimated concentrations of iodine-131 in air at Atomic City, Idaho, due to releases from the Idaho Chemical Processing Plant operations

The numbers represent each of 37 RaLa runs, and the arrows indicate the approximate date of each run. The dashed lines represent 95% confidence intervals

6.0 METHODS OF ESTIMATING DOSES TO MEMBERS OF THE PUBLIC

This section briefly describes the main features of the approach used to estimate doses to members of the public from exposure to I-131 released from the Idaho Chemical Processing Plant (ICPP). A detailed description of the modeling approach, including equations and parameter values, is given in Appendix A.

Releases of I-131 from a Radioactive Lanthanum (RaLa) run lasted for many days after fuel elements were first dissolved, in most instances until the next RaLa run. Thus RaLa operations can be considered to be an almost continuous source of I-131 to the atmosphere. Because releases following the criticality accident are similar in magnitude to releases during a normal RaLa run and they also occurred over a number of days, the criticality accident can be considered in the same way as a routine RaLa run from the point of view of estimating doses to the public. In this report, results are presented for all releases from the ICPP during the years 1957-1959, including releases following the criticality accident.

Modeling of transport of I-131 in the environment begins with the estimated I-131 concentrations in air (Bq m^{-3}) at selected locations (Table 5.1). To account for seasonal effects in the environmental transfer parameters, monthly average air concentrations were used as the starting point in our calculations (e.g., Figure 5.4).

The first step in the calculation is to estimate the activity of I-131 deposited on vegetation and soil during the passage of the plume over a selected location. To develop the governing equations for transfer of I-131 from air to vegetation and soil, the following assumptions were made on the basis of our current state of knowledge.

- Three physico-chemical forms of iodine are considered to be present in air, each of them having different deposition rates onto soil and vegetation:
  - a highly reactive form, assumed to be molecular or elemental iodine (I_2)
  - a particulate form, which accounts for iodine attached to atmospheric aerosols
  - a nonreactive form, assumed to be organic iodine (e.g., CH_3I)

- Iodine is deposited on vegetation by both dry and wet deposition processes. Dry deposition refers to the process of direct removal of iodine from air by collection on soil, vegetation, or water surfaces. Wet deposition refers to the transfer of iodine from air to land or water surfaces by various types of precipitation (i.e., rain and snow).

- Dry deposition occurs at all times, including periods of precipitation.

- Uptake of I-131 from soil by plants is negligible when compared to direct deposition from the atmosphere, due to dilution from mixing of deposited activity with soil in the root zone, and due to radioactive decay during the period of time between deposition on the soil surface and uptake by plants.

Estimated concentrations of I-131 in vegetation refer to concentrations in pasture grass and on fresh leafy vegetables. Due to the short half-life of I-131 (8.04 days), I-131 concentrations in

grains or fruits were not calculated because these crops are harvested only once a year and they are usually stored for a long time before they are consumed, thus allowing I-131 to decay.

Concentrations of I-131 in soil (Bq $kg^{-1}_{dry\ soil}$) and on pasture grass (Bq $kg^{-1}_{dry\ vegetation}$) were used to estimate intakes of I-131 by dairy cows or goats, beef cattle, and chickens. The modeling approach differentiates between commercially managed cows, which are kept in relatively large herds, with milk (or meat) sold commercially, and a "backyard cow," which is raised by its owner primarily for at-home consumption of dairy products. Differences between the two types of dairy cows involve the amount of fresh pasture grass consumed and amount of milk produced.

Concentrations of I-131 in milk (Bq L^{-1}), beef (Bq kg^{-1}) and eggs (Bq kg^{-1}) were estimated using measured transfer coefficients from feed to the food product of interest. The modeling approach distinguishes between locally produced commercial milk and regional commercial milk. Locally produced commercial milk refers to milk collected from cows raised at a given location (e.g., a creamery that collects milk from a given area). Regional commercial milk represents milk purchased from a store that receives milk from many dairy farms. Doses from consumption of regional milk are estimated on the basis of I-131 concentrations in milk averaged over an entire milk production area. Around INEL, most of the large commercial dairy farms are located relatively close to the Snake River, while smaller dairy farms operate further from the river (and closer to INEL) at locations where water is available. As with regional commercial milk, references to regional commercial beef or eggs indicate that these foods are purchased from a store and contain I-131 concentrations averaged over the production area.

Exposure of the public to I-131 occurs by ingestion of contaminated food items and by inhalation of contaminated air. The doses from external exposure to I-131 in air or on the ground are negligible compared with the doses from inhalation or ingestion and are ignored in this study. Ingestion exposure pathways account for the consumption of milk, beef, fresh leafy vegetables, and eggs. Inhalation doses are estimated using the predicted air concentrations at each location. As described in Section 4, each exposure scenario incorporates different assumptions about the source of contaminated food items and the amounts consumed. For example, a resident with a rural lifestyle is assumed to consume only locally produced food items. That is, he or she owns a backyard cow used as source of fresh milk, owns chickens, and has a vegetable garden. Beef and other meats also come from a local farm. In contrast, a resident with an urban lifestyle purchases most of his or her food products from grocery stores, which sells food from the extended INEL region. Milk is obtained from multiple large dairies where milk from many cows is mixed. Migrant farm workers could have obtained food either from local sources (i.e., like a rural resident) or from a store. However, they are physically present in the region at most from April through November.

Doses from inhalation of I-131 depend on the type of activity performed by each individual. An urban resident is assumed to spend limited amounts of time outside. A rural resident spends more time outdoors than an urban resident. A migrant farm worker is assumed to spend the greatest amount of time outdoors.

Once taken up by the human body by ingestion or inhalation, iodine accumulates in the thyroid gland. Iodine is used by the human body to label thyroid hormones, which are produced and stored for tens of days in the thyroid gland. Thus, ingestion and inhalation of I-131 result in

irradiation of the thyroid gland, mainly by beta particles emitted in decay of I-131, while other organs of the body receive negligible doses from emitted photons. Radiation doses estimated in this study are doses to the thyroid gland and effective doses (as defined by ICRP 1991). The effective dose allows comparisons of exposures to radionuclides that irradiate different organs.

Since 1957 was the first year of releases from the ICPP, doses are estimated for individuals who were infants or were 5, 10, 15, or 25 years old in 1957. Gender differences in estimated doses are observed for individuals over 15 years old at the time of exposure, so only doses for adults are reported for both males and females. In the case of infants, exposures are estimated assuming different levels of breastfeeding and different diets for the mother.

Uncertainties in estimated releases of I-131 and in atmospheric transport are described by Wichner et al. (2005a) and Radonjic et al. (2005), respectively (see also Section 2.1 and Section 5). Uncertainty in estimated doses is quantified by propagation of uncertainties through the environmental transport model, using 500-iteration mid-point Latin Hypercube samples (LHS). Each parameter used in the environmental transport model is described by a probability distribution function (see Appendix A), which represents our state of knowledge (Apostoaei et al. 1999, NCI 1997, NCRP 1996). If available, site-specific data were used to determine distribution parameter values. For instance, an extensive set of iodine experiments was performed at the INEL in the 1960s (i.e., the Controlled Environmental Radioiodine Tests – CERT; Bunch 1966 and 1968; Hawley et al. 1964). Elemental and organic iodine were released under controlled conditions on the INEL site and parameters such as the deposition velocity on pasture grass and feed-to-milk transfer coefficients were determined during all seasons of the year. The results of those experiments provide the most representative data for the present study, and they were used to the extent possible. Similarly, precipitation data for 1957-1959 recorded at the meteorological stations around INEL and reported by the National Climatic Data Center were used to model wet deposition of iodine.

A special computer code was developed for the purpose of analyzing the impact of radionuclide releases from the ICPP. The code was implemented using Analytica® programming software. Concentrations in different food products are provided as output and the code allows personalized estimation of doses. For instance, the code can estimate doses for any age at exposure and either gender (i.e., the user can enter any date of birth before 1957 and can specify the gender). The present report provides a sample of all possible results by showing doses for a limited number of ages at exposure. Similarly, the code calculates doses at any location listed in Table 4.2, and the present report summarizes the results at a selected number of locations. The computer code also gives the uncertainty in each estimated dose and allows the user to choose various ways to present uncertainties (e.g., probability density functions, cumulative distribution functions, different confidence intervals, and other statistics).

7.0 RESULTS AND DISCUSSION

This section summarizes the results of the detailed reconstruction of doses from exposure to I-131 released from the Idaho Chemical Processing Plant (ICPP). The releases of I-131 and other radionuclides are described by Wichner et al. (2005a, 2005b) and are summarized in Section 2 of this report. Results of the atmospheric transport modeling are given by Radonjic et al. (2005), and a summary of estimated concentrations of I-131 in air is presented in Section 5.3. Estimated concentrations of I-131 in food products are provided in Appendix B.

Doses from I-131 presented in this section are based on the offsite exposure scenarios described on Section 4. The importance of radionuclides other than I-131 in scenarios for offsite exposure is discussed in Section 7.2, and the importance of scenarios involving assumed onsite exposures of ranchers, hunters, and visiting members of the public is discussed in Section 7.3.

Unless otherwise specified, radiation doses presented in Section 7.1 represent exposures to all I-131 releases from the ICPP from February 1, 1957, through December 1959, including the October 16, 1959, criticality accident. Since I-131 accumulates in the thyroid gland, Section 7.1 reports thyroid doses. Effective doses can be estimated by multiplying the given thyroid doses by a tissue weighting factor (w_T) of 0.05 (ICRP 1991).[4] All doses presented in this report are given in cSv (1 cSv = 1 rem = 0.01 Sv).

Propagation of uncertainties is an integral part of the computations, so the results summarized in this report are given as the 50th percentile (i.e., central value) and as the 95% confidence interval of probability distribution describing the uncertainty in the estimated quantity (i.e., concentration or dose). The 95% confidence interval represents a "credibility" interval, which can be said with a high degree of confidence (i.e., subjective degree of belief) to contain the true but unknown value.

7.1 Estimated Radiation Doses from Offsite Exposure to Iodine-131

Estimated radiation doses from exposure to I-131 at offsite locations are the result of combining predicted concentrations of I-131 in air and in food products with the estimate of losses due to food storage and food processing, assumptions about dietary intake of foods and inhalation rates, and dose coefficients for ingestion and inhalation (i.e., doses per unit activity intake) for various ages at time of exposure. The following sections summarize results of the dose assessment for rural resident, urban resident, and migrant worker scenarios described in Section 4.

7.1.1 Rural Resident Scenario

In this study, a rural resident is defined as a homesteader who grows much of his or her own food or has access to locally grown foods, without regard for whether the place of residence is in a rural or urban location. Given the known importance of the milk ingestion pathway for exposure to I-131 (Apostoaei et al. 1999, NCI 1997), separate analyses are performed for two sources of

[4] The effective dose from an intake of I-131 is due almost entirely to the dose to the thyroid, and contributions from irradiation of other organs or tissues are negligible (ICRP 1993).

milk; a backyard cow or a backyard goat. A "backyard" cow or goat refers to an animal raised by its owner primarily for at-home consumption of dairy products.

Thyroid doses from I-131 were estimated using a set of milk consumption categories. Doses are reported for a person consuming the following:

- No milk
- Zero to two 8-oz glasses of milk per day (low consumption)[5]
- One to three 8-oz glasses per day (average consumption)
- Two to four 8-oz glasses per day (high consumption)
- Three to nine 8-oz glasses per day (very high consumption)

The high consumption rate (2 to 4 glasses/day) is considered normal for a rural resident who owns a dairy cow or goat, and who therefore has easy access to an abundant source of food.

To analyze how exposure to I-131 varied by location around INEL, thyroid doses for a one-year-old child consuming a high amount of backyard cow milk were calculated (Table 7-1). Estimated doses are total doses from all I-131 releases from the ICPP during the period February 1, 1957, through December 1959, including the October 16, 1959, criticality accident. The highest doses were calculated at Mud Lake, Idaho, while the lowest doses were at Fort Hall, Idaho. In general, the locations of interest can be divided in three broad categories of thyroid doses:

- A highest-dose group (i.e., doses larger than 0.3 cSv): Mud Lake, Terreton, Howe, Monteview, Atomic City, Hamer, and Butte City

- A medium-dose group (i.e., doses between 0.1 and 0.3 cSv): Dubois, Arco, Roberts, Lost River, Moore, Menan, and Lewisville

- A lowest-dose group (i.e., doses less than 0.1 cSv): Aberdeen Junction, Idaho Falls, Grandview, Spencer, Shelley, Firth, Basalt, Mackay, Fort Hall

[5] A uniform distribution was used to describe the uncertainty in a milk consumption rate (see Appendix A); that is, any milk consumption rate in the assumed range was assigned equal probability. The same comment applies to all milk consumption categories.

Table 7.1 Estimated thyroid doses in rural resident scenario for a child born in 1956 who consumed large amounts of milk from a backyard cow

	Community or town	Population in 2000†	Thyroid dose (cSv)* 95% confidence interval		
			Lower bound	Central estimate†	Upper bound
1	Aberdeen Junction	1840	0.013	0.067	0.47
2	Arco‡	1026	0.037	0.22	1.6
3	Atomic City‡	25	0.068	0.39	3.2
4	Basalt	419	0.0090	0.051	0.32
5	Blackfoot‡	10,419	0.0083	0.048	0.31
6	Butte City‡	76	0.049	0.32	2.3
7	Dubois	647	0.048	0.27	1.8
8	Firth	408	0.0094	0.049	0.31
9	Fort Hall	3193	0.0068	0.035	0.25
10	Grandview	470	0.012	0.069	0.42
11	Hamer	12	0.062	0.33	2.2
12	Howe‡	20§	0.087	0.52	3.7
13	Idaho Falls‡	50,730	0.011	0.066	0.43
14	Lewisville	467	0.022	0.12	0.80
15	Lost River	26	0.023	0.14	1.1
16	Mackay	566	0.0068	0.041	0.27
17	Menan	707	0.021	0.12	0.75
18	Moore	196	0.018	0.12	0.84
19	Monteview‡	10§	0.086	0.48	3.1
20	Mud Lake‡	270	0.11	0.56	3.7
21	Roberts‡	647	0.031	0.17	1.2
22	Shelley	3813	0.010	0.053	0.36
23	Spencer	38	0.011	0.058	0.37
24	Terreton	1537	0.10	0.51	3.4

* Doses were obtained assuming a milk consumption rate of 2 to 4 glasses/day.

† 50th percentile of the probability distribution function describing the uncertainty in the estimated doses.

‡ Locations included in the DOE 1991 dose reconstruction for INEL.

§ Source: INEL 2002c.

Based on the 2000 population census, about 93% of the population near INEL lives in the lowest-dose group of communities, about 4% in the medium-dose group, and about 3% in the highest-dose group. The cities with the largest population in each of these groups are Idaho Falls, Arco, and, Terreton, respectively. According to the population size and magnitude of the doses, a representative city was chosen for each of the three dose groups: Terreton for the highest-dose group, Roberts for the medium-dose group, and Idaho Falls for the low-dose group. The remainder of Section 7.1 presents doses estimated for people who lived in those cities.

The communities included in this analysis (Table 4.2) are located in a circular area delimited by a 50-mile radius from the ICPP. This area does not include Pocatello (2000 population 51,466) and Rexburg (17,257), which are located just outside the 50-mile radius. At these distances, air concentrations decreased slowly with increasing distance from the ICPP, so doses to people who lived in these cities were not appreciably different from doses estimated for people who lived in the lowest-dose group of communities. Thus, doses reported for Idaho Falls are representative of doses for Pocatello and Rexburg.[6]

Typical doses from exposure to I-131 for a person with a rural lifestyle are presented in Table 7.2. These doses are estimated assuming a backyard-cow milk consumption rate of two to four 8-oz glasses per day for all age groups other than an infant, who was assumed to be breastfed for the first year of life. The largest doses in this case are obtained for a 1 year old (in 1957). A breastfed infant received a lower dose than a typical 1 year old because I-131 was less concentrated in mother's milk than in cow's milk. The longer an infant was breastfed, the lower the dose he or she received (Table 7.3).

The highest thyroid doses estimated at representative locations are presented in Table 7.4. These doses are based on an assumed consumption of very large amounts of milk (three to nine 8-oz glasses per day) for every day during the 3 years of release. An infant was assumed to consume no mother's milk or formula, but only large amounts of backyard cow or goat milk (two to four 8-oz glasses per day). Such consumption rates are possible, but probably were very uncommon. The largest doses at each selected location again are estimated for a person 1 year of age in 1957. The largest dose is calculated for a 1 year old living in Mud Lake, Idaho (not listed in the table): 4.9 cSv with a 95% C.I. of 0.53 – 33 cSv for consumption of goat milk, and 1.1 cSv with a 95% C.I. of 0.16 – 6.7 cSv for consumption of backyard cow milk.

Thyroid doses from exposure to I-131 decrease with increasing age at exposure (Tables 7.2 and 7.4), mainly because of the increase in thyroid mass with age. At the same consumption rate, an adult receives a dose about 7 times less than the dose received by a 1 year old.

[6] When Pocatello and Rexburg are considered, the percentage of the population that lived in the lowest-dose group of communities becomes 96%, while 2% remains in each of the medium- and highest-dose groups.

Table 7.2 Estimated thyroid doses to rural residents of different ages who consumed large amounts of milk from a backyard cow

Age in 1957	Thyroid dose (cSv)* 95% Confidence Interval		
	Lower bound	Central Estimate†	Upper bound
Terreton, Idaho			
Born in 1959	0.00090	0.0053	0.040
Born in 1958	0.0147	0.077	0.51
infant	0.068	0.36	2.4
1	0.10	0.51	3.4
5	0.068	0.32	2.2
10	0.036	0.17	1.2
15	0.018	0.088	0.61
25 - female	0.016	0.077	0.53
25 - male	0.014	0.069	0.44
Roberts, Idaho			
Born in 1959	0.00032	0.0019	0.013
Born in 1958	0.0043	0.025	0.15
infant	0.022	0.11	0.75
1	0.031	0.17	1.2
5	0.020	0.11	0.74
10	0.011	0.058	0.43
15	0.0057	0.029	0.21
25 - female	0.0050	0.025	0.18
25 - male	0.0046	0.023	0.15
Idaho Falls, Idaho			
Born in 1959	0.00011	0.00076	0.0047
Born in 1958	0.0018	0.010	0.056
infant	0.0076	0.044	0.27
1	0.011	0.066	0.43
5	0.0076	0.041	0.27
10	0.0048	0.022	0.16
15	0.0021	0.012	0.072
25 - female	0.0018	0.010	0.063
25 - male	0.0017	0.0088	0.056

* Based on a consumption rate of 2 to 4 glasses of backyard cow milk per day

† 50^{th} percentile of the probability distribution function describing the uncertainty in the estimated doses

Table 7.3 Estimated thyroid doses to a rural resident born in 1957 as a function of the amount of mother's milk consumed

Breast feeding status	Thyroid dose (cSv)* 95% confidence interval		
	Lower bound	Central Estimate†	Upper bound
Terreton, Idaho			
No breast feeding	0.10	0.53	3.8
3 months on mother's milk	0.10	0.51	3.7
6 months on mother's milk	0.086	0.45	3.2
12 months on mother's milk	0.068	0.36	2.4
18 months on mother's milk	0.035	0.18	1.2
Roberts, Idaho			
No breast feeding	0.032	0.17	1.3
3 months on mother's milk	0.030	0.16	1.2
6 months on mother's milk	0.027	0.14	1.0
12 months on mother's milk	0.022	0.11	0.75
18 months on mother's milk	0.012	0.062	0.38
Idaho Falls, Idaho			
No breast feeding	0.012	0.07	0.47
3 months on mother's milk	0.011	0.066	0.44
6 months on mother's milk	0.010	0.057	0.38
12 months on mother's milk	0.0076	0.044	0.27
18 months on mother's milk	0.0040	0.024	0.14

* After the infant is weaned, it is assumed that he or she consumes high amounts of backyard cow milk (two to four 8-oz glasses per day)

† 50^{th} percentile of the probability distribution function describing the uncertainty in the estimated doses

Table 7.4 Highest estimated thyroid doses to rural residents of different ages

Age in 1957	Goat milk			Backyard cow milk		
	Thyroid dose (cSv)*					
	Lower bound	Central Estimate†	Upper bound	Lower bound	Central Estimate†	Upper bound
Terreton, Idaho						
Infant	0.31	2.6	17	0.10	0.53	3.7
1	0.46	4.7	33	0.16	1.0	6.1
5	0.30	3.0	21	0.10	0.63	3.9
10	0.16	1.6	11	0.055	0.34	2.1
15	0.081	0.79	5.7	0.028	0.17	1.0
25	0.069	0.68	4.9	0.025	0.15	0.89
Roberts, Idaho						
Infant	0.12	0.84	5.6	0.033	0.17	1.3
1	0.16	1.5	10	0.050	0.31	2.1
5	0.11	1.0	6.4	0.032	0.20	1.3
10	0.055	0.52	3.4	0.018	0.11	0.7
15	0.028	0.26	1.7	0.0093	0.055	0.36
25	0.024	0.22	1.4	0.0078	0.047	0.30
Idaho Falls, Idaho						
Infant	0.043	0.33	2.3	0.012	0.07	0.47
1	0.063	0.58	4.1	0.020	0.12	0.85
5	0.041	0.36	2.6	0.013	0.080	0.52
10	0.021	0.20	1.4	0.0069	0.043	0.27
15	0.011	0.10	0.71	0.0034	0.022	0.14
25	0.0095	0.084	0.60	0.0030	0.019	0.12

* Doses are based on a very high consumption of milk (three to nine 8-oz glasses per day) for all age groups other than an infant, for whom a high consumption of milk (two to four glasses per day) was assumed. These consumption rates were assumed to occur for the entire duration of releases from the ICPP included in this study (1957-1959). Also, it was assumed that the infant received no mother's milk or formula (i.e., all the milk came from the source of milk specified).

† 50th percentile of the probability distribution function describing the uncertainty in the estimated doses.

Table 7.5 Estimated thyroid doses to a rural resident born in 1952 (age 5 in 1957) as a function of milk consumption rate

Milk consumption Rate*	Thyroid dose (cSv) 95% confidence interval		
	Lower bound	Central estimate†	Upper bound
Terreton, Idaho			
No Milk	0.0062	0.027	0.23
Low	0.014	0.13	1.1
Average	0.040	0.24	1.4
High	0.068	0.32	2.2
Very High	0.10	0.63	3.9
Roberts, Idaho			
No Milk	0.0021	0.010	0.089
Low	0.0048	0.042	0.34
Average	0.013	0.076	0.49
High	0.020	0.11	0.74
Very High	0.032	0.20	1.3
Idaho Falls, Idaho			
No Milk	0.00077	0.0039	0.031
Low	0.0019	0.016	0.13
Average	0.0050	0.029	0.18
High	0.0076	0.041	0.27
Very High	0.013	0.079	0.52

* The milk consumption categories are based on the following consumption rates (in 8-oz glasses per day): low = 0-2 glasses/day; average = 1-3 glasses/day; high = 2 to 4 glasses/day; and very high = 3 to 9 glasses/day.

† 50^{th} percentile of the probability distribution function describing the uncertainty in the estimated doses.

Table 7.6 Estimated thyroid doses from each exposure pathway for a child born in 1957 who lived in Terreton in a family with a rural resident lifestyle

Exposure pathway	Thyroid dose (cSv) 95% confidence interval		
	Lower bound	Central estimate*	Upper bound
Goat milk (very high consumption rate†)	0.46	4.7	33
Backyard cow milk (very high consumption rate†)	0.16	1.0	6.1
Goat milk (high consumption rate‡)	0.28	2.0	17
Backyard cow milk (high consumption rate‡)	0.10	0.51	3.4
Inhalation§	0.00026	0.015	0.32
Eggs	0.0013	0.0058	0.022
Inadvertent soil ingestion	0.000072	0.0026	0.080
Vegetables	0.00025	0.0014	0.0088
Beef	0.000052	0.00085	0.014

* 50th percentile of the probability distribution function describing the uncertainty in the estimated doses

† Three to nine 8-oz glasses per day

‡ Two to four 8-oz glasses per day

§ Assumes large amounts of time spent outdoors

The effect of consumption of different amounts of milk is presented in Table 7.5 for a child 5 years of age in 1957. These results show that doses from exposure to I-131, taking into account all exposure pathways, essentially are proportional to the amount of milk ingested. This result is due to the fact that the dose from ingestion of milk is much larger than the dose from all other exposure pathways (Table 7.6).

The gender of an exposed individual does not significantly affect estimated doses. For children and young teenagers, doses are essentially the same for either gender. Estimated doses are somewhat different for males and females who are exposed at ages 16 and older, mainly due to gender differences in thyroid mass.[7] For the same exposure conditions (i.e., the same milk consumption rate), doses received by adult females are about 10% higher than doses received by adult males.

The uncertainty in doses estimated for a rural resident is about a factor of 7 (measured as the ratio of the 97.5th and 50th percentiles). This uncertainty is slightly larger than the uncertainty in estimated doses due to I-131 releases from the RaLa plant in Oak Ridge, Tennessee (Apostoaei et al. 1999), an effect due mainly to the larger uncertainty in the site-specific deposition velocities used in this study (see Appendix A). However, the uncertainty is similar to the uncertainty in doses from I-131 released from Hanford (Snyder et al. 1994) and from weapons testing at the Nevada Test Site (NTS) (NCI 1997).

[7] Dose is calculated as the energy deposited by radiation divided by the mass of the organ in which the energy was deposited. Assuming the same milk consumption rate, adult males and females receive similar amounts energy to the thyroid gland, but the mass of the thyroid is smaller in females than in males.

To assess the dose to a person living a rural lifestyle, one has to know the age at time of exposure, the type and amount of milk consumed during the years of releases, and the location where the person lived. That is, a person who lived at the specified locations can use the tables included in this section to identify his or her personal dose according to their own milk consumption habits and their age in 1957.

7.1.2 Migrant Worker Scenario

During the 1950s and 1960s, migrant workers traveled from Texas for temporary jobs during the farming season. Many activities took place in April, May, and June involving sowing/planting and plant care in the first stages of plant development (irrigation, insecticide spraying). Activity was reduced during July and August; however, some workers stayed in the area to help with weeding. A large number of workers came during the harvest season (September and October). Harvest was important enough that local schools would close for a few days, so that students could help harvest crops.

Some migrant workers became Idaho residents, but they continued in a similar lifestyle by migrating from farm to farm for different jobs within the same area. Migrant farm workers were adult males and females, but sometimes they traveled as families, including children of all ages. Older children would help with some of the farm work. In the 1950s and 1960s, migrant workers were housed in temporary dwellings at a farm or in migrant worker camps located near the farm. Workers were responsible for procuring their own food, although some employers provided meals during certain jobs. In addition to farms, food-processing plants required a large number of employees during the harvest season.

Doses to an adult male migrant farm worker were estimated at all locations included in this study (Table 7.7). Doses were estimated assuming a high consumption rate (2 to 4 glasses/day) of backyard cow milk (assumed to be provided by the employer) and residence at the specified location from April to November in each year of release. Since children could have been traveling with their parents, they would have received larger doses than adults under the same conditions (Table 7.8). Doses to children of migrant farm workers (Table 7.8) and the doses to children of rural residents (Table 7.1) are nearly the same, even though migrant workers were not present in the area during wintertime, because concentrations of I-131 in milk during wintertime were negligible compared with the concentrations during other seasons (see Appendix B, Figure B.1).

Table 7.7 Estimated thyroid doses to an adult male migrant worker who consumed large amounts of milk from a backyard cow

	Community or town	Population in 2000†	Thyroid dose (cSv)* 95% confidence interval		
			Lower bound	Central estimate‡	Upper bound
1	Aberdeen Junction	1840	0.0015	0.0078	0.053
2	Arco§	1026	0.0054	0.027	0.15
3	Atomic City§	25	0.0079	0.043	0.31
4	Basalt	419	0.0012	0.0058	0.037
5	Blackfoot§	10,419	0.0011	0.0054	0.038
6	Butte City§	76	0.0083	0.038	0.23
7	Dubois	647	0.0069	0.031	0.19
8	Firth	408	0.0012	0.0059	0.039
9	Fort Hall	3,193	0.00092	0.0041	0.025
10	Grandview	470	0.0016	0.0078	0.055
11	Hamer	12	0.0083	0.038	0.25
12	Howe§	20¶	0.013	0.062	0.41
13	Idaho Falls§	50,730	0.0015	0.0071	0.051
14	Lewisville	467	0.0027	0.013	0.092
15	Lost River	26	0.0035	0.016	0.097
16	Mackay	566	0.0010	0.0050	0.031
17	Menan	707	0.0028	0.013	0.087
18	Moore	196	0.0028	0.015	0.091
19	Monteview§	10¶	0.012	0.055	0.33
20	Mud Lake§	270	0.013	0.063	0.44
21	Roberts§	647	0.0039	0.018	0.12
22	Shelley	3,813	0.0013	0.0062	0.038
23	Spencer	38	0.0015	0.0067	0.043
24	Terreton	1,537	0.012	0.058	0.38

* It is assumed that the person was born in 1932, and that he or she consumed 2 to 4 glasses of backyard cow milk per day.

† Data from the US Census Bureau: factfinder.census.gov and www.digital-neighbors.com.

‡ 50th percentile of the probability distribution function describing the uncertainty in the estimated doses.

§ Locations included in the DOE (1991) dose reconstruction for INEL.

¶ Source: INEL (2002c).

Table 7.8 Estimated thyroid to a child of a migrant worker born in 1956 who consumed large amounts of milk from a backyard cow

	Community or town	in 2000[†]	Thyroid dose (cSv)[*] 95% confidence interval		
			Lower bound	Central estimate[‡]	Upper bound
1	Aberdeen Junction	1840	0.011	0.059	0.41
2	Arco[§]	1026	0.040	0.20	1.2
3	Atomic City[§]	25	0.059	0.32	2.4
4	Basalt	419	0.009	0.044	0.29
5	Blackfoot[§]	10,419	0.0079	0.042	0.29
6	Butte City[§]	76	0.061	0.28	1.83
7	Dubois	647	0.048	0.24	1.5
8	Firth	408	0.009	0.045	0.29
9	Fort Hall	3193	0.0065	0.031	0.20
10	Grandview	470	0.011	0.058	0.42
11	Hamer	12	0.060	0.29	1.9
12	Howe[§]	20[¶]	0.095	0.47	3.2
13	Idaho Falls[§]	50,730	0.011	0.054	0.40
14	Lewisville	467	0.019	0.10	0.68
15	Lost River	26	0.025	0.12	0.76
16	Mackay	566	0.0072	0.038	0.23
17	Menan	707	0.019	0.10	0.65
18	Moore	196	0.021	0.11	0.72
19	Monteview[§]	10[¶]	0.088	0.42	2.6
20	Mud Lake[§]	270	0.10	0.49	3.3
21	Roberts[§]	647	0.028	0.14	0.94
22	Shelley	3813	0.010	0.047	0.30
23	Spencer	38	0.011	0.051	0.34
24	Terreton	1537	0.089	0.44	2.9

* The doses were obtained assuming a milk consumption rate of 2 to 4 glasses/day

† Data from the US Census Bureau: factfinder.census.gov and www.digital-neighbors.com

‡ 50th percentile of the probability distribution function describing the uncertainty in the estimated doses

§ Locations included in the DOE (1991) dose reconstruction for INEL

¶ Source: INEL (2002c)

Doses to different age groups of migrant workers or family members at selected locations are presented in Tables 7.9 and 7.10 for high backyard cow milk consumption and average local commercial milk consumption, respectively. Backyard cow milk was assumed to be provided by the employer, while commercial milk was purchased from a local store and was assumed to be produced by a local dairy farm.

The doses presented in Tables 7.7 through 7.10 assume that the migrant worker and his family were present in the INEL region from April through November during the 3 years of releases from the ICPP. As discussed above, migrant workers could have been present in the area only during the spring and fall of each year, while traveling out of the area during July and August. However, doses during the summer months (especially for 1957 and 1958) were very low as no RaLa runs took place at the ICPP (see Figure 5.4). Thus, an assumed absence from the region during the summer months does not significantly reduce the doses in Tables 7.7 through 7.10, as indicated in Figure 7.1. The reduction in doses due to a limited amount of time in the area is much less important than the type of milk diet.

Table 7.9 Estimated thyroid doses to members of a migrant worker family of different ages who consumed large amounts of milk from a backyard cow

Age in 1957	Thyroid dose (cSv)* 95% confidence interval		
	Lower bound	Central Estimate†	Upper bound
Terreton, Idaho			
Born in 1959	0.00066	0.0043	0.031
Born in 1958	0.014	0.077	0.58
infant	0.065	0.33	2.2
1	0.089	0.44	2.9
5	0.057	0.28	1.9
10	0.031	0.15	1.0
15	0.016	0.078	0.50
25 - female	0.014	0.066	0.44
25 - male	0.012	0.058	0.38
Roberts, Idaho			
Born in 1959	0.00023	0.0013	0.012
Born in 1958	0.0042	0.021	0.15
infant	0.019	0.094	0.64
1	0.028	0.14	0.94
5	0.018	0.088	0.60
10	0.010	0.048	0.33
15	0.0051	0.025	0.16
25 - female	0.0044	0.021	0.14
25 - male	0.0039	0.018	0.12
Idaho Falls, Idaho			
Born in 1959	0.000085	0.00050	0.0044
Born in 1958	0.0014	0.0088	0.069
infant	0.0075	0.037	0.26
1	0.011	0.054	0.40
5	0.0070	0.035	0.26
10	0.0039	0.019	0.14
15	0.0020	0.010	0.070
25 - female	0.0017	0.0081	0.059
25 - male	0.0015	0.0071	0.051

* Based on a consumption rate of 2 to 4 glasses of backyard cow milk per day.

† 50^{th} percentile of the probability distribution function describing the uncertainty in the estimated doses.

Table 7.10 Estimated thyroid doses to members of a migrant worker family of different ages who consumed average amounts of commercial cow milk

Age in 1957	Thyroid dose (cSv)* 95% confidence interval		
	Lower bound	Central Estimate†	Upper bound
Terreton, Idaho			
Born in 1959	0.00038	0.0022	0.013
Born in 1958	0.0076	0.041	0.27
Infant	0.030	0.14	0.75
1	0.038	0.16	1.0
5	0.025	0.11	0.64
10	0.014	0.061	0.35
15	0.0072	0.032	0.17
25 - female	0.0066	0.028	0.15
25 - male	0.0054	0.024	0.13
Roberts, Idaho			
Born in 1959	0.00011	0.0007	0.004
Born in 1958	0.0020	0.012	0.070
Infant	0.0085	0.041	0.23
1	0.011	0.054	0.30
5	0.0067	0.034	0.20
10	0.0041	0.019	0.11
15	0.0021	0.010	0.054
25 - female	0.0020	0.0089	0.046
25 - male	0.0016	0.0075	0.041
Idaho Falls, Idaho			
Born in 1959	0.000045	0.00028	0.0018
Born in 1958	0.00080	0.0048	0.034
Infant	0.0032	0.016	0.10
1	0.0041	0.021	0.13
5	0.0028	0.013	0.079
10	0.0015	0.0075	0.044
15	0.00084	0.0038	0.022
25 - female	0.00074	0.0034	0.019
25 - male	0.00061	0.0029	0.017

* Based on a consumption rate of 1-3 glasses of commercial cow milk per day.

† 50^{th} percentile of the probability distribution function describing the uncertainty in the estimated doses.

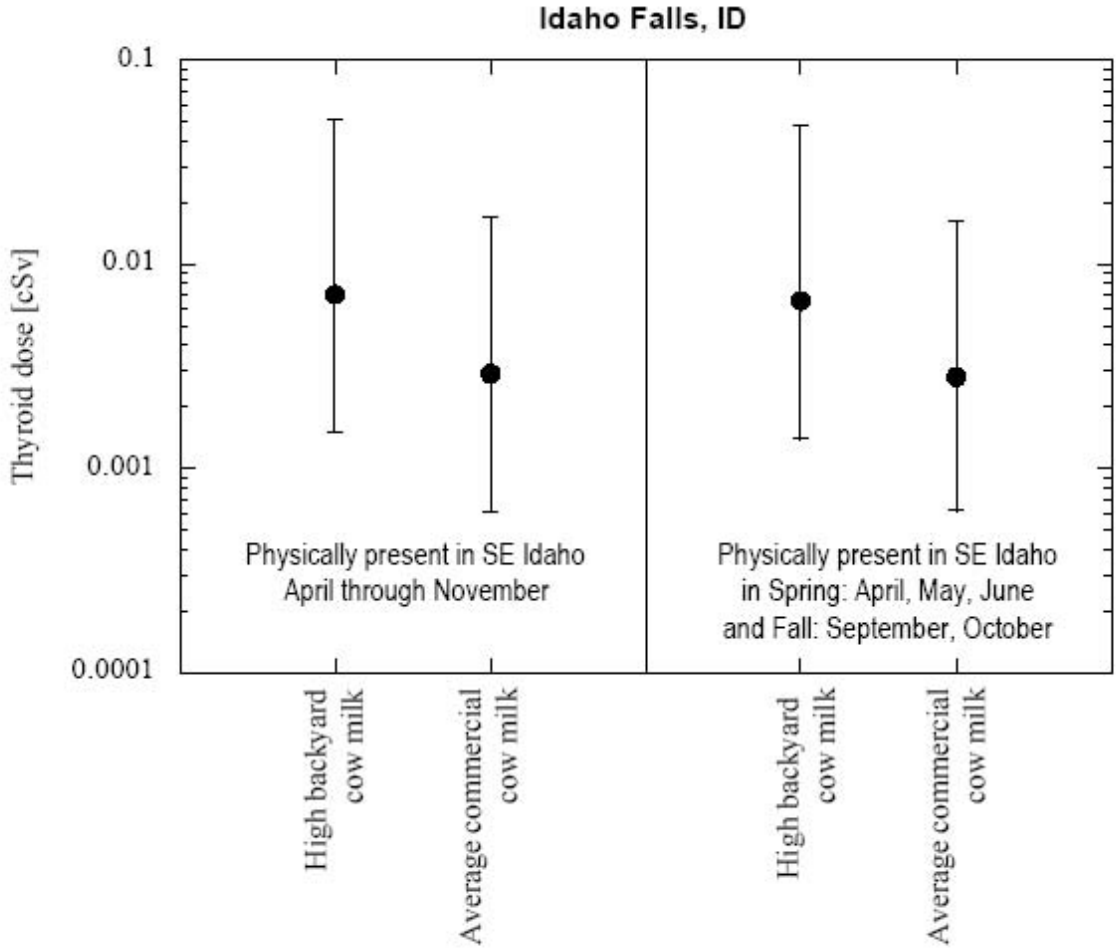

Figure 7.1 Differences between doses to an adult migrant farm worker due to different diet of milk and different amounts of time spent in the region

A high milk consumption rate means 2 to 4 glasses/day, while an average consumption rate means 1-3 glasses/day. The vertical lines represent 95% confidence intervals.

7.1.3 Urban Resident Scenario

As used in this report, an urban resident is representative of an individual who purchased most of his or her food products from grocery stores without regard for whether the place of residence is a rural or urban location. An urban resident scenario assumes that stores obtained food from local producers (local commercial food) or from producers in the extended INEL region (regional commercial food). For instance, a store can obtain milk from a local dairy farm or from multiple large dairies. Thus, doses to an urban resident are reported for a diet of locally produced food and a diet of regionally produced food. The concentrations of I-131 in regionally produced food represent an average concentration over an area within about 10 miles of the portion of Snake River that passes through the study domain (Figure 4.1). This area contains the major dairy farms in the region.

Doses received from consuming regionally produced foods are similar to the doses received by an individual living in Roberts, ID, on a diet of locally produced commercial foods (Tables 7.11 through 7.14). The highest doses from consumption of one to three 8-oz glasses of commercial milk per day were obtained for an individual born in 1956 and living in Mud Lake, Idaho (0.21 cSv; 95% C.I. = 0.042 to 1.1 cSv).

For the same consumption of milk, doses to an urban resident are generally lower than doses to a rural resident because the concentration of I-131 in commercial milk is lower than the concentration in backyard cow milk (see Appendix B; Table B.1). Furthermore, most of the large commercial dairies were located near the Snake River, where concentrations of I-131 in air are low compared to concentrations at locations near the site boundary.

Table 7.11 Estimated thyroid doses to an urban resident born in 1956 who consumed average amounts of local or regional commercial cow milk

	Community or town	Population 2000[†]	Thyroid dose (cSv)* 95% confidence interval		
			Lower bound	Central estimate[‡]	Upper bound
Local Commercial Foods					
1	Aberdeen Junction	1840	0.0050	0.026	0.14
2	Arco[§]	1026	0.016	0.086	0.44
3	Atomic City[§]	25	0.026	0.14	0.78
4	Basalt	419	0.0039	0.019	0.10
5	Blackfoot[§]	10,419	0.0035	0.018	0.10
6	Butte City[§]	76	0.023	0.12	0.66
7	Dubois	647	0.021	0.10	0.55
8	Firth	408	0.0035	0.019	0.10
9	Fort Hall	3193	0.0027	0.014	0.073
10	Grandview	470	0.0051	0.025	0.14
11	Hamer	12	0.024	0.13	0.65
12	Howe[§]	20[¶]	0.040	0.20	1.1
13	Idaho Falls[§]	50,730	0.0044	0.025	0.13
14	Lewisville	467	0.0089	0.047	0.25
15	Lost River	26	0.0094	0.051	0.30
16	Mackay	566	0.0033	0.016	0.084
17	Menan	707	0.0091	0.045	0.24
18	Moore	196	0.0094	0.047	0.25
19	Monteview[§]	10[¶]	0.039	0.19	1.0
20	Mud Lake[§]	270	0.042	0.21	1.2
21	Roberts[§]	647	0.013	0.064	0.35
22	Shelley	3813	0.0040	0.021	0.11
23	Spencer	38	0.0044	0.022	0.13
24	Terreton	1537	0.040	0.20	1.1
Regional Commercial Foods					
	All locations	---	0.013	0.061	0.33

* Doses were obtained assuming a milk consumption rate of 1-3 glasses/day.

† Data from the US Census Bureau: factfinder.census.gov and www.digital-neighbors.com.

‡ 50th percentile of the probability distribution function describing the uncertainty in the estimated doses.

§ Locations included in the DOE (1991) dose reconstruction for INEL.

¶ Source: INEL (2002c).

Table 7.12 Estimated thyroid doses to urban residents of different ages who consumed average amounts of local commercial cow milk

Age in 1957	Thyroid dose (cSv)* 95% confidence interval		
	Lower bound	Central Estimate†	Upper bound
Terreton, Idaho			
Born in 1959	0.00050	0.0028	0.019
Born in 1958	0.0073	0.037	0.24
Infant	0.029	0.15	0.83
1	0.040	0.20	1.1
5	0.027	0.13	0.71
10	0.015	0.072	0.39
15	0.0078	0.036	0.19
25 - female	0.0072	0.031	0.18
25 - male	0.0065	0.028	0.15
Roberts, Idaho			
Born in 1959	0.00010	0.00070	0.00530
Born in 1958	0.0017	0.0083	0.050
Infant	0.0088	0.045	0.25
1	0.013	0.064	0.34
5	0.0089	0.041	0.23
10	0.0052	0.024	0.13
15	0.0026	0.012	0.063
25 - female	0.0024	0.010	0.056
25 - male	0.0022	0.0093	0.046
Idaho Falls, Idaho			
Born in 1959	0.000037	0.00025	0.0018
Born in 1958	0.00068	0.00329	0.01956
Infant	0.0036	0.017	0.083
1	0.0044	0.025	0.13
5	0.0034	0.016	0.082
10	0.0020	0.009	0.047
15	0.00090	0.0045	0.024
25 - female	0.00080	0.0038	0.021
25 - male	0.00070	0.0035	0.018

* Based on a consumption rate of 1-3 glasses of commercial cow milk per day.

† 50^{th} percentile of the probability distribution function describing the uncertainty in the estimated doses.

Table 7.13 Estimated thyroid doses to urban residents of different ages who consumed low amounts of local commercial cow milk

Age in 1957	Thyroid dose (cSv)* 95% confidence interval		
	Lower bound	Central Estimate†	Upper bound
Terreton, Idaho			
Born in 1959	0.00050	0.0028	0.019
Born in 1958	0.0067	0.034	0.21
Infant	0.015	0.088	0.50
1	0.011	0.11	0.65
5	0.0093	0.070	0.44
10	0.0062	0.039	0.24
15	0.0029	0.020	0.12
25 - female	0.0027	0.018	0.11
25 - male	0.0027	0.016	0.092
Roberts, Idaho			
Born in 1959	0.00010	0.00070	0.0053
Born in 1958	0.0010	0.0066	0.043
Infant	0.0089	0.043	0.23
1	0.0045	0.034	0.23
5	0.0035	0.024	0.15
10	0.0022	0.013	0.080
15	0.0010	0.0067	0.041
25 - female	0.00075	0.0058	0.038
25 - male	0.00058	0.0056	0.032
Idaho Falls, Idaho			
Born in 1959	0.000037	0.00025	0.0018
Born in 1958	0.00039	0.0027	0.019
Infant	0.0012	0.0098	0.063
1	0.0016	0.014	0.084
5	0.0014	0.0094	0.056
10	0.00083	0.0052	0.033
15	0.00042	0.0027	0.015
25 - female	0.00038	0.0024	0.014
25 - male	0.00038	0.0023	0.012

* Based on a consumption rate of 0-2 glasses of commercial cow milk per day.

† 50^{th} percentile of the probability distribution function describing the uncertainty in the estimated doses.

Table 7.14 Estimated thyroid doses to urban residents of different ages who consumed milk from regional commercial dairies

Age in 1957	Thyroid dose (cSv)* 95% confidence interval		
	Lower bound	Central Estimate†	Upper bound
High Regional Commercial Milk Consumption (2 to 4 Glasses/Day)			
Born in 1959	0.00017	0.00077	0.0046
Born in 1958	0.0028	0.013	0.067
Infant	0.012	0.062	0.31
1	0.018	0.086	0.41
5	0.013	0.055	0.27
10	0.0069	0.030	0.14
15	0.0035	0.015	0.072
25 - female	0.0029	0.013	0.067
25 - male	0.0028	0.0117	0.056
Average Regional Commercial Milk Consumption (1-3 Glasses/Day)			
Born in 1959	0.00017	0.00077	0.00457
Born in 1958	0.0023	0.0113	0.065
Infant	0.0090	0.044	0.24
1	0.013	0.061	0.33
5	0.0078	0.038	0.21
10	0.0046	0.021	0.11
15	0.0023	0.011	0.054
25 - female	0.0021	0.009	0.050
25 - male	0.0018	0.0083	0.042
Low Regional Commercial Milk Consumption (0-2 Glasses/Day)			
Born in 1959	0.000089	0.00052	0.0037
Born in 1958	0.0010	0.0064	0.044
Infant	0.0027	0.024	0.15
1	0.0035	0.032	0.20
5	0.0031	0.020	0.13
10	0.0018	0.011	0.074
15	0.00086	0.0059	0.035
25 - female	0.00078	0.0053	0.034
25 – male	0.00087	0.0048	0.028

* Given the definition of regional-commercial food products (see text), these results apply at any location.

† 50^{th} percentile of the probability distribution function describing the uncertainty in the estimated doses.

7.1.4 A Perspective on Doses from Exposure to Iodine-131

Thyroid doses to members of the public estimated in this analysis are generally low due to the low amounts of I-131 released to the atmosphere from the ICPP during the period of concern, the large distances from the ICPP to the communities where people live, and the locations of many communities in directions perpendicular to the main wind direction (i.e., crosswind as opposed to downwind). Table 7.15 shows a comparison with doses from other atmospheric releases of I-131 in the United States. Releases of I-131 from the ICPP during 1957-1959 are lower than releases from any other source for which detailed dose reconstructions have been performed. Doses estimated for ICPP releases are more than one order of magnitude lower than doses from the sources listed in Table 7.15, regardless of the type and amount of milk assumed.

For a person living near the INEL site during 1957-1959, doses due to I-131 released from the ICPP also are lower than doses that the same person received due to exposure to I-131 in fallout from nuclear weapons tests that took place in Nevada during the same time period (Table 7.16). This statement is true even for residents who lived as close to the ICPP as Atomic City, Howe, Terreton, or Arco, Idaho, or for people with dates of birth different from those in Table 7.16.

A typical thyroid dose from natural sources of radiation is about 0.09 cSv y^{-1} (NCRP 1987). Thus, an individual living near INEL would have received a thyroid dose of about 0.3 cSv from natural background radiation during 1957–1959, and the background dose from 1957 until 2003 would be about 4 cSv. Typical doses to a rural resident (Tables 7.1 and 7.2) are lower than doses from natural background during the period of release. However, there are a few locations (e.g., Mud Lake, Terreton, Atomic City) where the highest estimated doses exceed doses from natural background (Tables 7.1, 7.2, and 7.4).

Table 7.15 Comparison of doses from exposure to iodine-131 from various dose reconstruction studies at the sites within the United States

Source of releases	Activity* of I-131 released [10^3 Ci]	Location of the individual	Thyroid dose (cSv) 95% Confidence Interval			Years of exposure	Year of birth
			Lower bound	Central Estimate†	Upper bound		
Maximum Exposed Individual Female Child On Backyard Cow Milk Diet							
Hanford	892	Richland, WA	54	240	870	1944-1951	Not specified
Nevada Test Site	151,000	Gunnison Co, CO	7	120	1700	1952-1971	1952‡
X-10 RaLa Oak Ridge, TN	9 - 43	Gallaher Bend, TN	6	39	250	1952-1956	1952
ICPP	2.4 - 5.1	Mud Lake, ID (high milk consumption)	0.11	0.56	3.7	1957-1959	1956
Typically Exposed Individual Female Child On Commercial Cow Milk Diet							
Hanford	892	Eastern Washington	3.2	10	32	1944-1951	Not specified
Nevada Test Site	151,000	Milwaukee, WI	2.0	7.5	33	1952-1971	1952‡
X-10 RaLa Oak Ridge, TN	9 - 43	Norwood, TN	0.20	1.0	10	1952-1956	1952
ICPP	2.4 - 5.1	Roberts, ID (high milk consumption)	0.013	0.064	0.35	1957-1959	1956
ICPP	2.4 - 5.1	Roberts, ID (average milk consumption)	0.0089	0.043	0.23	1957-1959	1956

* Source: UNSCEAR 2000, Hoffman et al. 2002, Bouville et al. 2002, and Section 2 of this report.

† 50^{th} percentile of the probability distribution function describing the uncertainty in the estimated doses.

‡ Dose to an individual born in June.

Table 7.16 Comparison of thyroid doses due to exposure to iodine-131 released from the Idaho Chemical Processing Plant and local fallout of iodine-131 from nuclear weapons testing at the Nevada Test Site

Source of I-131	Location†	Thyroid Dose (cSv)* 95% Confidence Interval		
		Lower bound	Central Estimate‡	Upper Bound
NTS fallout	Bingham County, ID	0.34	1.2	4.3
ICPP	Atomic City, ID	0.050	0.25	2.0
ICPP	Blackfoot, ID	0.0063	0.035	0.22
NTS fallout	Bonneville County, ID	0.49	1.9	7.3
ICPP	Idaho Falls, ID	0.0076	0.044	0.27
NTS fallout	Butte County, ID	0.55	3.2	19
ICPP	Arco, ID	0.024	0.15	1.0
ICPP	Howe, ID	0.059	0.37	2.4
NTS fallout	Custer County, ID	0.44	3.4	26
ICPP	Mackay, ID	0.0052	0.030	0.20
NTS fallout	Jefferson County, ID	1.0	5.0	25
ICPP	Terreton, ID	0.068	0.36	2.4
ICPP	Roberts, ID	0.022	0.11	0.75

* Doses are for a female born in 1957 who was breastfed during the first 12 months of life and then consumed large amounts of backyard cow milk.

† Doses from NTS fallout (NCI 1997; http://ntsi131.nci.nih.gov) are estimated for an entire county and they can be interpreted as applicable to all cities and communities in that county.

‡ 50th percentile of the probability distribution function describing the uncertainty in the estimated doses.

7.1.5 A Comparison with Previous Idaho National Engineering Laboratory Dose Reconstruction Studies

Doses estimated in this study can be compared with those estimated by DOE in the dose reconstruction study published in 1991. DOE (1991) summarized doses to a maximally exposed individual assuming different ages at exposure (infant, 10-year-old child, and adult). DOE (1991) reported doses from "operational" releases of radionuclides (i.e., chronic releases from routine operations) and doses from episodic releases, which include short-term releases from initial engine tests from the Aircraft Nuclear Propulsion Program and short-term accidental releases from other sources. For the purpose of this comparison, doses reported by DOE (1991) for operational releases were used. Those doses include releases of radionuclides from normal operations of all nuclear facilities at INEL during 1957-1959. Doses estimated by DOE for the time period 1957-1959 were apparently determined primarily by releases of I-131 from RaLa operations at the ICPP (DOE 1991; page 29). Thus, doses estimated by DOE (1991) from releases during 1957-1959 should be comparable to doses estimated in this study for I-131 releases from the ICPP.

For 1957 and 1958, doses estimated by DOE (1991) for infants and children are higher than the central estimate, but lower than the upper bound of the 95% confidence interval of doses estimated in this study for a diet of backyard cow milk (Table 7.17). For 1959, doses estimated in this study for a backyard cow milk diet are generally lower than doses estimated by DOE (1991) for all ages at exposure. This result can be attributed to differences in modeling approaches. Doses estimated by DOE (1991) are based on annual averages of I-131 concentrations in air, while monthly averages were used in this study. In 1959, a significant fraction of releases of I-131 occurred in February and March when cows where not yet out to pasture (Figure 5.4). Thus, the approach used in this study predicts lower average concentrations in milk over the entire year. In addition to differences in the modeling approaches, DOE (1991) could have used significantly different values for environmental transport parameters.

Table 7.17 Comparison of thyroid doses estimated in this study and doses reported by the Department of Energy (DOE 1991) for an individual living in Atomic City, Idaho

Year of release	Age at exposure	Thyroid dose (cSv)			
		DOE (1991)*	This Study† – Backyard Cow Milk Diet 95% Confidence Interval		
			Lower Bound	Central Estimate	Upper Bound
1957	Infant	0.88	0.029	0.19	1.7
1958	Infant	0.65	0.037	0.19	1.7
1959	Infant	0.14	0.0038	0.021	0.16
1957	Child	0.37	0.019	0.11	0.88
1958	Child	0.28	0.023	0.13	0.85
1959	Child	0.63	0.0022	0.013	0.10
1957	Adult	0.23	0.0066	0.043	0.35
1958	Adult	0.17	0.0071	0.044	0.32
1959	Adult	-- ‡	0.00070	0.0048	0.034

* Doses reported by DOE (1991) are for a maximally exposed individual and represent exposures to operational releases from the INEL.

† Doses based on high consumption rate (2 to 4 glasses per day) of backyard cow milk for infants and very high consumption rate (3 to 9 glasses per day) of backyard cow milk for 10-yr-old children and adults. Infants are assumed not to consume mother's milk.

‡ DOE (1991) predicted that in 1959, the skin was the organ receiving the maximum dose from exposure of adults to releases from INEL (and not thyroid).

7.2 Consideration of Offsite Doses from Other Radionuclides

The detailed dose reconstruction presented in Section 7.1 focuses on offsite doses to the public from exposure to I-131. The decision to focus on I-131 was based on a previous screening analysis to select radionuclides of concern in releases from the ICPP (Kocher 2005b). That analysis, which is summarized in Section 3, indicated that I-131 is by far the most important radionuclide when members of the public who resided near the INEL site boundary consumed substantial quantities of locally produced milk and other foods. Such exposure scenarios result in much higher estimates of dose than other scenarios for offsite exposure that do not involve consumption of locally produced foodstuffs.

Twelve additional radionuclides listed in Table 3.1 were identified in the previous screening analysis (Kocher 2005b) for consideration in a dose reconstruction for releases from the ICPP. However, a detailed reconstruction of doses to offsite members of the public from radionuclides other than I-131 is not presented in this report. Rather, a preliminary assessment was performed to estimate credible upper bounds of the effective dose and organ doses from exposure to all other radionuclides of concern combined. An approach of developing bounding estimates of dose, in lieu of a more rigorous dose reconstruction with full accounting of uncertainty, was based on an expectation that doses from other radionuclides of concern would be low. That expectation is indicated by the generally low doses from exposure to I-131 that were obtained in the dose reconstruction presented in Section 7.1 and the much lower doses from other radionuclides compared with I-131 that were calculated in the previous screening analysis to select radionuclides of concern (Kocher 2005b).

A preliminary assessment to investigate the potential importance of radionuclides other than I-131 in scenarios for exposure of the public at locations beyond the INEL site boundary is presented in Appendix C.2. That assessment indicates that the effective dose from all other radionuclides combined would be less than 0.1 cSv, and that the dose to any organ would be less than 1 cSv. These bounding estimates of dose were obtained for a worst-case scenario in which a high consumption rate of goat's milk by a young child is assumed. In a more common scenario involving a high consumption rate of cow's milk by a young child, bounding estimates of dose should be about a factor of 2 or 3 lower.

Given the low values of bounding estimates of doses to offsite members of the public from the other radionuclides of concern in releases from the ICPP, we believe that the merit of conducting a more detailed dose reconstruction for those radionuclides should be considered before proceeding.

7.3 Consideration of Onsite Exposure Scenarios

All scenarios considered in Sections 7.1 and 7.2 involve exposure at locations beyond the INEL site boundary. Those scenarios considered exposure of young children as well as adults, and doses to young children from exposure to I-131 were the highest of any age group. As described in Section 4, additional scenarios for exposure of the public at locations on the INEL site are considered in the dose reconstruction. Those scenarios, which apply only to adults, involve exposures of the following:

- An onsite rancher
- A hunter who consumes meat obtained from game that grazed on the site, but who does not come onto the site while hunting
- A one-time or regular visitor to the site

Such scenarios are potentially important in a dose reconstruction because onsite exposure (including exposure of livestock or game within the INEL site boundary) occurred at locations closer to the ICPP than offsite locations, and airborne concentrations of radionuclides over much of the site were higher than at any location beyond the boundary. Adults who are included in the

assumed scenarios also could have been exposed as rural or urban residents (see Sections 7.1.1 and 7.1.3). However, the scenarios listed above are concerned only with exposure on the INEL site, and they do not include additional exposures that could have occurred beyond the site boundary.

A detailed reconstruction of doses in the assumed scenarios for onsite exposure is not presented in this report. Rather, similar to the assessment of offsite doses from exposure to radionuclides other than I-131 discussed in the previous section, a preliminary assessment was performed to estimate credible upper bounds of effective doses and organ doses from onsite exposure of the public. An approach of developing bounding estimates of dose was based on the expectation that doses in onsite exposure scenarios would be low. That expectation is indicated by several considerations, including (1) the generally low doses from exposure to I-131 in scenarios for offsite exposure, as discussed in Section 7.1, (2) the lesser importance of other radionuclides compared with I-131 in scenarios for offsite exposure, as discussed in the previous section, (3) the limited number of exposure pathways that apply to onsite exposures, especially the absence of a milk pathway, which is the most important pathway for I-131 when it occurs, and (4) the generally lower exposure times at onsite locations compared with exposure times beyond the site boundary.

A preliminary assessment to investigate the potential importance of the assumed scenarios for onsite exposure of the public is presented in Appendix C.3. All radionuclides of potential importance in releases from the ICPP (see Table 3.1) were taken into account. The preliminary assessment indicates that the effective dose in the assumed scenarios for onsite exposure would be less than 0.01 cSv, and that the dose to any organ would be less than 0.1 cSv. These upper bounds are less than upper confidence limits of doses to adult rural residents at locations beyond the INEL site boundary that were obtained in the dose reconstruction discussed in Section 7.1, and they are much less than upper bounds of doses to infants and children at offsite locations.

Given the low values of bounding estimates of doses from all radionuclides released from the ICPP in assumed scenarios for onsite exposure of the public, we believe that the merit of conducting a more detailed dose reconstruction for those scenarios should be considered before proceeding.

8.0 SUMMARY AND CONCLUSIONS

This report presents estimates of doses to members of the public from exposure to I-131 released to the atmosphere from the Idaho Chemical Processing Plant (ICPP) at INEL during the years 1957-1959. During those years, releases of I-131 and other radionuclides during Radioactive Lanthanum (RaLa) process operations were the highest. The estimated radiation doses are the result of combining (1) estimates of releases of I-131 from the ICPP during the period of concern (Wichner et al. 2005a, b), (2) estimates of monthly average concentrations of I-131 in air at various locations beyond the INEL site boundary and on the site, which were obtained using detailed site-specific meteorological data and the CALPUFF atmospheric dispersion model (Radonjic et al. 2005), and (3) estimates of doses to individuals per unit time-integrated concentration of I-131 in air, which were obtained using assumed exposure scenarios and exposure pathway models.

Uncertainties in all doses from exposure to I-131 presented in this report were quantified. A central estimate was reported for each dose, representing the 50th percentile of the probability distribution describing the uncertainty in an estimated dose, and a 95% confidence interval is also provided. This confidence interval is interpreted as a "credibility" interval, meaning that there is a high degree of confidence (i.e., a subjective degree of belief) that the true dose to an individual with a defined lifestyle is within the specified lower and upper bounds.

The highest thyroid doses from I-131 were calculated for individuals living in the Mud Lake/Terreton area. Doses similar to those at Mud Lake/Terreton were estimated at Howe, Monteview, Atomic City, Hamer, and Butte City. For individuals born in 1956 who lived at those locations and consumed 2 to 4 glasses of milk from a backyard cow per day, the central estimate of thyroid doses varies from 0.32 to 0.56 cSv.[8] An overall uncertainty range among all locations extends from 0.049 to 3.7 cSv.[9] The uncertainty in the dose at a given location is approximately a factor of 7 about the central estimate (measured as the ratio of the 97.5th and 50th percentiles).

The lowest thyroid doses from I-131 were calculated for individuals living in Mackay or Fort Hall. Similar low doses were estimated at Aberdeen Junction, Idaho Falls, Grandview, Spencer, Shelley, Firth, and Basalt. For individuals born in 1956 who lived at those locations and consumed 2 to 4 glasses of milk from a backyard cow per day, the central estimate of thyroid doses varies from 0.035 to 0.069 cSv, while the overall uncertainty range extends from 0.0068 to 0.42 cSv.

Individuals who lived in Dubois, Arco, Roberts, Lost River, Moore, Menan, and Lewisville received intermediate doses. For individuals born in 1956 who lived at those locations and consumed 2 to 4 glasses of milk from a backyard cow per day, the central estimate of thyroid doses varies from 0.12 to 0.27 cSv, while the overall uncertainty range extends from 0.022 to 1.8 cSv.

[8] 1 cSv = 0.01 Sv = 1 rem.

[9] Measured between the lowest lower bound and the highest upper bound of the 95% confidence intervals of doses at all locations in this dose group.

Based on the 2000 population census, about 93% of the population near INEL lives in the lowest-dose group of communities, about 4% in the intermediate-dose group, and about 3% in the highest-dose group. The cities with the largest populations in the highest, intermediate and lowest-dose groupings of communities are Terreton, Arco, and Idaho Falls, respectively.

Consumption of contaminated milk is the main contributor to doses from I-131. Ingestion of other contaminated food products and inhalation of I-131 are minor contributors to doses if the exposed individual consumed milk. Given the importance of the milk pathway, the doses depend strongly on the type of milk consumed (e.g., backyard cow milk, commercial cow milk, or goat milk) and the amount of milk consumed. Consumption of goat milk produces the highest doses for individuals living at a given location. That is, doses from ingestion of goat milk are about 5 times higher than doses from ingestion of backyard cow milk (assuming similar consumption rates). On the other hand, doses from consumption of locally produced commercial milk are about a factor of 2 lower than the doses from consumption of backyard cow milk.

Another strong determinant of the magnitude of thyroid doses from I-131 is age at time of exposure. For the same type and amount of milk consumed, an adult receives a dose of only one-seventh of the dose received by a 1 year old child. The gender of an exposed individual does not significantly affect estimated doses. For children and young teenagers, doses are essentially the same for either gender. Estimated doses are somewhat different for males and females who are exposed at ages 16 and older, mainly due to gender differences in thyroid mass. For the same exposure conditions (i.e., the same milk consumption rate), doses received by adult females are about 10% higher than doses received by adult males.

Doses from exposure to I-131 released from the ICPP are substantially lower than offsite doses from other sources of I-131 in the United States, including the Hanford site at Richland, Washington; the Oak Ridge Reservation in Oak Ridge, Tennessee; and fallout from nuclear weapons testing at the Nevada Test Site (NTS). Moreover, for an individual residing at any location outside the borders of the INEL reservation, the doses from exposure to I-131 from NTS fallout are substantially higher than from exposure to I-131 from the ICPP. In general, the dominant importance of doses from NTS fallout increases with increasing distance from INEL.

A typical thyroid dose from natural sources of radiation is about 0.1 cSv y^{-1}. Thus, an individual living near INEL would have received a thyroid dose of about 0.3 cSv from natural background radiation during 1957–1959. Typical doses to individuals of different ages from I-131 released from the ICPP (Table 7.2) usually are much lower than thyroid doses received from natural background radiation during the period of release. However at most locations, the highest estimated doses to infants and young children in 1957 exceed doses from natural background.

A previous assessment of doses due to releases of radionuclides from the INEL in general and the ICPP in particular was performed by the Department of Energy (DOE 1991). Thyroid doses estimated in this study can be compared with those published by DOE (1991). The single-valued estimates of doses reported by DOE (1991) are higher than the central estimate and lower than the upper bound of the 95% confidence interval of doses estimated in this study for individuals who consumed 2 to 4 glasses of backyard cow milk per day.

Doses from offsite exposure to radionuclides released from the ICPP other than I-131 also were considered in the dose reconstruction. Additional radionuclides of concern had been identified in a previous screening analysis (Kocher 2005b). A detailed dose reconstruction for those radionuclides has not been performed. However, a preliminary assessment has indicated that the effective dose from offsite exposure to all other radionuclides combined would be less than 0.1 cSv, and that the dose to any organ would be less than 1 cSv. Thus, offsite doses from exposure to other radionuclides released from the ICPP were insignificant compared with doses from exposure to I-131.

Exposures of the public also could have occurred at locations within the INEL site boundary. Three different scenarios for onsite exposure were considered. These scenarios involve (1) an onsite rancher who consumed beef obtained from cattle that grazed on the site, (2) a hunter who consumed meat obtained from game that grazed on the site, but who did come onto the site while hunting, and (3) a one-time or regular visitor to the site. Exposed individuals in all scenarios were assumed to be adults. A detailed dose reconstruction for the assumed onsite exposure scenarios has not been performed. However, a preliminary assessment has indicated that the effective dose in all onsite exposure scenarios would less than 0.01 cSv, and that the dose to any organ would be less than 0.1 cSv. These upper-bound estimates are less than doses to adults from exposure to I-131 at locations of higher exposure beyond the site boundary, and they are much less than doses to infants and children at many offsite locations.

Given the small magnitude of doses estimated in this report for offsite exposures to radionuclides other than I-131 and for onsite exposures to any radionuclides, the merit of performing a more detailed analysis for these exposure situations should be considered before proceeding.

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APPENDIX A

METHODOLOGY FOR ASSESSMENT OF DOSES FROM EXPOSURE TO IODINE-131

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A. METHODOLOGY FOR ASSESSMENT OF DOSES FROM EXPOSURE TO IODINE-131

Introduction

This appendix describes the environmental transport model used to assess doses from exposure to I-131 released to atmosphere from the Idaho Chemical Processing Plant (ICPP) during 1957-1959, starting from reported concentrations of I-131 in air at selected locations around INEL (Section 4). Daily atmospheric releases of I-131 from the ICPP were estimated by Wichner et al. (2005), for the entire period from February 1, 1957, to December 31, 1959. Monthly average concentrations of I-131 in air at each location were estimated using the CALPUFF atmospheric dispersion modeling system (Radonjic et al. 2005).

The environmental transport model consists of nine major exposure pathways relevant to I-131 present in air:

- Ingestion of contaminated:
  - milk from family-owned (backyard) cows
  - milk from commercial dairies
  - goat's milk
  - mother's milk
  - beef
  - leafy vegetables
  - eggs
- Inadvertent ingestion of soil
- Inhalation of contaminated air

Uncertainties in doses from ingestion and inhalation of I-131 were quantified by propagation of uncertainties in all input parameters through the environmental transport model, using 500-iteration mid-point Latin Hypercube samples (LHS). Probability distribution functions were used to describe the uncertainties in all input parameters. For all parameters, the selection of probability distributions is based on an interpretation of the relevancy of the available data to the conditions around INEL during 1957-1959. This appendix presents the rationale behind the selection of probability distributions for all input parameters.

Since the probability distributions reflect the judgment of the investigators involved in this study, they can be called "subjective" probability distributions. Consequently, the uncertainty range obtained for any estimated dose represents a "credibility" interval, meaning that there is a high degree of confidence (i.e., a subjective degree of belief) that the true dose to an individual with a defined lifestyle is within the specified lower and upper bounds of the uncertainty range.

A.1 Transfer from Air to Vegetation and Soil

Iodine-131 in the atmosphere is deposited on both vegetation canopy and ground. Deposition can occur under dry conditions, or it can be associated with precipitation. Of the total contamination intercepted by vegetation surfaces, some may be removed by environmental weathering processes, such as wind, rain, and aging of the plant, while the rest is retained (Miller and Hoffman 1983).

Two categories of vegetation are considered in this study. The first is “pasture grass,” which is consumed by grazing animals and contributes to the contamination of milk and meat. The second category, "fresh leafy vegetables," is composed of vegetables such as lettuce, cabbage, and spinach. These vegetables are usually consumed a short time after harvest and thus may carry significant levels of contamination. The contamination of other types of vegetation is considered to be of negligible importance, due to low surface area available for direct interception, low concentration in surface soil that is available for root uptake, long period of time from harvest to human consumption, and losses from food processing and preparation.

A.1.1 Description of Processes and Model Assumptions

The following assumptions about the behavior of I-131 have been made, based on past experience in dose reconstruction at Hanford (Snyder et al. 1994) and Oak Ridge (Apostoaei et al. 1999).

- Three physico-chemical forms of iodine are considered to be present in air, each having different deposition rates onto soil and vegetation:
  - A highly reactive form, assumed to be molecular or elemental iodine (I_2)
  - A particulate form, which accounts for iodine attached to atmospheric aerosols
  - A nonreactive form, assumed to be organic iodine (e.g., CH_3I)
- Iodine is deposited on vegetation by both dry and wet deposition processes. Dry deposition refers to the process of iodine being removed directly from air by collection on vegetation, soil, or water surfaces. Wet deposition refers to the removal of iodine from air by various types of precipitation, including rain and snow.
- Dry deposition occurs even during periods of precipitation.
- Uptake of I-131 from soil by plants is negligible when compared to direct deposition from the atmosphere, due to dilution from mixing of deposited activity with soil in the root zone, and due to radioactive decay during the period of time between deposition on the soil surface and uptake by plants.

A.1.2 Modeling Approach

Concentrations of I-131 on pasture grass and on vegetation are derived from calculated ground-level air concentrations using a constant deposition rate equal to the monthly average deposition rate. This model is a steady-state approach and is applicable to a continuous release described by the monthly average concentration of iodine in air.

The transfer of radionuclides from air to pasture grass is given by Equation A.1, the transfer to leafy vegetables is given by Equation A.2, and the transfer to soil is given by Equation A.3. The concentration in soil is used to estimate the intake of iodine by animals and humans due to ingestion of contaminated soil. As stated before, the transfer of iodine from soil to plant via root uptake is negligible compared to direct deposition of iodine from air onto plant surfaces.

$$C_p = C_a \cdot AP$$
$$= C_a \cdot \left[V_d \cdot \left(\frac{r}{Y}\right)_{dry} + V_w \cdot \left(\frac{r}{Y}\right)_{wet} \right] \cdot \frac{1 - \exp\left(-\lambda_{eff} \cdot t_{ag}\right)}{\lambda_{eff}} \quad \text{(A.1)}$$

$$C_v = C_a \cdot AV$$
$$= C_a \cdot \left[V_d \cdot \left(\frac{r}{Y}\right)_{dry} + V_w \cdot \left(\frac{r}{Y}\right)_{wet} \right] \cdot \gamma \cdot \frac{1 - \exp\left(-\lambda_{eff} \cdot t_{ag}\right)}{\lambda_{eff}} \quad \text{(A.2)}$$

$$C_s = C_a \cdot AS$$
$$= C_a \cdot \left[V_d + V_w\right] \cdot \frac{1}{\lambda_{eff,soil} \cdot SD \cdot \rho_s} \quad \text{A.3)}$$

where,

C_p	=	the radionuclide concentration in pasture grass [Bq $kg^{-1}_{dry\ mass}$]
C_a	=	the radionuclide concentration in air [Bq m^{-3}_{air}]
AP	=	the transfer factor from air to pasture grass [Bq $kg^{-1}_{dry\ mass}$ per Bq m^{-3}_{air}]
C_v	=	radionuclide concentration in leafy vegetables [Bq $kg^{-1}_{fresh\ mass}$]
AV	=	the transfer factor from air to leafy vegetables [Bq $kg^{-1}_{fresh\ mass}$ per Bq m^{-3}_{air}]
C_s	=	concentration in soil [Bq kg^{-1}_{soil}]
AS	=	the transfer factor from air to soil [Bq m^{-2}_{soil} per Bq m^{-3}_{air}]
V_d	=	the total dry deposition velocity [m d^{-1}]
V_w	=	the total wet deposition velocity [m d^{-1}]
$(r/Y)_{dry}$	=	the mass interception factor (dry deposition) [m^2 $kg^{-1}_{dry\ mass}$]
$(r/Y)_{wet}$	=	the mass interception factor (wet deposition) [m^2 $kg^{-1}_{dry\ mass}$]
γ	=	the ratio of the mass of dry matter of vegetation and its fresh mass [kg_{dry} (kg^{-1}_{wet})]

t_{ag} = the time period of exposure of the standing crop biomass [d]

λ_{eff} = the effective removal rate constant from vegetation [d^{-1}] = $\lambda_R+\lambda_w$

where,

λ_R = radiological decay constant [d^{-1}]

λ_w = removal rate from vegetation due to weathering [d^{-1}]

$\lambda_{eff,\ soil}$ = the effective removal rate constant from soil surface [d^{-1}] = $\lambda_R+\lambda_S$

where,

λ_R = radiological decay constant [d^{-1}]

λ_w = removal rate from soil surface due to leaching [d^{-1}]

SD = the surface depth of soil consumed by animals [m]

ρ_s = the density of soil [kg m^{-3}]

The parameters for modeling transfer of iodine to pasture and to leafy vegetables are the assumed to be same. The concentration in pasture grass is expressed per dry mass (i.e., Bq $kg^{-1}_{dry\ mass}$), while the concentration in leafy vegetables is expressed per fresh (or wet) mass (i.e., Bq $kg^{-1}_{fresh\ mass}$).

Equations A.1 through A.3 are applicable to each physico-chemical form of iodine (see Section A.1.3). Concentrations of I-131 in air, deposition velocities, and mass interception factors have different values for each physico-chemical form, while the other parameters are the same for all forms of iodine. The total amount of iodine in a given environmental media (i.e., pasture grass, leafy vegetables, and soil) is obtained by summing the contributions from deposition of each physico-chemical form.

A.1.3 Chemical Forms of Iodine in Air

Iodine was released from the ICPP essentially in elemental (or molecular) form (I_2). This form of iodine is chemically reactive and during the atmospheric transport, it interacts with molecules in the air forming organic iodine compounds (e.g., CH_3I or methyl iodine). Also, elemental iodine attaches to atmospheric aerosols. The iodine chemistry in the atmosphere is complex, but some important aspects are presented below:

- Elemental iodine (I_2) can photodissociate rapidly in the presence of sunlight with a half-time of 20 s.

- Methyl iodide can photodissociate in the atmosphere in the presence of sunlight with a half-time of 64 h, which indicates that only the iodine released in elemental form undergoes significant photodissociation.

- Knowledge about the formation of methyl iodide from elemental iodine is important for this study because, as discussed below, measurements have indicated that this transformation does occur in the atmosphere.

In addition to chemical transformations, all forms of I-131 undergo radioactive decay with a half-life of 8.04 d. For travel times of a few hours, within which most of the chemical transformations occur, removal of I-131 by radioactive decay would be insignificant.

An experimental study was conducted at Hanford (Ludwick 1964) in which iodine was released into the atmosphere in elemental form. Measurements of different forms of iodine in the atmosphere indicated that beyond a distance of roughly 3 km, 30% of iodine was in particulate form, 36% was in organic form, and the remaining 34% was in the elemental form. Ludwick (1967) also used stack gas measurements and measurements 5 miles downwind of the stack to estimate that the original iodine in elemental form partitioned into 15%, 43%, and 42% particulate, organic, and elemental forms, respectively. Ramsdell et al. (1994), after a review of several papers, concluded that the partitioning of iodine into different forms at 3.2 km in Ludwick's experiments (Ludwick 1964) was consistent with the results of other measurements of iodine in the plumes from other stacks at the Hanford site (Ludwick 1967, Perkins 1963 and 1964), with the partitioning of iodine in the plume following the Chernobyl accident (Aoyama et al. 1986, Bondietti and Brantley 1986, Cambray et al. 1987, Mueck 1988), and with the partitioning of natural iodine in the atmosphere (Voilleque 1979).

In this study, the fraction of iodine in different physico-chemical forms was assumed to be independent of distance from the source, but those fractions were assumed to be uncertain. The fraction of iodine in elemental form was described by a uniform probability distribution between 30% and 50% (with an average of 40%), while the fraction of iodine attached to particles was assumed to vary uniformly from 5% to 45% (with an average of 25%; Table A.1). The fraction of iodine in organic form was calculated as 100% minus the sum of the amounts of the other two chemical forms, and was found to range from 10% to 60% (with an average of 35%). These uncertainty ranges are consistent with those used in other studies (Nair et al. 2000).

The concentration of "total" iodine (i.e., in all physico-chemical forms; C_a in Bq m^{-3}) was estimated at any given location by using atmospheric dispersion models (Radonjic et al. 2005). The concentration of a specific physico-chemical form k of iodine in air was estimated by multiplying C_a by the fractions of iodine in each physico-chemical form defined above.

$$C_{a,k} = C_a \times F_k \qquad \text{(A.4)}$$

where,

$C_{a,k}$	=	the concentration of physico-chemical form k of I-131 in air [Bq m^{-3}_{air}]
C_a	=	the concentration of I-131 in air [Bq m^{-3}_{air}]
$F_{,k}$	=	the fraction of I-131 that in physico-chemical form k [unitless]

The index for the physico-chemical form of iodine is $k = e$ for the elemental form, $k = p$ for the particulate form, and $k = o$ for the organic form.

The concentrations of I-131 in air for each physico-chemical form of iodine are used in Equations A1 through A.3 to determine concentrations of iodine in soil and vegetation (for each

physico-chemical form). The concentrations of "total" iodine (i.e., all forms) in soil and vegetation are obtained by summing the concentrations for each iodine form.

Table A.1 Assumed fraction of iodine-131 in different physico-chemical forms

Physico-chemical form	Fraction
Elemental (F_e)	Uniform (min=0.3, max=0.5)
Particulate (F_p)	Uniform (min=0.05, max=0.45)
Organic (F_o)	$1 - (F_e + F_p)$

A.1.4 Parameters Used to Model the Transfer from Air to Vegetation and Soil

A.1.4.1 Dry Deposition and Interception

The combination of processes by which an airborne contaminant is transferred to vegetation without being carried by rain or other precipitation is referred to as *dry deposition.* In Equations A.1 and A.2, dry deposition onto vegetation is described by the total deposition velocity (V_d) and by the mass interception factor $(r/Y)_{dry}$. The *total* dry deposition velocity relates the deposition flux [Bq m^{-2} s^{-1}] to the concentration in air [Bq m^{-3}], and accounts for the total deposition per unit time per unit area, including vegetation, detritus, the root mat, and soil. However, many measurements (Chamberlain and Chadwick 1953 and 1966, Chamberlain 1960, Heinemann and Vogt 1980) were performed for the *vegetation* dry deposition velocity (V_v), which accounts for the deposition only to the vegetation. The *vegetation* dry deposition velocity (V_v) is given by the product of the *total* dry deposition velocity (V_d) and the interception fraction (r), which expresses the fraction of the net flux to the ground that is retained on the vegetation (Equation A.6):

$$V_v = V_d \cdot r \tag{A.5}$$

Furthermore, the *vegetation* dry deposition velocity (V_v) can be normalized to the biomass of the vegetation (Y). The *normalized* dry deposition velocity (V_D) is given by the product of the *total* dry deposition velocity (V_d) and the mass interception factor $(r/Y)_{dry}$ (Equation A.7):

$$V_D = V_d \cdot \left(\frac{r}{Y}\right)_{dry} = V_v \cdot \left(\frac{1}{Y}\right) \tag{A.6}$$

The *vegetation* dry deposition velocity (V_v) is generally determined from field experiments as the total deposition on grass (Bq m^{-2}) divided by the time-integrated concentration in air at the location of the deposition (Bq s m^{-3}), and the normalized deposition velocity (V_D) is determined using the biomass of the vegetation (Y). Normalized deposition velocity (V_D) is a better quantity than total dry deposition velocity (V_d) for comparing measurements performed at different sites. Similarly, the mass interception fraction $(r/Y)_{dry}$ is a more stable quantity than the biomass (Y). While the biomass (Y) changes with season, or is different from site to site, $(r/Y)_{dry}$ shows less variation.

The results of a set of measurements designed to determine iodine deposition parameters have been reported by Vogt et al. (1976) and Heinemann and Vogt (1980). Another extensive set of iodine experiments was performed at the INEL in the 1960s (i.e., the Controlled Environmental Radioiodine Tests – CERT) (Bunch 1966 and 1968, Hawley et al. 1964). Elemental and organic iodine were released under controlled conditions on the INEL site and the deposition velocities were determined during all seasons of the year and for different conditions of the grass (e.g., dry grass, irrigated green and growing grass, freshly mowed grass, etc.). These experiments provide the most representative data set for the present study.

In this study, the mass-normalized deposition velocity (V_D; $cm^3\ g^{-1}\ s^{-1}$) measured in the CERT studies was used for elemental iodine (Figure A.1). Dry deposition of particulate and organic iodine was modeled using the *total* dry deposition velocity (V_d; $cm\ s^{-1}$) and the mass interception factor (r/Y_{dry}; $m^2\ kg^{-1}_{dry\ mass}$) derived from other studies (Vogt et al. 1976, Heinemann and Vogt 1980, Apostoaei et al. 1999). Table A.2 presents the probability distributions of the deposition velocities used in this study. The mass interception factors for the particulate and organic forms of iodine are presented in Table A.3. A mass interception fraction for elemental iodine is not used, since dry deposition for this form of iodine is modeled using the measured mass-normalized deposition velocity (V_D; $cm^3\ g^{-1}\ s^{-1}$).

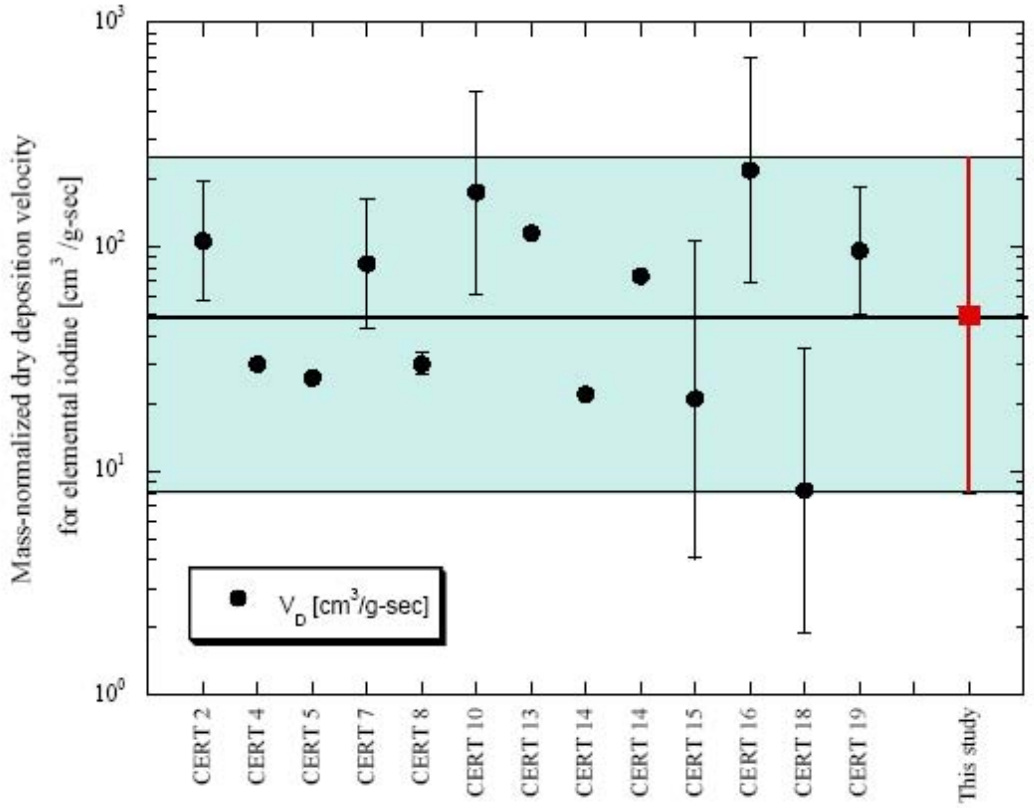

Figure A.1 The mass-normalized dry deposition velocity for elemental iodine as derived from the CERT studies performed on the Idaho National Engineering Laboratory site (Bunch 1968), and the deposition velocity used in this study

Table A.2 Assumed probability distributions for the *total* or mass-normalized dry deposition velocity (V_d or V_D)

Chemical form of I-131			V_d [cm s^{-1}] or V_D [cm^3 g^{-1} s^{-1}]			
			Lower	Upper	Mode	Distribution
Elemental	V_D	[cm^3 g^{-1} s^{-1}]	8	250	50	log-triangular
Particulate	V_d	[cm s^{-1}]	0.05	0.5	0.15	log-triangular
Organic	V_d	[cm s^{-1}]	0.001	0.05	0.0071	log-triangular

Table A.3 Assumed probability distributions for the mass interception factor $(r/Y)_{dry}$ for dry deposition

Chemical form of I-131	$(r/Y)_{dry}$ [m^2 kg^{-1} (dry mass)]*			
	Lower	Upper	Mode	Distribution
Particulate	0.5	4.0	2.3	triangular
Organic	0.8	4.0	2.4	triangular

* Source: Apostoaei et al. 1999

A.1.4.2 Wet Deposition

The combination of processes by which a contaminant present in air is transferred to vegetation by rainfall is referred to as *wet deposition*. The mechanisms which contribute to wet deposition are in-cloud and below-cloud scavenging of contaminants by raindrops. The contaminant is first transferred to a rain droplet, then mixed within the rain droplet, and finally deposited on plant and soil surfaces.

A monthly averaged deposition rate of iodine (Bq m^{-2} d^{-1}) is estimated by multiplying the monthly averaged concentration of iodine in air (Bq m^{-3}) and the wet deposition velocity (m d^{-1}) defined by Equation A.7.

$$V_w = WR \cdot R \qquad \text{(A.7)}$$

WR = the washout ratio [m^3_{air} L^{-1}_{rain}] (see A.1.4.2.2)

R = the monthly average precipitation rate [mm d^{-1}]

A.1.4.2.1 Monthly Average Precipitation Rate

Average precipitation rates in each month have been derived from data reported by the National Climatic Data Center[1] for five locations near the INEL site for 1957-1959. A triangular distribution defined by the minimum, mode, and maximum values given in Table A.4 was

[1] Original precipitation data retrieved from http:// www.ncdc.noaa.gov/oa/climate/climateproducts.html (accessed in October 2003).

assumed for each month. The probability distributions are applied to each year of the study period (i.e., 1957, 1958, and 1959).

Table A.4 Monthly averaged precipitation rate expected near the Idaho National Engineering Laboratory site

Month*	Monthly-averaged precipitation rate [mm d^{-1}]		
	Minimum	Mode	Maximum
January	0.24	0.67	1.1
February	0.20	0.73	1.4
March	0.06	0.52	1.0
April	0.0085	0.77	1.5
May	0.33	1.5	3.6
June	0.22	0.80	1.6
July	0	0.28	0.95
August	0	0.50	2.2
September	0	0.62	1.9
October	0	0.25	0.84
November	0	0.36	1.5
December	0.19	0.66	0.98

* During winter months, part of the precipitation is snow. During snowstorms, deposition on vegetation and soil has minimal contribution to the dose from iodine, because cows do not consume fresh grass (see also Section A.2.2.2 and Table A.9).

A.1.4.2.2 Washout Ratio

Transfer of iodine to rain water is described by a *washout ratio* (*WR*, $m^3_{air}\ L^{-1}_{rain}$), defined as the ratio of the concentration of iodine in the rainwater (Bq L^{-1}_{rain}) to the concentration in air (Bq m^{-3}_{air}) at ground level. During a rain event, the washout ratio depends on the rain intensity (mm d^{-1}). The larger the rain intensity, the smaller the washout ratio (Slinn 1978, NRC/CEC 1994). The washout ratio can be expressed as follows:

$$WR = WR^{d}_{1\ mm} \cdot I^{-s} \qquad \text{(A.8)}$$

where

$WR^{d}_{1\ mm}$ = the washout ratio for a rain event with a 1 mm d^{-1} rain intensity

I = the precipitation intensity (i.e., precipitation rate per rain event) defined as the amount of rain divided by the total duration of rain [mm d^{-1}]

s = the "rain exponent," which is an empirical parameter [unitless] that accounts for the observed decrease in the washout ratio with increasing intensity of precipitation

As discussed below, probability distributions were derived for WR^{d}_{1} for each chemical form of iodine, and precipitation intensity was estimated for each month during 1957-1959.

Washout Ratio for Elemental Iodine (I_2)

The washout ratio for a gas is a measure of the solubility of the gas; as stated by Slinn (1978), essentially all raindrops attain their equilibrium concentration of a gas in less than 10 m of fall.

The probability distribution of WR^d_1 for the elemental form of I-131 was assumed to be log-triangular with a minimum value of 2,100 $m^3_{air}\,L^{-1}_{rain}$, a mode of 5,000 $m^3_{air}\,L^{-1}_{rain}$, and a maximum value of 10,700 $m^3_{air}\,L^{-1}_{rain}$ (Table A.5). This range is based on a review of NCRP recommendations (NCRP 1993), on an expert elicitation performed for the U.S. Nuclear Regulatory Commission and the European Community (NRC/CEC 1994), and studies by the NCI (1997) and Slinn (1978).

Washout Ratio for Particulate Iodine

The physical diameter of the particles to which iodine attaches is assumed to be very small (d < 1 μm). Particles may be removed by rain both during the formation of rain in the cloud and by scavenging below the cloud; the process is highly dependent on the rain event (type, rate, drop size, etc.).

The probability distribution of WR^d_1 for particulate I-131 is assumed to be log-triangular between 1,000 and 5,400 $m^3_{air}\,L^{-1}_{rain}$, with a mode of 2,500 $m^3_{air}\,L^{-1}_{rain}$ (Table A.5). This range is based on measurements made in the U.S. in the aftermath of the Chernobyl accident and summarized by Richmond et al. (1988) and an expert elicitation performed for the U.S. Nuclear Regulatory Commission and the European Community (NRC/CEC 1994).

Washout Ratio for Organic Iodine

The probability distribution of WR^d_1 for organic forms of I-131 is assumed to be log-triangular between 4.0 and 18.0 $m^3_{air}\,L^{-1}_{rain}$ with a mode of 8 $m^3_{air}\,L^{-1}_{rain}$ (Table A.5). This distribution is based on the expert elicitation performed for the U.S. Nuclear Regulatory Commission and the European Community (NRC/CEC 1994) and on studies by the NCI (1997) and Slinn (1978).

Table A.5 Assumed probability distributions of washout ratios for iodine ($m^3_{air}\,L^{-1}_{rain}$)

Form of Iodine	Washout Ratio ($m^3_{air}\,L^{-1}_{rain}$)			
	Distribution	Minimum	Mode	Maximum
Elemental	Log-triangular	2,100	5,000	10,700
Particulate	Log-triangular	1,000	2,500	5,400
Organic	Log-triangular	4	8	18

Precipitation Intensity

The monthly averaged precipitation intensity (rate of rain per rain event) has been derived from precipitation data reported by the National Climatic Data Center for five locations near the INEL

site for 1957-1959 (see footnote one for the source of data). A triangular distribution defined by the minimum, mode, and maximum values given in Table A.6 was assumed for each month. The probability distributions are applied to each year of the study period (i.e., 1957, 1958, and 1959).

Rain Exponent

The rain exponent of 0.7 used by NCI (1997) is slightly larger than the $s \cong 0.4$ values reported by Slinn (1978) and NRC/CEC (1994). A literature review for the rain exponent is summarized in Table A.7.

The rain exponent (s) was also considered an uncertain variable. Based on the available information (Table A.7), a relatively large range (0.2 - 0.6, central value 0.4; uniform distribution) was chosen for this parameter.

Table A.6 Assumed precipitation intensity by month near the Idaho National Engineering Laboratory site

Month*	Precipitation intensity [mm d^{-1}]		
	Minimum†	Mode	Maximum
January	0.016	2.7	8.4
February	0.043	2.8	10
March	0.27	2.3	5.6
April	0	3.3	11
May	0.24	4.4	15
June	0.073	4.6	14
July	0	3.0	13
August	0.032	3.6	21
September	0	4.7	24
October	0	3.7	11
November	0	3.0	7.1
December	0.17	3.0	12

* During winter months, part of precipitation can be snow. During snowstorms, deposition on vegetation and soil has minimal contribution to the dose from iodine, because cows do not consume fresh pasture grass.

† In some months, no precipitation was observed around INEL, which leads to a zero rain intensity as a minimum value. However, the data in the table is used to simulate the precipitation intensity as the amount of rain per rain event, which by definition cannot be zero. Monte Carlo or Latin Hypercube samples from a triangular probability distribution function that has a minimum equal to zero are never zero. That is, the simulated precipitation intensity (the amount of rain per rain event) can be very low, but it is never zero.

Table A.7 Summary of literature review on the rain exponent (*s*)

Form of iodine	s	Reference
Elemental	0.4	Brenk and Vogt (1981)
Aerosols	0.5	Brenk and Vogt (1981)
Elemental	0.4	Schwarz (1985) cited by NRC/CEC (1994)
Methyl	0.4	Schwarz (1985) cited by NRC/CEC (1994)
Particulate	0.3 ± 0.12	NRC/CEC (1994); Expert B
Gaseous	0.3	NRC/CEC (1994); Expert C
Particulate	0.25	NRC/CEC (1994); Expert G
Particulate	0.25	NRC/CEC (1994); Expert H

A.1.4.2.3 Interception and Initial Retention by Pasture Vegetation

The transfer of iodine carried by water droplets to vegetation surfaces is described in this work by the mass interception factor $(r/Y)_{wet}$ [m^2 $kg^{-1}_{dry\ mass}$], defined as the fraction of material in rain deposited per square meter of the ground surface that is intercepted and retained on the plant, normalized to the dry mass of the vegetation per unit area of soil. For I-131, such factors have been experimentally determined using simulated rainfall under field conditions by Hoffman et al. (1992).

Iodine is present in rainwater in ionic form, and also as particles or organic and inorganic forms dissolved in water. The mass interception factor for wet deposition is a function of the biomass *Y*, the rainfall rate *R*, the amount of rain during the deposition process (i.e., during the rain event), and the vegetation type (Hoffman et al. 1992). Negative ionic forms of iodine are not readily retained by the leaves of the plant, because the plant surface is usually negatively charged. On the other hand, positive ions, particles, and neutral molecules are more readily retained on the plant.

In the case of wet deposition of iodide, biomass and vegetation type are of minor importance in comparison to the intensity of rain (Hoffman et al. 1992).

In the case of wet deposition of insoluble particles in rainwater, retention is higher than for soluble compounds, and rain intensity and vegetation biomass are of about the same importance.

The experiment described by Hoffman et al. (1992) determined that the mass interception factor depends strongly on the amount of rain per rain event.

$$\left(\frac{r}{Y}\right) = \left(\frac{r}{Y}\right)_{1mm} \cdot I^{K} \qquad \text{(A.9)}$$

where

$(r/Y)_{1\ mm}$	=	the initial value of the mass interception factor a precipitation amount of 1 mm [$m^2\ kg^{-1}_{dry\ mass}$ per mm d^{-1}]
I	=	the precipitation amount per rain event [mm d^{-1}]
K	=	the coefficient of the functional fit for increasing amounts of precipitation [unitless]

For iodide, the value for the mass interception factor $(r/Y)_{1\ mm}$ for a unit rain amount was measured to range between 1 and 4 $m^2\ kg^{-2}_{dry\ mass}$ (Hoffman et al. 1992). A uniform distribution was assigned to represent the uncertainty in this parameter. The exponent describing the dependence on the rain amount has a value of K_{iodide} = -0.909 (Hoffman et al. 1992), indicating that for soluble forms of iodine, the mass interception factor decreases nearly in proportion to the amount of rain per rain event.

Iodine released from INEL became attached to small aerosol particles during atmospheric transport. Thus, out of the entire set of experiments performed by Hoffman et al. (1992), measurements using 3-μm insoluble microspheres are considered the most appropriate to describe wet interception of iodine attached to particles. The mass interception factor $(r/Y)_{1\ m}$ for a unit rain amount for particulates is found to range between 2 and 6 $m^2\ kg^{-1}_{dry\ mass}$ (Hoffman et al. 1992). Uncertainty in this parameter is expressed using a uniform distribution. The exponent relating the mass interception factor to the rain amount was found by Hoffman et al. (1992) to be $K_{particulate}$ = -0.207, indicating only a moderate decrease with increasing amounts of precipitation.

Elemental Iodine

Atmospheric elemental iodine is probably found in rainwater in either anionic or molecular form. The concentration of the molecular dissolved form is assumed to be negligible in comparison with the ionic forms; that is, elemental iodine from the atmosphere is transferred in rainwater as iodide. Therefore, the effective mass interception factor for atmospheric elemental iodine is equal to the mass interception factor measured for iodide by Hoffman et al. (1992).

$$\left(\frac{r}{Y}\right)_{wet}^{elemental} = \left(\frac{r}{Y}\right)_{iodide} \tag{A.10}$$

Particulate Iodine

An important fraction of iodine trapped to the particles in a water droplet will be released into solution as ions. The fraction of iodine that is still trapped on the particles is denoted as "*d*," and an effective mass interception factor is calculated as follows.

$$\left(\frac{r}{Y}\right)_{wet}^{particulate} = d \cdot \left(\frac{r}{Y}\right)_{particles} + (1-d) \cdot \left(\frac{r}{Y}\right)_{iodide} \tag{A.11}$$

The fraction "*d*" of iodine still trapped on the particles was also treated as an uncertain variable in order to account for the lack of knowledge about the value of this parameter. The selected range for *d* is 0.1–0.5, and the selected probability distribution is uniform.

Organic iodine

Dissolution is assumed to be the only process by which atmospheric organic iodine is transferred into rainwater. Dissolved organic iodine is assumed to be retained on the plant surfaces as efficiently as iodide.

$$\left(\frac{r}{Y}\right)_{wet}^{organic} = \left(\frac{r}{Y}\right)_{iodide} \tag{A.12}$$

A.1.4.3 Weathering from Vegetative Surfaces

After radionuclides are deposited on vegetation, removal processes combine with radioactive decay to reduce the initially retained quantity. Reduction in the initial concentration by cuticle sloughing, growth dilution, wind and water, or grazing by insects and larger herbivores are some of the processes that produce the effect often referred to as "weathering." These factors are combined in the following expression for the effective removal rate constant λ_{eff} (d^{-1}), presented also as a function of the effective half-time (T_{eff}).

$$\lambda_{eff} = \lambda_R + \lambda_w = 0.693/T_{eff} \tag{A.13}$$

where

λ_R = radiological decay constant [d^{-1}] = $0.693/T_R$

λ_w = removal rate from vegetation due to weathering [d^{-1}] = $0.693/T_w$

$$T_{eff} = \frac{T_w \cdot T_r}{T_w + T_r} \tag{A.14}$$

T_w = the weathering half-time

T_r = the radioactive half-life

The radiological half-life of I-131 is 8.04 d (ICRP 1983). Experimental values of the weathering half-time are summarized by Miller and Hoffman (1983), Mück et al. (1994) and the IAEA (1996).

A uniform probability distribution was selected for the weathering half-time of I-131 on pasture grass with a range of 6 to 17 days. This choice gives an effective half-time (T_{eff}) for I-131 of 4.7 days with a 95% confidence range from 3.5 to 5.4 days.

A.1.4.4 Dry-to-Fresh Weight Conversion

The dry-weight-to-fresh-weight conversion factor for leafy vegetables measures the amount of dry matter present in vegetation [$kg_{dry}\ kg^{-1}_{wet}$]. For instance, a conversion factor of 0.1 [$kg_{dry}\ kg^{-1}_{wet}$] indicates that vegetation contains 10% dry-matter and 90% water. Measurements for this parameter vary over a large range, with values as high as 0.35 [$kg_{dry}\ kg^{-1}_{wet}$]. However, this parameter is used to estimate the concentration of I-131 in fresh leafy vegetables, which should contain enough water to be considered "fresh." Thus, a uniform probability distribution between 0.05 and 0.09 [$kg_{dry}\ kg^{-1}_{wet}$] was assumed for this parameter. This range is similar to results of the literature review by Snyder et al. (1994).

A.1.4.5 Transfer to Soil

Estimation of the transfer of I-131 to humans is primarily based on the ingestion of milk and meat from cows and goats that have consumed contaminated pasture vegetation. However, this study has included the contribution of I-131 in the animal diet due to soil ingestion by cows and goats. Also, this study accounts for inadvertent ingestion of soil by humans (See section A.3.6). This section discusses the parameters used to estimate the concentration of I-131 in soil.

Depth of Surface Soil

Iodine-131 deposited on the ground mixes almost instantaneously with soil within the first 1-mm layer from the surface,[2] and rapidly with soil within the first 1 cm from the surface. Ingestion of soil by grazing animals and humans involves mostly soil from the first 1-mm layer of the surface, but consumption of soil from the top 1 cm of soil is also possible (see Stevens et al. 1992 and Whicker and Kirchner 1987 for soil ingestion by animals, and Simon 1998 for soil ingestion by humans).

To estimate the concentration is soil, this study uses a probability distribution function for the depth of the surface soil described by a 40% weight to 0.001 m (i.e., 1 mm) and a 60% weight to a log-uniform probability distribution with a range from 0.001 m (1 mm) to 0.01 m (1 cm).

Soil Density

The soil density used in this study is based on an analysis of 222 agricultural soils in the United States by Baes and Sharp (1983). The distribution of pasture soil density may be described by a lognormal distribution with a geometric mean of 1.35 g cm^{-3} and a geometric standard deviation of 1.2.

Removal from Soil

Initial concentration of radionuclides in soil is reduced by soil erosion, surface runoff, mixing with uncontaminated soil, and downward migration by leaching, as well as by radioactive decay. These processes are modeled using an effective removal rate constant $\lambda_{eff\ soil}$ (d^{-1}). As for

[2] Soil does not have a smooth surface, due to the size of soil particles. Mixing of iodine in the first 1-mm layer also accounts for soil roughness.

removal from vegetative surfaces, the effective removal rate is the sum of the radiological decay constant λ_R (d^{-1}) and a removal rate from soil due to processes other than radioactive decay λ_S (d^{-1}).

IAEA (2001) indicates a removal rate of iodine from the 0-15 cm root zone of 0.0014 d^{-1} (in the absence of radioactive decay), which is equivalent to a half-time of about 500 days. Removal from the top 1 cm of soil is probably faster than a half-life of 500 days, but it is still much slower than the radiological half-life of I-131 (8.04 days). In this study, it was assumed that I-131 removal rate from soil due to processes other than radioactive decay is negligible compared to the radiological decay rate. Thus, $\lambda_{eff,\,soil} = \lambda_R + \lambda_S \cong \lambda_R = 0.0862\ d^{-1}$.

A.1.4.6 Summary of Model Parameters

A summary of assumed probability distributions for the input parameters used to estimate the transfer of I-131 from air to vegetation and soil is presented in Table A.8.

Table A.8 Assumed probability distributions for parameters used to estimate the transfer of iodine-131 from air to vegetation and soil

Parameter	Symbol	Physico-chemical form of iodine	Units	Distribution			
				Min.	Max.	Mode	Shape
Dry deposition velocity							
normalized	V_D	Elemental*	[$cm^3\ g^{-1} s^{-1}$]	8	250	50	log-triangular
total	V_d	Particulate*	[$cm\ s^{-1}$]	0.05	0.5	0.15	log-triangular
total	V_d	Organic*	[$cm\ s^{-1}$]	0.001	0.05	0.0071	log-triangular
Mass interception factor for dry deposition	$(r/Y)_{dry}$	particulate	[$m^2\ kg^{-1}$ dry]	0.5	4	2.3	triangular
		organic	[$m^2\ kg^{-1}$ dry]	0.8	4	2.4	triangular
Washout ratio per unit precipitation intensity	$WR^d_{1\ mm}$	elemental	[$m^3_{air}\ L^{-1}_{rain}$]	2100	10700	5000	log-triangular
		particulate	[$m^3_{air}\ L^{-1}_{rain}$]	1000	5400	2500	log-triangular
		organic	[$m^3_{air}\ L^{-1}_{rain}$]	4	18	8	log-triangular
Rain exponent	s	all species	[unitless]	0.2	0.6		uniform
Mass interception factor for wet deposition	$(r/Y)_{1\ mm}$	as iodide	[$m^2\ kg^{-1}_{dry}$]	1.0	4.0		uniform
		as particulate	[$m^2\ kg^{-1}_{dry}$]	2.0	6.0		uniform
Coefficient of the power function fit for wet interception	K	iodide	[unitless]			- 0.909	fixed value
		particulate	[unitless]			- 0.207	fixed value

Table A.8 Assumed probability distributions for parameters used to estimate the transfer of iodine-131 from air to vegetation and soil (continued)

Parameter	Symbol	Physico-chemical form of iodine	Units	Distribution			
				Min.	Max.	Mode	Shape
Fraction of iodine trapped on dissolved particles	d	--	[unitless]	0.1	0.5		uniform
Monthly average precipitation rate	R	--	[mm d^{-1}]	Refer to Table A.4			
Precipitation intensity†	I	--	[mm d^{-1}]	Refer to Table A.5			
Weathering half-time on vegetation	T_w	all species	[d]	6	17		uniform
Time period of exposure of the standing crop biomass	t_{ag}	all species	[d]	10	45		uniform
Dry-to-fresh weight relationship	γ	all species	[unitless]	0.05	0.09		uniform
Surface soil depth consumed by animals	SD	--	[m]	0.001	0.01	0.001	40% weight; 60% weight to LU‡
Soil density	ρ_s	--	[kg m^{-3}]			1350§ (1.2)	lognormal
Removal half-time from soil	T_S	all species	[d]			∞	see text for details

* Elemental = reactive form of iodine; particulate = iodine attached to aerosols; organic = nonreactive form of iodine
† Amount of rain per rain event
‡ 40% weight to a value of 0.001 m (i.e., 1 mm) and a 60% weight to a log-uniform (LU) probability distribution with a range from 0.001 m (1 mm) to 0.01 m (1 cm).
§ Geometric mean (geometric standard deviation)

A.2 Transfer from Pasture and Soil to Food Products

Contamination deposited onto pasture or ground surface is ingested by grazing animals (i.e., cows, goats) and by chickens and is transferred to animal food products, such as milk, meat, and eggs. In addition, lactating women who consume I-131-contaminated foods will produce breast milk containing I-131.

Detailed measurements of the concentrations of I-131 in food products are not available for the time periods during which I-131 was released from the INEL site. As a result, mathematical models must be used to estimate the transfer of I-131 from pasture and soil to food products. This section discusses the approach used in the present study for estimating concentrations of I-131 in cow's and goat's milk, meat, eggs, and mother's milk. The modeling approach and parameters for the transfer from pasture and soil to cow's milk, goat's milk, beef, and eggs are presented in Sections A.2.1 and A.2.2. The transfer to mother's milk is described in Section A.2.3.

A.2.1 Description and Assumptions for Modeling the Transfer from Pasture and Soil to Milk, Meat and Eggs

This study examines the transfer of I-131 to milk of cows and goats, and to meat products and eggs. Two general categories of cows are considered in the present study. The first, a "commercial cow," is normally raised and managed in a herd, with milk (and meat) sold commercially. The second, a "backyard cow," is a cow raised by its owner primarily for at-home consumption of dairy products. Differences between the two types of cows include the amount of pasture consumed and, in the case of dairy cattle, the amount of milk produced. Small local commercial dairy herds are assumed to have been located on farms distributed throughout the entire INEL region. Large commercial dairies, however, were located within 10 miles of the portion of Snake River passing through the INEL region (see Section 6). Milk from local commercial dairies is pooled for retail sale locally, while milk from the large dairies was distributed throughout the entire INEL region. On the other hand, family-owned (or backyard) cows are assumed to have grazed entirely on local pastureland and milk from these cows was consumed by family members and neighbors.

Estimation of the transfer of I-131 to milk and meat is based primarily on ingestion of I-131-contaminated pasture vegetation by grazing animals. However, this study has also included ingestion of I-131 in soil by cows or goats. It was assumed that inhalation of I-131 by cows or goats is negligible in comparison with ingestion of I-131 in contaminated feed and soil. Chickens' diet consists mostly of grains, which are stored for long periods of time allowing for I-131 to decay. However, free-ranging chickens consume small amounts of fresh pasture grass, which was assumed to be the main source of I-131 for transfer to eggs.

The following equations are used to estimate the transfer of I-131 from ingested feed or soil to milk of cows or goats, meat, or eggs. The time-integrated concentrations of I-131 in pasture grass and soil for each month during 1957-1959 are estimated according to Equations A.15 and A.16, respectively:

$$TIC_p = C_p \cdot ED \quad \text{(A.15)}$$

$$TIC_s = C_s \cdot ED \quad \text{(A.16)}$$

where

TIC_p	=	the time-integrated concentration of iodine in pasture grass [Bq d $kg^{-1}_{dry\ mass}$]
C_p	=	the average concentration on pasture grass per month [Bq $kg^{-1}_{dry\ mass}$] (see Equation A.1)
TIC_s	=	the time-integrated concentration of iodine in soil [Bq d kg^{-1}]
C_s	=	the average concentration in soil per month [Bq kg^{-1}] (see Equation A.3)
ED	=	exposure duration; the number of days per month [d $month^{-1}$]

The time-integrated concentration of I-131 in milk is estimated using Equation A.17:

$$TIC_m = \left[\left(TIC_p \cdot Q_{mp} \cdot p_m\right) + \left(TIC_s \cdot Q_{ms}\right)\right] \cdot F_m \quad \text{(A.17)}$$

TIC_m	=	the time-integrated concentration of iodine in milk [Bq d L^{-1}_{milk}] at time of milking
TIC_p	=	the time-integrated concentration of iodine in pasture grass [Bq d $kg^{-1}_{dry\ mass}$] (see Equation A.15)
Q_{mp}	=	the pasture ingestion rate for dairy cows or goats [$kg_{dry\ mass}$ d^{-1}]
p_m	=	the fraction of feed that is contaminated for dairy animals [unitless]
TIC_s	=	the time-integrated concentration of iodine in soil [Bq d kg^{-1}] (see Equation A.16)
Q_{ms}	=	the soil ingestion rate for cow or goat [$kg_{dry\ mass}$ d^{-1}]
F_m	=	the feed-to-milk transfer coefficient for cow or goat [d L^{-1}_{milk}]

This equation applies to milk from backyard cows, commercial cows, and goats. The values of the parameters are different for each animal, as discussed in the following sections.

The concentration of I-131 in the muscle of beef cattle is estimated using the following equation:

$$TIC_f = \left[\left(TIC_p \cdot Q_{fp} \cdot p_f\right) + \left(TIC_s \cdot Q_{fs}\right)\right] \cdot F_f \quad \text{(A.18)}$$

where

TIC_f = the time-integrated concentration of iodine in beef [Bq d kg^{-1}_{beef}] at time of slaughter

TIC_p = the time-integrated concentration of iodine in pasture grass [Bq d $kg^{-1}_{dry\ mass}$] (see Equation A.15)

Q_{fp} = the pasture ingestion rate for beef cows [$kg_{dry\ mass}$ d^{-1}]

p_f = the fraction of feed that is contaminated for beef cows [unitless]

TIC_s = the time-integrated concentration of iodine in soil [Bq d kg^{-1}] (see Equation A.16)

Q_{fs} = the soil ingestion rate for beef cows [$kg_{dry\ mass}$ d^{-1}]

F_f = the feed-to-beef transfer coefficient for cows [d kg^{-1}]

The concentration of I-131 in eggs is estimated using the following equation:

$$TIC_e = \left(TIC_p \cdot Q_{ep} \cdot p_e\right) \cdot F_e \qquad \text{(A.19)}$$

where

TIC_f = the time-integrated concentration of iodine in the content of eggs [Bq d $kg^{-1}_{egg\ content}$] at time of collection

TIC_p = the time-integrated concentration of iodine in pasture grass [Bq d $kg^{-1}_{dry\ mass}$] (see Equation A.15)

Q_{ep} = the pasture ingestion rate for chickens [$kg_{dry\ mass}$ d^{-1}]

p_e = the fraction of feed that is contaminated for chickens [unitless]

F_e = the feed-to-egg content transfer coefficient [d kg^{-1}]

A.2.2 Parameters Used to Model the Transfer from Pasture and Soil to Milk, Meat and Eggs

The ranges of input parameters and the shapes of the subjective probability distributions used to estimate the transfer of I-131 from pasture and soil to milk, beef, and eggs were selected on the basis of a review of the literature. The rationales for the choices of specific parameter values are presented in the following sections.

A.2.2.1 Intake Rates for Cows, Goats, and Chickens

Cows

In the United States, management of ruminant herbivores includes various practices. Backyard cows are allowed to graze on open pastures during the entire grazing season (uncontrolled

grazing). On the other hand, pasture management for commercial dairy cows includes strip and rotational grazing, in which animals are moved two to six times from one pastureland to another during the grazing season. For both backyard and commercial cows, stored feed is usually provided to complement their diet of fresh pasture grass. More stored feed is provided during the winter season.

The feed intake rates (Q_m) for dairy cows, beef cattle were selected based on the recommendations of Miller (1996) and on studies by Koranda (1965) and NCI (1997). Other significant literature was also reviewed (Hoffman and Baes 1979, Schwarz and Hoffman 1980). The intake rates, expressed in kilograms of dry mass per day, apply for every day of the year.

For commercial cows, Koranda (1965) estimated a feed intake rate of 11.8 $kg_{dry\ mass}\ d^{-1}$ for dairy cows managed in strip or rotational grazing systems. For dairy cows grazing on open pastures (backyard cows), Koranda (1965) reported an average ingestion rate of 9.1 $kg_{dry\ mass}\ d^{-1}$.

Backyard cows were assumed to graze freely on open pastures. The selected range for the daily intake rate for backyard cows is from 7 to 14 $kg_{dry\ mass}\ d^{-1}$. A triangular distribution with a mode of 9 $kg_{dry\ mass}\ d^{-1}$ was assigned.

This study assumes that beef cattle were also allowed to graze freely, but stored feed was provided as a supplement during the entire year. The ingestion rate of feed for beef cattle (Q_f) was thus assumed to be the same as the ingestion rate of backyard cows.

The selected range for the daily intake rate for commercial cows is from 10 and 18 $kg_{dry\ mass}\ d^{-1}$ (Koranda 1965). A triangular distribution with a mode of 12 $kg_{dry\ mass}\ d^{-1}$ was assigned.

Dairy and beef cattle consume soil at a rate of 4% to 6% of their dry matter intake (Fries et al. 1982, Green and Dodd 1988, Healy 1968, IAEA 1994). The soil intake rate used in this study is 0.5 kg d^{-1}. A triangular probability distribution with a minimum of 0.4 kg d^{-1}, a mode of 0.5 kg d^{-1}, and a maximum of 0.75 kg d^{-1} was used to describe the uncertainty in this parameter.

Goats

Hoffman and Baes (1979) indicate that dairy goats consume feed at rates between 0.7 and 4.0 $kg_{dry\ mass}\ d^{-1}$. The most probable value for the ingestion rate is 2.0 $kg_{dry\ mass}\ d^{-1}$. A triangular distribution with a minimum of 0.7 $kg_{dry\ mass}\ d^{-1}$, a mode of 2.0 $kg_{dry\ mass}\ d^{-1}$, and maximum of 4.0 $kg_{dry\ mass}\ d^{-1}$ was used in this study to describe the uncertainty in the ingestion rate for goats.

Whicker and Kirchner (1987) report a soil ingestion rate for goats of 0.14 kg d^{-1}. IAEA (1994) indicates that consumption of soil by goats is about 10% of their feed intake rate or about 0.2 kg d^{-1}. An informal expert elicitation regarding this parameter was performed as part of this study (Galeriu 2003, Pröhl 2003, Santucci 2003). These experts agreed that, in general, goats ingest more soil per kilogram of feed consumed than cows, and the soil ingestion rate by goats is about 10% of the feed intake rate. In this study, a lognormal probability distribution with a GM = 0.2 kg $d^{-1}$and a GSD = 1.46 is being used in this model to characterize soil ingestion rate for goats. This distribution has a 95% confidence interval of 0.095 to 0.41 kg d^{-1}.

Chickens

According to Ikenberry (1982), the diet of chickens consists of grains and pasture grass (or alfalfa). During the spring and summer, chickens consume 90 g d^{-1} of grain and 5 g d^{-1} pasture (per dry mass). During the fall and winter, they consume the same amount, but they eat alfalfa instead of regular grass (90 g d^{-1} of grain and 5 g d^{-1} alfalfa, dry mass).

For the ingestion rate, Ikenberry (1982) gives 5 grams of grass dry mass per day. This study assumes a uniform distribution from 0 to 10 g d^{-1}, which has a central value of 5 g d^{-1}.

In this study, it is assumed that the amount of I-131 in grain is negligible due to low interception and translocation and a long storage time prior to ingestion. It is also assumed that the concentration of I-131 in pasture grass is equal to the concentration in alfalfa.

A.2.2.2 Fraction of Feed That Is Contaminated

Due to the short half-life of I-131 (8.04 days), feed provided to animals that is stored much longer than 8 days is considered uncontaminated. Ingestion of fresh pasture grass is the most important route by which I-131 is transferred to dairy cows or beef cattle, because fresh grass is consumed without being stored. In the INEL region during winter months (December through February), snow covers the ground and temperatures are often below freezing. During these months, feed provided to cows contains no fresh pasture grass.

This study assumes that animals receive all of their feed from stored sources during the months of January, February, and December. During early spring and late fall there is very little fresh pasture grass available for consumption. For March and November, this study has assumed that up to 30% of feed consumed by backyard cows, goats, and beef cattle, and up to 10% for commercial cows, comes from fresh pasture grass (Table A.9). During April and October, it is assumed that backyard cows, goats, and beef cattle receive from 40% to 70% of their feed from fresh pasture, while commercial cows consume from 20% to 50% of their feed from fresh pasture.

In general, commercial cows are managed for high milk production, so dairy farms do not rely on fresh pasture grass during early spring and late summer. That is, a larger fraction of diet was assumed to be pasture grass for backyard cows than for commercial cows.

The main season for grazing lasts from May to September. During this period, approximately 75% of the feed for a backyard cow is fresh pasture grass. A triangular distribution is assumed, ranging from 60% to 100%, with a central estimate of 75% (Table A.9). During the same period, a typical commercial cow receives slightly less fresh pasture grass (triangular distribution ranging from 40% to 95%, with a mode of 50%).

According to Miller (1996), during grazing season, goats receive stored feed in amounts that represent up to 70% of the total intake. However, it is conceivable that no food supplements are provided, in which case 100% of the feed is contaminated pasture grass. For INEL releases, the selected distribution for the fraction of contaminated feed is uniform between 50% and 100%.

Table A.9 Fraction of feed that is fresh pasture, in different grazing seasons

Month	Backyard Cow	Commercial Cow	Goat	Beef Cattle
January	0	0	0	0
February	0	0	0	0
March	U(0, 0.3)*	U(0, 0.1)	U(0, 0.3)	U(0, 0.3)
April	U(0.4, 0.7)	U(0.2, 0.5)	U(0.4, 0.7)	U(0.4, 0.7)
May	T(0.6, 0.75, 1)†	T(0.4, 0.5, 0.95)	U(0.50, 1)	T(0.4, 0.6, 0.75)
June	T(0.6, 0.75, 1)	T(0.4, 0.5, 0.95)	U(0.50, 1)	T(0.4, 0.6, 0.75)
July	T(0.6, 0.75, 1)	T(0.4, 0.5, 0.95)	U(0.50, 1)	T(0.4, 0.6, 0.75)
August	T(0.6, 0.75, 1)	T(0.4, 0.5, 0.95)	U(0.50, 1)	T(0.4, 0.6, 0.75)
September	T(0.6, 0.75, 1)	T(0.4, 0.5, 0.95)	U(0.50, 1)	T(0.4, 0.6, 0.75)
October	U(0.40, 0.70)	U(0.20, 0.50)	U(0.40, 0.70)	U(0.40, 0.70)
November	U(0, 0.3)	U(0, 0.1)	U(0, 0.3)	U(0, 0.3)
December	0	0	0	0

* Uniform (minimum, maximum)
† Triangular (minimum, mode, maximum)

During the main grazing period, it is assumed that 40% to 75% of the dry matter intake of beef cattle consists of fresh pasture grass, with a central estimate (mode) of 60% (triangular distribution).

These assumptions are in general agreement with the parameters used by NCI (1997) in their study of the I-131 doses to people in Idaho from nuclear weapons testing at the Nevada Test Site.

A.2.2.3 Transfer from Feed to Cow's Milk

This report distinguishes between the feed-to-milk transfer coefficient (F_m) for "backyard" cows and that for "commercial" cows in terms of milk production rate. A set of feed-to-milk transfer coefficients for dairy cows consisting of 77 measurements from 19 lactating dairy cows was obtained from Miller (1996). The data indicated that cows producing more than 10 L d^{-1} had, on average, a lower F_m than cows producing less that 10 L d^{-1} (Apostoaei et al. 1999). This finding suggests that high-milk producing "commercial" cows exhibit, on average, a lower transfer to milk for the same intake of I-131 than low-milk producing "backyard" cows.

In addition to milk production rate, there are other factors that influence the feed-to-milk transfer, such as lactation period, effect of season, and according to some authors (NCI 1997), breed of dairy cows. These factors lead to an inter-cow variability of F_m. That is, measurements of F_m on different cows at the same time, or on the same cow at different times show variations of a factor of 3. When commercial milk is pooled from a population of animals, the effect of uncertainty due to inter-cow variability of F_m is reduced considerably.[3]

[3] The effect of milk pooling is similar to the effect determining the average F_m from a population of F_m samples.

A review of the available measurements of F_m for dairy cows, as well as a review of the F_m values used in other dose reconstruction studies, are presented in Table A.10 and Figure A.2. These reviews include the results of the recent NRC/CEC expert elicitation (Brown et al. 1997). More importantly, these reviews include F_m measurements collected during the Controlled Environmental Radioiodine Tests (CERT) performed during 1960s on the INEL site (Bunch 1966).

Table A.10 Summary of literature documenting the variability of the feed-to-milk transfer coefficient (F_m) for iodine-131 in dairy cows

	Central value	GSD*	Lower bound	Upper bound	Comments
Köhler et al. (1991)			1.0×10^{-3}	7.3×10^{-3}	Chernobyl fallout (range of 5 values)
Snyder et al. (1994)	9.2×10^{-3}	2.1	1.6×10^{-3}	5.2×10^{-2}	Hanford site study - Backyard cows (lognormal distribution)
Snyder et al. (1994)	1.2×10^{-2}	(0.002)†	7.3×10^{-3}	1.6×10^{-2}	Hanford site study - Commercial cows (normal distribution)
NCI (1997)	4.4×10^{-3}	2.1	4.0×10^{-4}	2.1×10^{-2}	Lognormal distribution
Whicker and Kirchner (1987)	8.4×10^{-3}				
Brown et al. (1997)	7.1×10^{-3}	2.4	5.3×10^{-4}	3.7×10^{-2}	Aggregated results of 10 experts' opinions
Miller (1996)	9.0×10^{-3}	1.9	4.0×10^{-3}	3.5×10^{-2}	Dairy cows with milk production less than 10 L d^{-1} (1960s Tennessee data)
Miller (1996)	6.0×10^{-3}	1.5	2.0×10^{-3}	1.5×10^{-2}	Dairy cows with milk production greater than 10 L d^{-1} (1960s Tennessee data)
Bunch (1966)	8.4×10^{-3}	1.96	2.2×10^{-3}	3.2×10^{-2}	CERT tests 1,2 and 7 at INEL
This study‡	9.0×10^{-3}	1.8	2.6×10^{-3}	3.2×10^{-2}	Backyard cows - lognormal distribution (routine releases)
This study‡	6.0×10^{-3}	1.4	2.7×10^{-3}	1.3×10^{-2}	Commercial cows - lognormal distribution (routine releases)

* Geometric standard deviation
† Standard deviation of the normal distribution
‡ The values used by Apostoaei et al. (1999) were adopted for this study

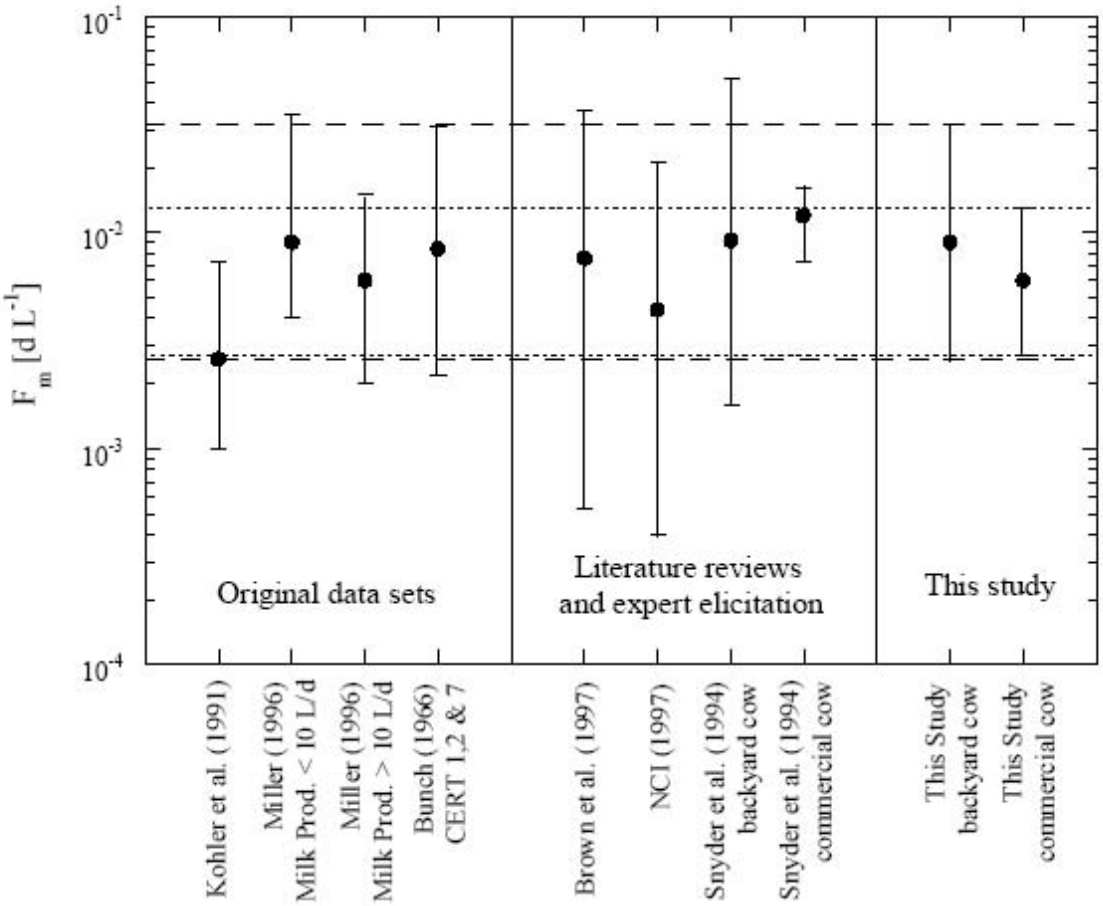

Figure A.2 Comparison of various studies that document variability of the iodine-131 feed-to-milk transfer coefficient

The heavy and light dashed lines indicate the 95% subjective confidence intervals on the distributions selected in this study for backyard cows and commercial cows, respectively.

Based on these reviews, the selected probability distribution for the feed-to-milk transfer coefficient in backyard cows is lognormal with a geometric mean of 9.0×10^{-3} d L^{-1} and a geometric standard deviation of 1.8.

For commercial cows, the selected subjective probability distribution for the feed-to-milk transfer coefficient in commercial cows is lognormal with a geometric mean of 6.0×10^{-3} d L^{-1} and a geometric standard deviation of 1.4. As compared to the probability distribution for backyard cows, the distribution for commercial cows has a lower geometric mean to account for the higher milk yield of commercial cows, and a lower geometric standard deviation to account for the effect of reduced variability due to milk pooling.

The probability distribution for backyard cows is not different from the F_m values determined during the CERT studies performed on the INEL site (Figure A.2).

A.2.2.4 Transfer from Feed to Goat's Milk

The fraction of the total amount of ingested radioiodine that is transferred to the entire release of milk produced by goats is larger than the fraction transferred to cow's milk by a factor of about 5 (NCI 1997), because the mammary gland of goats is a very efficient trap for iodine. In addition, the milk yield of a goat is about 10 times less than that of a dairy cow. After a daily intake of the same activity, radioiodine in the milk of goats can be up to 50 times more concentrated than in the milk produced by dairy cows. Individuals drinking contaminated goat's milk receive a much higher dose than do those consuming similar amounts of contaminated cow's milk, even accounting for the fact that goats ingest much less vegetation on a daily basis than do dairy cows. On the other hand, the number of people consuming goat's milk on a regular basis is far less than the number of people who drink cow's milk.

Hoffman (1978) summarized experimental values of the feed-to-goat's milk transfer coefficient from 10 different studies. The reported values range from 0.06 d L^{-1} to 0.65 d L^{-1} with a geometric mean of 0.34 d L^{-1}. Snyder et al. (1994) reviewed various experimental studies and assigned a lognormal distribution with an average of 0.27 d L^{-1} and a range of 0.04 to 1.15 d L^{-1}. The research performed by the National Cancer Institute (NCI 1997) added 11 more studies to those summarized by Hoffman (1978), and generated a lognormal distribution for the goat's milk transfer coefficient with a geometric mean of 0.22 d L^{-1} and a geometric standard deviation of 2.5, which produces a 95% confidence interval of 0.037 d L^{-1} to 1.33 d L^{-1}.

By using a lognormal distribution as indicated by NCI (1997), the upper limit for the transfer coefficient exceeds 1.0 d L^{-1}, which is an unrealistic value for goats producing more than 1 liter of milk per day. Thus in this study, the subjective probability distribution for the feed-to-milk transfer coefficients for goats was selected to be log-triangular (instead of a lognormal) distribution. The mode of the distribution was set to 0.22 d L^{-1}. The minimum and maximum values of this distribution are 0.03 and 0.8 d L^{-1}, respectively.

A.2.2.5 Transfer from Feed to Beef

Intake-to-beef transfer coefficients give activity concentrations of radionuclides in meat (Bq kg^{-1}) per activity intake per day by beef cattle (Bq d^{-1}) and, thus, are given in units of d kg^{-1}.

On the basis of data reviewed by Ng (1982) and other early measurements, Apostoaei et al. (1999) assumed that the intake-to-beef transfer coefficient for iodine can be described by a log-triangular probability distribution with a minimum at 5×10^{-4} d kg^{-1}, mode at 3×10^{-3} d kg^{-1}, and maximum at 2×10^{-2} d kg^{-1}. In a second review paper by Ng et al. (1982), the transfer coefficient for beef was reported to be 3.6×10^{-3} d kg^{-1}.

A review by the IAEA (1994) gave the recommended range of beef transfer coefficients as 7×10^{-3} to 5×10^{-2} d kg^{-1}. The transfer coefficients in beef recommended by the IAEA (1994) are based on more recent measurements than those recommended previously by Ng et al. (1982).

On the basis of data adopted by the IAEA (1994), Ng et al. (1982), and Apostoaei et al. (1999), we assume that intake-to-beef coefficients for iodine can be described by lognormal probability

distributions with a geometric mean of 3.2×10^{-3} d kg^{-1} and a geometric standard deviation of 2.6. The corresponding 95% subjective confidence interval ranges from 5×10^{-4} to 2×10^{-2} d kg^{-1}.

A.2.2.6 Transfer from Feed to Eggs

Intake-to-eggs transfer coefficients give activity concentrations of radionuclides in egg contents (Bq kg^{-1}) per activity intake per day by chickens (Bq d^{-1}) and, thus, are given in units of d kg^{-1}.

On the basis of measurements of fallout I-131 in cow's milk and egg contents summarized by the National Cancer Institute (NCI 1997), Apostoaei et al. (1999) assumed that the average concentration in eggs relative to the average concentration in milk for the same amount of iodine deposited on pasture vegetation could be described by a lognormal probability distribution with a median at 1.0 and a geometric standard deviation of 1.4.

Taking into account the assumed probability distribution of the intake-to-milk transfer coefficient in commercial dairy cows (Section A.2.2.3) and assumptions that chickens in the studies summarized by the NCI (1997) consumed about 0.005 kg d^{-1} of contaminated grass, the estimated intake-to-egg transfer coefficient is in the range of 5 to 30 d kg^{-1}. This range does not take into account uncertainties in the assumed intake rates of contaminated grass. When the uncertainty in the grass intake rates is taken into account, the intake-to-egg content transfer coefficient can be as high as 60 d kg^{-1}.

In a review by Ng et al. (1982), the transfer coefficients for egg contents ranged from 1.8 to 3.8 d kg^{-1}. This range differs somewhat from values given previously by Ng (1982).

Finally, the review by the IAEA (1994) recommended a range of 2 to 4 d kg^{-1} for the transfer coefficient for egg contents. The transfer coefficients in egg contents recommended by the IAEA (1994) are the same as those recommended previously by Ng et al. (1982).

On the basis of data adopted by the IAEA (1994), Ng et al. (1982), and Apostoaei et al. (1999), we assume that the intake-to-egg transfer coefficient for iodine can be described by a lognormal probability distribution with a geometric mean of 7.75 d $kg^{-1}$and a geometric standard deviation of 2.84. This distribution has a 95% subjective confidence interval of 1 to 60 d kg^{-1}.

A.2.2.7 Summary of Transfer Factors

A summary of the model parameters used to estimate the transfer of I-131 from pasture and soil to milk, beef, and eggs is presented in Table A.11.

Table A.11 Assumed probability distributions for parameters used to estimate the transfer of iodine-131 from pasture to milk, beef, and eggs

Parameter	Units	Distribution			
		Minimum	Maximum	Central	Shape
Backyard cow					
Q_m	[kg $_{dry}$ d^{-1}]	7	14	9*	triangular
p_m	[%]	Refer to Table A.9			
F_m	[d L^{-1}]			0.009† (1.8)‡	lognormal
Commercial cow					
Q_m	[kg $_{dry}$ d^{-1}]	10	18	12	triangular
p_m	[%]	Refer to Table A.9			
F_m	[d L^{-1}]			0.006† (1.4)‡	lognormal
Goat					
Q_m	[kg $_{dry}$ d^{-1}]	0.7	4.0	2.0	triangular
$p_{m,}$	[%]	Refer to Table A.9			
F_m	[d L^{-1}]	0.03	0.8	0.22	log-triangular
Beef cattle					
Q_f	[kg $_{dry}$ d^{-1}]	7	14	9	triangular
p_f	[%]	Refer to Table A.9			
F_f	[d kg^{-1}]			0.0032† (2.56)‡	lognormal
Egg Content					
Q_f	[kg $_{dry}$ d^{-1}]	0	0.01		uniform
p_e	[%]			100%	constant
F_e^C	[d kg^{-1}]			7.75† (2.84)‡	lognormal

* Mode
† Geometric mean
‡ Geometric standard deviation

A.2.3 Modeling Transfer of I-131 to Mother's Milk

Lactating mothers consuming I-131-contaminated food will transfer I-131 to their milk, resulting in I-131 exposure of their breast-fed infants. The accumulation of radioiodine in mother's milk is estimated using a diet-to-milk transfer coefficient (F_{mm}). The model used to estimate the intake of iodine by an infant is described in Section A.3.

Simon et al. (2002) reviewed and analyzed relevant data on the transfer of radioiodine into human milk. Estimates of milk transfer coefficients for the normal-excretion group were described by a lognormal distribution with a geometric mean of 0.37 d L^{-1} and a geometric standard deviation of 1.5.

The geometric mean value of 0.37 d L^{-1} is larger than the feed-to-milk transfer coefficient for backyard cows (0.009 d L^{-1}) or commercial cows (0.006 d L^{-1}). However, for the same ground deposition, the intake of I-131 by women is much lower than the amount ingested by grazing animals. Consequently, the concentration of I-131 in mother's milk is significantly lower than the concentration in milk of grazing animals.

The subjective probability distribution for the diet-to-mother's milk transfer coefficient adopted for this assessment is based on the geometric mean (0.37 d L^{-1}) and geometric standard deviation (1.5) reported above by Simon et al. (2002).

A.3 Human Intake from Food Consumption and Inhalation

Airborne radioiodine released from the Idaho Chemical Processing Plant (ICPP) was transferred from the air to vegetation, which was then consumed by animals and humans. Contaminated human foodstuffs included vegetables, meat and milk obtained from cattle or goats consuming contaminated pasture and grains, and eggs obtained from chickens consuming small amounts of contaminated vegetation while roaming freely. Breast milk consumption is also considered for infants (0-18 months old), due to the fact that mothers may have consumed foods contaminated with radioiodine. This section describes the assumptions made for food consumption patterns of Idaho residents of both genders and all age groups. The model for inhalation of I-131 in air is also described.

The subjective probability distributions provided for the consumption and inhalation rates represent our current state of knowledge about these parameters.

A.3.1 Ingestion of Contaminated Milk

Contaminated milk could have been obtained from a backyard cow, a local commercial dairy, a regional commercial dairy, or a goat. For infants, consumption of mother's milk containing I-131 is also included as an exposure pathway. The intake described in Equation A.20 applies to milk collected from backyard cows, commercial dairies, and goats. The intake described in Equation A.21 applies to the consumption of mother's milk by infants.

$$INT_{milk} = TIC_m \cdot \exp\left(-\lambda_R \cdot T_{d,m}\right) \cdot U_m \cdot F_{cm} \qquad \text{(A.20)}$$

where

INT_{milk} = the intake of I-131 due to the ingestion of contaminated milk [Bq]

TIC_m = the time-integrated concentration of iodine in milk [Bq d L^{-1}_{milk}] at time of milking (see Equation A.17)

λ_R = radioactive decay constant [d^{-1}]

$T_{d,m}$ = delay time between milking and consumption [d]

U_m = ingestion rate of milk from all sources [L d^{-1}_{milk}]

F_{cm} = fraction of milk consumed by an individual that is obtained from contaminated sources [unitless]

$$INT_{mm} = TIC_m \cdot \exp\left(-\lambda_R \cdot T_{d,m}\right) \cdot U_m \cdot F_{cm} \cdot P_{mm} \cdot F_{mm} \quad \text{(A.21)}$$

where

INT_{mm} = the intake of I-131 by an infant due to the ingestion of mother's milk [Bq]

TIC_m = the time-integrated concentration of iodine in milk [Bq d L^{-1}_{milk}] at time of milking (i.e., milk consumed by mother) (see Equation A.17)

λ_R = radioactive decay constant [d^{-1}]

$T_{d,m}$ = delay time between milking and consumption by mother [d]

U_m = ingestion rate of milk from all sources by mother [L d^{-1}]

F_{cm} = fraction of milk consumed by a mother that is obtained from contaminated sources [unitless]

P_{mm} = milk production rate of mother [L d^{-1}]

F_{mm} = transfer coefficient for mother's milk [d L^{-1}] (see Section A.2.3)

The intake of I-131 from mother's milk is calculated for each month of life from birth to one and a half years of age (i.e., 18 month). The model has the option of estimating thyroid doses by including either 3, 6, 12, or 18 months of breastfeeding.

Input Parameters

Delay Time from Collection to Consumption

For people drinking milk from backyard cows or goats, a minimum delay time between milking and human consumption is usually about 8 hours (0.33 days). This is the time required for the fresh milk to cool down (Simon et al. 1990). The upper limit of the holdup time was chosen to be about 2 days. For the delay between milking and consumption for backyard cow's milk or goat's milk, a uniform distribution between 0.33 and 2 days was assumed.

For milk from commercial dairies, at least 1 day is necessary for the transportation of milk from the producer to the consumer. Commercial milk was kept in grocery stores for at most 3 to 4 days, and then consumed within 1 or 2 days from the day of purchase. As a result, some individuals might have consumed milk up to 6 days after milking. A triangular distribution with a minimum of 1 day, a mode of 3 days and a maximum of 6 days was assumed for the delay time between milking and consumption for commercial milk.

Milk Ingestion Rate

Children are the critical group for this pathway due to their smaller thyroid mass and greater sensitivity to radiation exposures as compared to adults. For infants, the model assumes that cow's milk or breast milk may be provided during the first year of life. The consumption rates of cow's milk or mother's milk are assumed to be identical. However, iodine transfer rates from feed-to-milk for cows and from diet-to-breast milk for mothers are different (see Section A.2.2.3 and A.2.3). Breast milk is considered only for infants (0-18 months old), since breast milk consumption is considered to decrease on average after the first 18 months of life.
Doses to people who were infants during 1957-1959 were estimated also by assuming a goat's milk diet. These doses would represent an upper bound, because the feed-to-milk transfer for goats is very high (Section A.2.2.4), and thus consumption of goat's milk produces doses larger than those from consumption of cow's milk or mother's milk. Even though consumption of goat's milk by infants is possible, it is, however, highly improbable.

Although there are many studies available in the literature regarding milk consumption patterns for humans, doses estimated in this study are based on general milk consumption categories (Table A.12). A person exposed to the 1957-1959 atmospheric releases of I-131 can determine his or her possible dose, by identifying the milk consumption category that is most representative for his or her lifestyle.

Table A.12 Milk consumption categories and associated consumption rates (in 8-oz. glasses per day) used in this study

Milk Consumption Category	Assumed Consumption Rate (8 oz. glasses per day)		
	Minimum	Maximum	Distribution
No milk	0	0	Constant
Low	0	2	Uniform
Average	1	3	Uniform
High	2	4	Uniform
Very high	3	9	Uniform

Fraction of Milk That Is Contaminated

It has been assumed for this study that 100% of the milk consumed comes from a contaminated source.

Milk Production Rate of the Mother

The milk production rate of mothers is assumed to be equal to the consumption rate of infants. During the first few months of life, infants are fed every 3 to 4 hours, with each serving consisting of 4-6 oz. (0.12-0.18 L) of milk (Eisenberg et al. 1994). As the infant develops, the number of feedings decreases, but the amount consumed increases (Eisenberg et al. 1994). An infant being fed every 3 hours would consume no more than 32 oz. (0.98 L). A uniform

distribution was chosen, with a minimum of 0.12 L d^{-1} and a maximum of 0.98 L d^{-1}, to represent the milk production rate by mothers.

A.3.2 Ingestion of Contaminated Beef

Radioiodine is deposited on pasture and grain crops; therefore, chickens, cattle, and swine consuming these products transfer radioactivity to their meat, which is then consumed by humans. In this study, beef is considered the surrogate for all meat, including poultry and swine. Equation A.22 describes the approach used in this study to estimate the intake of contaminated meat:

$$INT_{meat} = TIC_f \cdot \exp\left(-\lambda_R \cdot T_{d,f}\right) \cdot U_f \cdot F_{cf} \cdot F_{rf} \qquad \text{(A.22)}$$

where

INT_{meat} = the intake of I-131 due to the ingestion of contaminated beef [Bq]

TIC_f = the time-integrated concentration of iodine in beef [Bq d kg^{-1}] at time of slaughter (see Equation A.18)

λ_R = radioactive decay constant [d^{-1}]

$T_{d,f}$ = delay time between slaughter and consumption [d]

U_f = ingestion rate of beef from all sources [kg d^{-1}]

F_{cf} = fraction of beef consumed by an individual that is obtained from contaminated sources [unitless]

F_{rf} = fraction of contamination remaining in meat after food preparation [unitless]

Input Parameters

Delay Time from Collection to Consumption

Part of the meat obtained when an animal was slaughtered was consumed soon after the sacrifice of the animal because refrigeration systems were not widely available during the 1950s. However, meat could have been treated with salt or smoked in a smokehouse, in which case it could have been stored for longer periods of time, during which I-131 decayed. For the meat not treated for long-time storage, the delay time between slaughter and consumption of meat was judged to be at least 1 week, but no longer than 5 weeks. This was assumed to be the case for all exposure scenarios included in this study (rural resident, urban resident, or migrant worker—see Section 4). A uniform distribution between 7 and 35 days was chosen for the delay time between slaughtering and consumption. This distribution is consistent with assumptions employed for the dose reconstruction at Hanford, Washington (Snyder et al. 1994).

Fraction of Beef That Is Contaminated

As argued above, only part of the meat obtained from an animal was consumed shortly after slaughtering, while the rest was stored either by refrigeration (not widely spread in late 1950s), by treatment with salt or by smoking in smokehouse. Also, meat could have been obtained from sources located far from the INEL site. This meat was considered to be uncontaminated. In this study, it was assumed that 30% – 90% of the meat was contaminated with I-131 from INEL. The upper bound accounts for people who have access on a regular basis to a local source of fresh meat, such as a local store or beef cattle ranch. The lower bound refers to people who stored meat from their own animals (so they consumed a lower amount of fresh meat), or to people who consumed meat from a store that obtained meat from sources located far from the INEL site. A uniform probability distribution between 0.3 and 0.9 was used to describe the uncertainty in this parameter. This distribution was applied to all exposure scenarios included in this study (rural resident, urban resident, or migrant worker—see Section 4).

Beef Ingestion Rate

Infants consume insignificant amounts of meat compared to intake of milk. As children age, their consumption rate of meat increases. Ranges are provided for 1–8 year olds (both genders combined), 9–14 year olds (males and females individually), and adults (males and females individually). The consumption rates of beef from all sources are provided in Table A.13.

The ranges provided for children are consistent with the average daily consumption rate of beef, pork, and poultry reported by the USDA (1965 and 1980) and the "best estimate" reported by Rupp (1980). For children 1 year old and younger, large quantities of milk are being consumed, so their intake of meat, if any, will be minimal (Apostoaei et al. 1999). For individuals aged 15 to 75 plus, the ranges are consistent with consumption rates of meat reported by ATSDR (1992), Cochrane (1945), Rupp (1980), and USDA (1944, 1949, 1965, 1980).

Fraction of Contamination Remaining after Preparation

IAEA (1994) summarizes the fractional losses of contamination due to food preparation for different types of foods. For meat, fractional losses are listed for various types of food preparation (i.e., boiling, frying, marinating, mincing, or sausage production). In this study, the fraction of I-131 remaining after cooking meat was considered to vary from 0.2 to 0.9. A uniform probability distribution with the above limits was used to describe the uncertainty in this parameter.

Table A.13 Beef consumption rates from all sources (kg d^{-1}) used in this study

Age Category	Minimum kg d^{-1} (oz d^{-1})[a]	Maximum kg d^{-1} (oz d^{-1})	Shape
Ages 0-1	0.005 (0.2)	0.015 (0.5)	Uniform
Ages 1-8	0.05 (2)	0.11 (4)	Uniform
Ages 9-14, male	0.08 (3)	0.14 (5)	Uniform
Ages 9-14, female	0.07 (2.5)	0.13 (4.5)	Uniform
Adult males	0.15 (5)	0.32 (12)	Uniform
Adult females	0.10 (4)	0.20 (8)	Uniform

* The values given in parentheses represent the number of ounces of beef consumed per day.

A.3.3 Ingestion of Contaminated Leafy Vegetables

Contamination of "leafy" vegetables has the possibility of being high due to the large surface area of the leaf exposed to the contaminated ground-level air. However, contamination is substantially reduced by washing. In addition, fresh vegetables are seasonal food products and are a source of exposure only during the harvest period. Equation A.23 describes the approach used to estimate the intake of contaminated fresh leafy vegetables:

$$INT_{veg} = TIC_{veg} \cdot \exp\left(-\lambda_R \cdot T_{d,veg}\right) \cdot U_{veg} \cdot F_{cv} \cdot F_{wv} \qquad \text{(A.23)}$$

where

INT_{veg} = the intake of I-131 due to the ingestion of contaminated fresh leafy vegetables [Bq]

TIC_{veg} = the time-integrated concentration of iodine in fresh leafy vegetables [Bq d $kg^{-1}_{\text{fresh mass}}$] at time of harvest

$$TIC_{veg} = C_v \cdot ED$$

C_v = the average concentration on fresh leafy vegetables per month [Bq $kg^{-1}_{\text{fresh mass}}$] (see Equation A.2),

ED = exposure duration; the number of days per month [d $month^{-1}$]

λ_R = radioactive decay constant [d^{-1}]

$T_{d,veg}$ = delay time between harvest and consumption [d]

U_{veg} = ingestion rate of fresh leafy vegetables from all sources [kg $_{\text{fresh mass}}$ d^{-1}]

F_{cv} = fraction of vegetables consumed by an individual that is obtained from contaminated sources [unitless]

F_{wv} = fraction of contamination remaining on the plant after washing [unitless]

Input Parameters

Delay Time from Collection to Consumption

Leafy vegetables are assumed to be consumed fresh during the harvest months of the year, and the storage time for "fresh" vegetables is assumed not to exceed 1 week. The distribution of the delay time between harvesting and consumption of leafy vegetables is assumed to be uniform; between 0 and 7 days.

Vegetable Ingestion Rate

Leafy vegetables are considered to include lettuce, broccoli, cabbage, celery, and spinach, which are the most frequently consumed leafy vegetables (NCI 1997). Other vegetables, such as carrots, beans, and corn, are either not directly exposed to I-131 in air, or they are usually stored for a long time before consumption, allowing for I-131 to decay. The ingestion rates used in this study refer only to consumption of leafy vegetables, and thus they are only a fraction of the total vegetable intake for a normal person.

Consumption rates of leafy vegetables are provided for the following groups: infants (6 months to 1 year) and children (1-4 years) as one group; children ages 5-14 years (both genders); and adults (both genders). The average daily leafy vegetables consumption rates did not vary between males and females. Table A.14 provides the probability distributions used in this study to describe the fresh leafy vegetable consumption. The ranges listed are consistent with values reported by the EPA (1997), USDA (1965;1980) and NCI (1997).

Table A.14 Fresh leafy vegetables ingestion rates from all sources used in this study*

Age Category	Minimum kg d^{-1} (oz d^{-1})†	Maximum kg d^{-1} (oz d^{-1})	Shape
Ages 6 months to 4 years (males and females)	0.004 (0.15)	0.014 (0.5)	Uniform
Ages 5-14 years (males and females)	0.01 (0.35)	0.04 (1.5)	Uniform
Adults (males and females)	0.02 (0.7)	0.06 (2)	Uniform

* The values represent the consumption rates of leafy vegetables only (i.e., lettuce, spinach, cabbage). The total consumption rates of vegetables (i.e., all types of vegetables) are larger by a factor up to 10 than the values included in the table (EPA 1997).

† The values given in parentheses represent the number of ounces of vegetables consumed per day.

Fraction of Vegetable Consumption That Is from Contaminated Sources

Two distinct living conditions are considered with respect to the fraction of vegetables consumed that are contaminated: the rural resident and the urban resident. The rural resident is assumed to have grown and consumed home-grown vegetables. Leafy vegetables are not typically grown during the months of January, February, March, April, November, and December, due to snowfall and cold temperatures. During the months from May to October, a uniform probability distribution between 0.8 and 1.0 was chosen for the amount of leafy vegetables that are contaminated.

An urban resident is assumed to have had access to locally produced fresh leafy vegetables during summer months, but because he or she did not own a vegetable garden, a lower fraction of contaminated vegetables was assumed for May, September, and October. The assumed probability distributions for the fraction of vegetables consumed that are contaminated are listed in Table A.15. These distributions are consistent with those used in the dose reconstruction study for I-131 atmospheric releases from the RaLa processing in Oak Ridge, Tennessee (Apostoaei et al. 1999)

Table A.15 Assumed probability distributions used to represent fraction of vegetables consumed that are contaminated

Month	Rural Resident	Urban Resident
January	0	0
February	0	0
March	0	0
April	0	0
May	Uniform(0.8, 1)	Uniform(0.25, 0.5)
June	Uniform(0.8, 1)	Uniform(0.8, 1)
July	Uniform(0.8, 1)	Uniform(0.8, 1)
August	Uniform(0.8, 1)	Uniform(0.8, 1)
September	Uniform(0.8, 1)	Uniform(0.25, 0.5)
October	Uniform(0.8, 1)	Uniform(0, 0.5)
November	0	0
December	0	0

Fraction of Contamination Remaining after Washing

A review of the literature indicates that the amount of contamination remaining on the plant after washing is larger than 20%, but lower than 70% (Thiessen et al. 1996, IAEA 1992, IAEA 1994). A uniform distribution between 0.2 and 0.7 was considered for estimating the uncertainty in the fraction of contamination remaining on the plants.

A.3.4 Ingestion of Contaminated Eggs

Iodine-131 can accumulate in eggs if chickens are fed contaminated feed. In general, chicken feed is stored for periods of time longer than the half-life of I-131. However, if chickens are allowed to roam freely, they may consume small amounts of contaminated grass or soil. In this case, small amounts of I-131 are transferred to eggs.

Equation A.24 describes the approach used to estimate the intake of contaminated fresh leafy vegetables:

$$INT_{eggs} = TIC_e \cdot \exp\left(-\lambda_R \cdot T_{d,e}\right) \cdot U_e \cdot F_{ce} \cdot F_{re} \qquad \text{(A.24)}$$

where

INT_{eggs} = the intake of I-131 due to the ingestion of contaminated eggs [Bq]

TIC_e = the time-integrated concentration of iodine in eggs [Bq d kg^{-1}] at time of collection

λ_R = radioactive decay constant [d^{-1}]

$T_{d,e}$ = delay time between collection and consumption [d]

U_e = ingestion rate of eggs from all sources [kg d^{-1}]

F_{ce} = fraction of eggs consumed by an individual that is obtained from contaminated sources [unitless]

F_{re} = fraction of contamination that remains after preparation of eggs [unitless]

Input Parameters

Delay Time from Collection to Consumption

It has been assumed that eggs collected from a local farm are consumed within 8 hours to 2 days after collection. A uniform probability distribution, with a minimum value of 0.33 days and a maximum value of 2 days, has been chosen to represent the delay time for local eggs.

However, eggs available from commercial sources are not typically consumed as quickly. Including collection times, delivery times, and shelf life, commercial eggs are typically consumed between 3 days and 2 weeks. A triangular probability distribution, with a minimum value of 3 days, a maximum value of 14 days, and a mode of 7 days, has been chosen to represent the delay time for commercial eggs.

Egg Ingestion Rate

The ranges of the probability distributions describing the egg consumption rates (Table A.16) are consistent with those reported by the USDA for all urban areas (USDA 1980) and for rural farm residents (USDA 1965). The central values are similar to the median values for egg

consumption for children in the various age groups reported by the National Cancer Institute (NCI 1997). The range for infants was considered to include children (males and females) between the ages of 6 months and 1 year. Children less than 6 months were not included due to their limited intake [0 g d^{-1} median value reported for the intake rate of eggs for 0–2 months and 0.005 g d^{-1} median value reported for 3–5 months (NCI 1997)]. According to the USDA (1980), eggs were eaten by only one-tenth of the infants surveyed in the spring of 1977. Children between the ages of 1 and 8 years are considered without distinction by gender. At the age of 9, differences between males and females become apparent in rates of consumption of various food types. The egg consumption rates considered in this study include the eggs used for cooking of different meals (e.g., eggs used to bake a cake, etc.).

Fraction of Eggs That Is Contaminated

It has been assumed for this study that all eggs consumed come from a source located around the INEL site, and therefore are contaminated.

Fraction of Contamination Remaining after Preparation

The I-131 concentration in eggs is reduced during the process of cooking, frying, or boiling. Although no values for losses from cooking and boiling of eggs have been found in the literature, the IAEA (1992) reports that from 20% to 90% of I-131 remains with various food products after various methods of preparation. These data are used to specify a plausible range for I-131 losses from food preparation of eggs. It is assumed that anywhere from 20% to 90% of the contamination remains after preparation. Thus, a uniform distribution has been selected, with a minimum value of 0.2 and a maximum value of 0.9.

Table A.16 Values used in this study to characterize the egg consumption rates

Age Category	Minimum kg d^{-1} (eggs d^{-1})*	Maximum kg d^{-1} (eggs d^{-1})	Shape
Age 6 months to 1 year, males and females	0.01 (0.25)	0.04 (1)	Uniform
Ages 1-8 years, males and females	0.02 (0.5)	0.08 (1.5)	Uniform
Ages 9-14 years, males	0.03 (0.5)	0.10 (2)	Uniform
Ages 9-14 years, females	0.02 (0.5)	0.08 (1.5)	Uniform
Adult males	0.03 (0.5)	0.10 (2)	Uniform
Adult females	0.02 (0.5)	0.10 (2)	Uniform

* The values given in parentheses represent the *approximate* number of eggs consumed per day.

A.3.5 Summary of Parameters Used to Model Intake from Consumption of Contaminated Food Products

Intake of I-131 by consumption of milk, meat, leafy vegetables or eggs is estimated using Equations A.20 and A.21. All equations start with the time-integrated concentration of iodine in milk, meat, leafy vegetables, and eggs, respectively, which are described in Section A.2. Table A.17 provides a summary of the parameters used to model intake from consumption of each food products.

Table A.17 Assumed probability distributions for parameters used to estimate the intake of various food products

Parameter	Units	Distribution			
		Min.	Max.	Mode	Shape
Milk from backyard cows					
T_{d*}	[d]	0.33	2		uniform
U_m †	[L d^{-1}]	*Refer to Table A.12*			
F_{cm} ‡	[unitless]			1	constant
Milk from commercial sources					
T_{d*}	[d]	1	6	3	triangular
U_m †	[L d^{-1}]	*Refer to Table A.12*			
F_{cm} ‡	[unitless]			1	constant
Milk from goats					
T_{d*}	[d]	0.33	2		uniform
U_m †	[L d^{-1}]	*Refer to Table A.12*			
F_{cm} ‡	[unitless]			1	constant
Meat					
$T_{d, f*}$	[d]	7	35		uniform
U_f †	[kg d^{-1}]	*Refer to Table A.13*			
F_{cf} ‡	[unitless]	0.3	0.9		uniform
F_{rf} §	[unitless]	0.2	0.9		uniform
Vegetables					
$T_{d, v*}$	[d]	0	7		uniform
U_v †	[kg_{fresh} d^{-1}]	*Refer to Table A.14*			
F_{cv} ‡	[unitless]	*Refer to Table A.15*			
F_w §	[unitless]	0.2	0.7		uniform
Eggs					
T_{gg*}	[d]	3	7		uniform
U_e †	[kg d^{-1}]	*Refer to Table A.16*			
F_{ce} ‡	[unitless]			1.0	constant
F_{ef} §	[unitless]	0.2	0.9		uniform

* Delay time between milking, slaughtering, harvesting or collection and consumption of the food product.

† Consumption rate for each food product.

‡ Fraction of the amount consumed by an individual that is obtained from contaminated sources.

§ Factor accounting for reduction of the contamination by cooking (for meat and eggs) or washing (for vegetables).

A.3.6 Inhalation of Contaminated Air

Although ingestion of cow's milk is the most important route for human exposure to I-131, the inhalation pathway affects every individual in the population. Inhalation can become an important route of exposure if other pathways are not relevant. Exposure to I-131 through the inhalation pathway depends on the concentration of I-131 in air, on the breathing rate of the specific individual, and on the ability of each physico-chemical form of iodine to deposit in the respiratory system.

The modeling approach chosen for the inhalation pathway is based on the following major assumptions:

- An individual spends a fraction of time (f_o) outdoors
- The indoor concentration of I-131 in air is lower than the outdoor concentration of I-131 by a specified factor (r_{io})
- The amount of I-131 inhaled is only partially deposited in the respiratory system; the fraction deposited (D_k; Section A.4.2) is different for each physico-chemical form k of iodine
- The I-131 deposited in the respiratory system is totally absorbed and rapidly transferred to the bloodstream, from where it is metabolized in a manner similar to that of the ingested iodine

The total intake of I-131 from inhalation of contaminated air is given by the following equation:

$$INT_{INH} = (f_o + (1 - f_o) \cdot r_{io}) \cdot C_a \cdot BR \cdot D_k \cdot ED \qquad \text{(A.25)}$$

where

INT_{INH} = intake of $^{I\text{-}131}$ from inhalation [Bq]

f_o = fraction of time spent outdoors [unitless]

$r_{i\,o}$ = ratio of the indoor to outdoor concentrations of iodine in air [unitless]

C_a = concentration of iodine in outside air [Bq m^{-3}_{air}]

BR = breathing rate for an individual [m^3_{air} d^{-1}]

D_k = fraction of the total amount inhaled that deposits and is absorbed in different parts of the respiratory system for each physico-chemical form k (Section A.4.2)

ED = exposure duration, number of days in month of interest [d $month^{-1}$]

Due to the use of parameter D_k, the intake (INT_{inh}) estimated by Equation A.25 represents the activity of inhaled I-131 that is ultimately absorbed in the blood stream. Another name for this quantity could be the "total systemic uptake" of I-131 due to inhalation. The reason for using this approach is that INT_{inh} is equivalent to the intake of I-131 from ingestion of food (i.e., INT_m, INT_f, INT_e, Equations A.17 through A.19) or soil (INT_{soil}; Equation A.26). The intakes from Equations A.17 through A.19 represent the activity of iodine ingested. Because practically 100% of the ingested iodine is absorbed into blood, INT_m, INT_f, INT_e, INT_{soil} also represent the activity of I-131 transferred into blood. Thus, INT_{inh} and INT_m, INT_f, INT_e, INT_{soil} are equivalent and they can be summed into a total intake (see Equation A.27). The doses from both ingestion and inhalation can be estimated by using the total intake and the dose coefficient for ingestion of I-131 (see Equation A.28). This approach accounts for the correlations between the doses due to ingestion and those due to inhalation.

Input Parameters

Fraction of the Day Spent Outdoors

The fraction of time spent outdoors has been defined for three general categories: limited, large, and very large. The computer model allows the user to choose any of these categories for a given exposure scenario (i.e., rural resident, urban resident or migrant worker—see Section 4). The fraction of time spent outdoors is defined as the number of hours per day assumed, and, in this assessment, the fraction is dependent on age and the time of the year (e.g., an adult would likely spend more time outdoors in the summertime than in the wintertime). Uniform probability distributions were used to represent this parameter. The minimum and maximum values are given in Tables A.18 through A.20.

Table A.18 Assumed number of hours per day spent outdoors for an individual that spent a limited amount of time outdoors

Month	Number of hours per day spent outdoors (minimum – maximum)*					
	Infant	Age 1	Age 5	Age 10	Age 15	Adult
January	0 - 1	0 - 1	0 - 2	0 - 2	0 - 2	0 - 3
February	0 - 1	0 - 1	0 - 2	0 - 2	0 - 2	0 - 3
March	1 - 2	1 - 2	1 - 3	1 - 3	1 - 3	1 - 4
April	1 - 2	1 - 2	1 - 3	1 - 3	1 - 3	1 - 4
May	1 - 2	1 - 2	1 - 3	1 - 3	1 - 3	1 - 4
June	2 - 3	2 - 3	2 - 4	2 - 4	2 - 4	2 - 6
July	2 - 3	2 - 3	2 - 4	2 - 4	2 - 4	2 - 6
August	2 - 3	2 - 3	2 - 4	2 - 4	2 - 4	2 - 6
September	1 - 2	1 - 2	1 - 3	1 - 3	1 - 3	1 - 4
October	1 - 2	1 - 2	1 - 3	1 - 3	1 - 3	1 - 4
November	1 - 2	1 - 2	1 - 3	1 - 3	1 - 3	1 - 4
December	0 - 1	0 - 1	0 - 2	0 - 2	0 - 2	0 - 3

* A uniform probability distribution, with the given minimum and maximum, was chosen to represent the number of hours spent outdoors.

Table A.19 Assumed number of hours per day spent outdoors for an individual that spent a large amount of time outdoors

Month	Number of hours per day spent outdoors (minimum – maximum)[a]					
	Infant	Age 1	Age 5	Age 10	Age 15	Adult
January	1 - 2	1 - 2	1 - 3	1 - 3	1 - 3	2 - 6
February	1 - 2	1 - 2	1 - 3	1 - 3	1 - 3	2 - 6
March	2 - 3	2 - 3	2 - 4	2 - 4	2 - 4	4 - 8
April	2 - 3	2 - 3	2 - 4	2 - 4	2 - 4	4 - 8
May	2 - 3	2 - 3	2 - 4	2 - 4	2 - 4	4 - 8
June	3 - 4	3 - 4	3 - 6	3 - 6	3 - 6	6 - 8
July	3 - 4	3 - 4	3 - 6	3 - 6	3 - 6	6 - 8
August	3 - 4	3 - 4	3 - 6	3 - 6	3 - 6	6 - 8
September	2 - 3	2 - 3	2 - 4	2 - 4	2 - 4	4 - 8
October	2 - 3	2 - 3	2 - 4	2 - 4	2 - 4	4 - 8
November	2 - 3	2 - 3	2 - 4	2 - 4	2 - 4	4 - 8
December	1 - 2	1 - 2	1 - 3	1 - 3	1 - 3	2 - 6

* A uniform probability distribution, with the given minimum and maximum, was chosen to represent the number of hours spent outdoors.

Table A.20 Assumed number of hours per day spent outdoors for an individual that spent a very large amount of time outdoors

Month	Number of hours spent outdoors (minimum – maximum)*					
	Infant	Age 1	Age 5	Age 10	Age 15	Adult
January	2 - 3	2 - 3	2 - 4	2 - 4	2 - 4	6 - 10
February	2 - 3	2 - 3	2 - 4	2 - 4	2 - 4	6 - 10
March	3 - 4	3 - 4	3 - 5	3 - 5	3 - 5	8 - 12
April	3 - 4	3 - 4	3 - 5	3 - 5	3 - 5	8 - 12
May	3 - 4	3 - 4	3 - 5	3 - 5	3 - 5	8 - 12
June	4 - 6	4 - 6	5 - 8	5 - 8	5 - 8	8 - 14
July	4 - 6	4 - 6	5 - 8	5 - 8	5 - 8	8 - 14
August	4 - 6	4 - 6	5 - 8	5 - 8	5 - 8	8 - 14
September	3 - 4	3 - 4	3 - 5	3 - 5	3 - 5	8 - 12
October	3 - 4	3 - 4	3 - 5	3 - 5	3 - 5	8 - 12
November	3 - 4	3 - 4	3 - 5	3 - 5	3 - 5	8 - 12
December	2 - 3	2 - 3	2 - 4	2 - 4	2 - 4	6 - 10

* A uniform probability distribution, with the given minimum and maximum, was chosen to represent the number of hours spent outdoors.

Indoor to Outdoor Concentration Ratio

The air inside a building is expected to have a different concentration of I-131 than the air outside the building, unless free air exchange occurs through open windows or doors. When windows and doors are closed, some air is still exchanged between indoors and outdoors, either naturally through openings due to imperfect sealing or by a ventilation system. For rural areas, the ventilation systems were not well developed during the 1950s. Air exchange by window or door opening was probably a common practice, especially during warm weather. For a given outdoor concentration of a contaminant, the indoor concentration of a contaminant is expected to have been larger during the 1950s than in present times.

The concentration of a contaminant in indoor air is a function of the rate at which the contaminant is entering the building from the outdoor air, the rate of indoor production of the contaminant (not an issue for I-131), and the rate at which the contaminant is leaving the building. In the first approximation, the outdoor air can be considered an infinite source of contaminant at an ever-changing concentration (i.e., the outdoor concentration is not changed by the air exchange with the air in the building, but only by the movement of the outside air). Because of the dynamic exchange process, the concentration inside a building may be higher than the outside concentration at a given moment of time. An explanation for such a situation is that the air inside reaches a peak concentration after the contaminated plume passes the building. However, on a time-averaged basis, the inside concentration is lower than the outside concentration.

Various studies have been performed to determine a relationship between outdoor and indoor air concentrations. In this report, a simple relationship, defined as a long-term average ratio between the indoor and outdoor concentrations, is used. Three literature reviews of experimental measurements have been used to support values selected for the indoor-to-outdoor concentration ratio: Andersen (1972), Benson et al. (1972), and Snyder et al. (1994).

Andersen (1972) summarized 11 studies investigating indoor/outdoor ratios for sulfur dioxide (SO_2) and for suspended particle matter. The investigations took place from 1954 to 1969 in various part of the world, including Cincinnati, Ohio, and Hartford, Connecticut. For gaseous SO_2, the indoor/outdoor ratio varied from 20% to 100%. For suspended matter, a low range of values (20%–60%) was observed in Tokyo, Japan, while for other locations, the ratio varied from 40% to 95%. Andersen (1972) also reports his own set of measurements performed in Denmark: 51% for SO_2 and 83% for suspended particulate matter.

Benson et al. (1972) compiled many indoor/outdoor ratios for gases [SO_2, carbon monoxide (CO), and other gaseous substances], for "viable" particles (spores, pollen, and bacteria), and for particulate matter. A number of the reported measurements of the indoor/outdoor ratio are larger than 100%. As argued before, these values may be valid in a single measuring event, but they do not apply to a long-term average of the indoor/outdoor ratio. For gases, the values below 100% ranged from 20% to 75% for SO_2, from 59% to 100% for CO, and from 34% to 80% for other gases. For particulate matter, a minimum value of 16% was observed in a 1971 measurement in

Hartford, Connecticut, during wintertime. However, most of the measurements indicated values larger than 40%, with a maximum of 100%.

Two more recent studies (Hawley 1985, Christensen and Mustonen 1987) are cited by Snyder et al. (1994) in a review performed for the reconstruction of I-131 doses for people living around the Hanford Nuclear Facility in the state of Washington. The reported measurements of the indoor/outdoor ratio for Norwegian houses built in 1954 range from 40% to 86% (Christensen and Mustonen 1987). In another set of measurements, a minimum value of 35% was reported by Hawley (1985).

Snyder et al. (1994) made no distinction between different species of iodine (gases versus particulate matter), and they assigned a uniform distribution from 35% to 100% for the indoor-to-outdoor ratio. This range of values appears appropriate, because it eliminates very low values, which are probably artifacts of special measurement conditions (such as a cold wintertime when the exchange of air between indoors and outdoors is deliberately limited). The maximum value (100%) takes into account the situation when there is free airflow between indoors and outdoors.

The present study assumes that the indoor-to-outdoor ratios for gases (i.e., elemental and organic iodine) are the same as those for particulate matter. The probability distribution for this parameter was chosen to be uniform, with a minimum value of 0.3 (30%) and a maximum value of 1 (100%).

Age-dependent Breathing Rate

Age-dependent breathing rates are reported by Roy and Courtay (1991). These values were also recommended by the NCI (1997) as applicable to the general population. The values have been linearly interpolated between different age groups. The values in Table A.21 are medians of lognormal distributions for each age group. A geometric standard deviation of 1.3, applicable for all age groups, was chosen based on the work of Roy and Courtay (1991) and on recommendations from the NCI (1997).

Table A.21 Age-dependent breathing rates [m^3 d^{-1}] for exposed individuals

Age Category	Distribution		
	GM*	GSD†	Shape
Age 0 - 1	3.5	1.3	lognormal
Age 1 - 4	7.0	1.3	lognormal
Age 5 - 9	12.0	1.3	lognormal
Age 10 - 14	17.0	1.3	lognormal
Age 15 - 19 (females)	18.0	1.3	lognormal
Age 15 - 19 (males)	19.0	1.3	lognormal
Adult (females)	18.0	1.3	lognormal
Adult (males)	23.0	1.3	lognormal

* GM = geometric mean
† GSD = geometric standard deviation

A.3.7 Inadvertent Soil Ingestion

Inadvertent ingestion is, by definition, accidental and can take place, for instance, in dusty environments where inhalation of dirt is likely to occur as a result of wind-driven resuspension. Particles of soil accumulated in the nasal portion of the respiratory tract can be subsequently swallowed, leading to ingestion of soil. Soil ingestion can also occur by consuming unwashed vegetables or fruits, or other food items on which airborne soil particles have deposited. Soil ingestion can be observed to some degree by watching children playing outdoors; or adults working in construction, agriculture, or food gathering; or during high wind resuspension incidence.

A thorough review of the phenomena of soil ingestion by humans was published by Simon (1998), who also provided the necessary parameters for an assessment of radiation doses from this exposure pathway (Table A.22).

The intake of $^{I-131}$ from ingestion of contaminated soil is given by the following equation:

$$INT_{soil} = \sum_{month} \left(C_s \cdot U_s \cdot OF \cdot ED\right) \cdot EF \qquad \text{(A.26)}$$

where:

INT_{soil}	=	annual intake of I-131 from inadvertent soil ingestion [Bq]
C_s	=	concentration of I-131 in the top layer of soil [Bq kg^{-1}] (Section A.1.4.5)
U_s	=	inadvertent soil ingestion rate [kg d^{-1}]
OF	=	occupational factor [unitless]
ED	=	exposure duration, number of days in month of interest [d $month^{-1}$]; and
EF	=	exposure frequency [unitless]

The occupational factor accounts for high/medium/low risk practices and depends on occupation, type of activity (plowing, earthmoving, etc), and hobbies of the studied individual.

Table A.22 Parameters used in this study for the soil ingestion pathway

Parameter	Probability distribution function describing the uncertainty in the listed parameter*			
Soil ingestion rate				
Rural resident				
Child	GM = 0.2 g d^{-1}	GSD = 4.2		Lognormal
Adult	GM = 0.2 g d^{-1}	GSD = 3.2		Lognormal
Urban resident				
Child	GM = 0.1 g d^{-1}	GSD = 4.2		Lognormal
Adult	GM = 0.1 g d^{-1}	GSD = 3.2		Lognormal
Occupational Factor				
Child	1.0			Constant
Adult	Min=0.5	Mode=1.0	Max=2.0	Triangular
Exposure Frequency				
All age groups	Min=180 d yr^{-1}	Mode = 270 d yr^{-1}	Max = 270 d yr^{-1}	Triangular

* Source: Simon 1998

A.3.8 Exposure to Multiple Sources of Contamination

Individuals living near the INEL facility may have been exposed by more than one pathway. In this case, the total intake of I-131 is obtained by summing the intakes for individual pathways, as shown in Equation A.27.

$$INT_{Total} = INT_{milk} + INT_{beef} + INT_{veg} + INT_{eggs} + INT_{inh} + INT_{soil} \quad \text{(A.27)}$$

where

INT_{Total} = total annual intake during 1957, 1958 or 1959 [Bq]

INT_{milk} = annual intake from milk consumption [Bq] (see Equation A.20 or A.21)

INT_{beef} = annual intake from beef consumption [Bq] (see Equation A.22)

INT_{veg} = annual intake from fresh leafy vegetable consumption [Bq] (see Equation A.23)

INT_{eggs} = annual intake from the consumption of eggs [Bq] (see Equation A.24)

INT_{inh} = annual intake from inhalation [Bq] (see Equation A.25)

INT_{soil} = annual intake from inhalation [Bq] (see Equation A.26)

Equation A.27 describes in general terms the summation of intakes from various exposure pathways. The intake for an exposure pathway is calculated for each month during 1957-1959, according to the age of the individual. For a given exposure pathway, the intake over all months in a given year are first summed to obtain an annual intake from that exposure pathway. This operation is repeated for each year of release, so an intake from a given exposure pathway is

estimated for 1957, 1958 and 1959, respectively. Equation A.27 sums the annual intakes for each exposure pathway and produces *total* annual intakes for 1957, 1958 and 1959. Each *total* annual intake represents the intake of I-131 for an individual who is of a given age in that year. The correlation between age at the time of exposure and year of release is important because doses per unit intake (i.e., dose coefficients) depend strongly on age at exposure.

If the individual was an infant during 1957-1959, the intake due to consumption of milk (INT_{milk}) can be set to consumption of mother's milk, instead of cow's or goat's milk. The breastfeeding period can be varied from 3 to 18 months. Section 7 presents results for different combinations of milk diets for infants.

A.4 Dose Estimation

A.4.1 Description and Modeling Approach

Thyroid doses from exposure to I-131 are calculated using the total intake (Section A.3.8) and the dose coefficients discussed below:

$$D = \sum_{i=N_1}^{N_2} INT_{Total,i} \cdot DCF_i \qquad \text{(A.28)}$$

where

D	=	equivalent dose to the thyroid due to intake of I-131 [Sv]
$INT_{Total,i}$	=	age (i)-dependent total intake of I-131 [Bq] (see Equation A.27)
DCF_i	=	thyroid dose per unit intake at age i (dose coefficient) [Sv Bq^{-1}]
N_1	=	age at which exposure began
N_2	=	age at which exposure ended
i	=	age of the individual (i=0,1,2,…) in year of release j

The effective dose from an intake of I-131 is due almost entirely to the dose to the thyroid, and contributions from irradiation of other organs or tissues are negligible (ICRP 1993). Effective doses are estimated by multiplying the estimated thyroid doses by a tissue-weighting factor (w_T) of 0.05 (ICRP 1991).

A.4.2 Dose Coefficients

Dose Coefficients for Ingestion of I-131

A set of doses per unit intake (i.e., dose coefficients) and associated uncertainties were derived by Apostoaei et al. (1999 and 2004) for ingestion of I-131, based on the most recent summary of measurements in thyroid mass obtained by ultrasonography, a method that is less prone to errors than autopsy and that allows in-vivo examinations of large populations. The ultrasonography also indicated thyroid masses slightly lower than those derived by autopsy. The dose

coefficients were described by lognormal distributions with the medians and the geometric standard deviations shown in Table A.23.

Dose Coefficients for Inhalation of I-131

Dose coefficients for inhalation of iodine are reported in ICRP publications, but uncertainties in these coefficients are not well quantified. In this study, we made use of the fact that ingested iodine is rapidly and almost totally transferred from the gastrointestinal tract into blood. Thus, the ingestion dose coefficients are representative for the dose per unit activity introduced into blood. Given that after inhalation iodine is also entirely transferred into blood, the doses from inhalation can be estimated using the ingestion dose coefficients (Table A.23) multiplied by the activity inhaled, and by the fraction of I-131 deposited in the respiratory system and absorbed into blood.

In the case of releases from ICPP, iodine was released almost entirely in elemental form. During atmospheric transport, some of the elemental iodine attaches to particles already existing in the atmosphere and some transforms into organic iodine. By the time the plume arrives at the downwind location where iodine is inhaled, the fraction of iodine in organic and particulate form will be appreciable. Most particles will be small or very small in size (< 1 μm) and iodine attached to particles is more likely type *F*, rather than type *M* or type *S* (using the most recent ICRP 1996 absorption classes).

We performed a comparison between the thyroid dose coefficients for ingestion presented in ICRP Publication 67 (1993) and the thyroid dose coefficients for inhalation derived from ICRP Publication 72 (1996), which are based on the new ICRP lung model and contain the effect of partial deposition and absorption of I-131 in the respiratory tract. ICRP Publication 67 (1993) reports ingestion dose coefficients based on 100% absorption of iodine.

The comparison was performed by taking the ratios between the dose coefficients based on the new lung model and the ingestion dose coefficients (no deposition or absorption in the respiratory tract). Once iodine reaches the blood, the metabolism and dosimetry is the same in the two ICRP Publications. Thus, the estimated ratios are an indicator of the overall effect of the deposition and absorption of I-131 as incorporated in the new lung model. For elemental iodine, the ratio was 0.9, and for organic iodine, the ratio was 0.7. For the fast-absorbing particles (f_1 = 1), the ratio was 0.4, and for the medium-absorbing particles (f_1 = 0.1), the ratio was 0.1.

Based on this comparison and on information about iodine deposition summarized by Apostoaei et al (1999), the following distributions were used for the fraction of iodine deposited and absorbed in the respiratory tract (D_k):

- For elemental iodine - a uniform distribution between 0.8 and 1.0 (central value 0.9)
- For particulate iodine - a triangular distribution between 0.1 and 0.8 with a mode of 0.4
- For organic iodine - a uniform distribution between 0.6 and 0.8 (central value 0.7)

Table A.23 Geometric mean* of the age-specific thyroid dose coefficients† (Sv Bq^{-1}) derived for use in this study

Age	Males	Females
0	3.88×10^{-6}	3.88×10^{-6}
1	3.57×10^{-6}	3.57×10^{-6}
2	3.24×10^{-6}	3.24×10^{-6}
3	2.92×10^{-6}	2.92×10^{-6}
4	2.59×10^{-6}	2.59×10^{-6}
5	2.26×10^{-6}	2.26×10^{-6}
6	2.05×10^{-6}	2.05×10^{-6}
7	1.84×10^{-6}	1.84×10^{-6}
8	1.63×10^{-6}	1.63×10^{-6}
9	1.42×10^{-6}	1.42×10^{-6}
10	1.21×10^{-6}	1.21×10^{-6}
11	1.10×10^{-6}	1.10×10^{-6}
12	9.86×10^{-7}	9.86×10^{-7}
13	8.74×10^{-7}	8.74×10^{-7}
14	7.62×10^{-7}	7.62×10^{-7}
15	6.50×10^{-7}	6.50×10^{-7}
16	5.20×10^{-7}	5.20×10^{-7}
17	3.90×10^{-7}	3.90×10^{-7}
18	2.60×10^{-7}	2.60×10^{-7}
19	1.30×10^{-7}	1.30×10^{-7}
20	2.00×10^{-7}	2.00×10^{-7}
21	4.60×10^{-7}	5.00×10^{-7}
22	4.50×10^{-7}	5.00×10^{-7}
23	4.40×10^{-7}	4.90×10^{-7}
24	4.30×10^{-7}	4.90×10^{-7}
25	4.22×10^{-7}	4.88×10^{-7}

* A lognormal distribution has been chosen to represent the uncertainty in the dose coefficients, with the geometric means given above and a geometric standard deviation of 1.7.

† The dose coefficients represent the dose per unit of I-131 activity ingested. Since after ingestion, iodine is transferred entirely into blood, these dose coefficients represent the dose per unit of I-131 activity in blood. These coefficients can be used to determine the dose from inhalation, because the activity of I-131 transferred to blood after inhalation is estimated as d by equation described by Equation A.25. (See also discussion in Section A.4.2)

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APPENDIX B

ESTIMATED CONCENTRATIONS OF IODINE-131 IN FOOD PRODUCTS

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B. ESTIMATED CONCENTRATIONS OF IODINE-131 IN FOOD PRODUCTS

B.1 Summary of Estimated Concentrations in Food Products

The concentrations of I-131 in air summarized in Section 5.3 are used to estimate doses from inhalation and consumption of food products. This appendix summarizes concentrations of I-131 in major food products (i.e., milk, vegetables, meat, and eggs) that contribute to estimated ingestion doses. Since consumption of milk is the main exposure pathway for I-131, concentrations in milk are presented in more detail. For the other food products, average concentrations over the 3 years of release are presented.

Estimated concentrations of I-131 in food products are given at seven representative Idaho locations: Terreton, Roberts, Idaho Falls, Blackfoot, Atomic City, Arco and Howe. These locations represent (with some approximation) communities near the INEL with the largest number of inhabitants in each 22.5-degree sector that has a community within 50 miles of the Idaho Chemical Processing Plant (ICPP) (see Section 4, Figure 4.1).

The highest concentrations of I-131 in food products were estimated at Mud Lake, which also is the location where the highest average concentration in air was estimated. The estimated concentrations in milk averaged over the period of releases (1957-1959) for each milk category assumed in the exposure scenarios described in Section 4 are given in Table B.1. The different categories of milk are described as follows:

- ***Backyard cow milk*** represents milk from one or two dairy cows raised by the owner as a private source of milk.

- ***Goat milk*** represents milk from one or two goats raised by the owner as a private source of milk.

- ***Local commercial milk*** represents milk from a herd of dairy cows raised locally. Fresh milk from many cows is mixed and is available for other people living in the same area.

- ***Regional commercial milk*** represents milk a person can purchase from a store, and it is obtained from several large dairy farms. For the region around INEL, such large dairy farms are located within 10 miles of the Snake River, where water is available.

Table B.1 Average concentrations of iodine-131 in different types of milk at selected locations around the Idaho National Engineering Laboratory

Milk category	Average concentration in milk [Bq L^{-1}]* 95% Confidence Interval		
	Lower bound	Central Estimate†	Upper bound
Terreton, Idaho			
Backyard cow milk	0.47	2.0	10
Local commercial milk	0.38	1.3	4.8
Goat milk	1.5	10.1	45
Regional commercial milk	0.053	0.16	0.5
Roberts, Idaho			
Backyard cow milk	0.16	0.64	3.3
Local commercial milk	0.14	0.42	1.5
Goat milk	0.54	3.3	14
Regional commercial milk	0.053	0.16	0.54
Idaho Falls, Idaho			
Backyard cow milk	0.060	0.25	1.2
Local commercial milk	0.051	0.16	0.56
Goat milk	0.20	1.27	6.1
Regional commercial milk	0.053	0.16	0.54
Blackfoot, Idaho			
Backyard cow milk	0.044	0.19	0.93
Local commercial milk	0.039	0.12	0.43
Goat milk	0.15	0.93	4.3
Regional commercial milk	0.053	0.16	0.54
Atomic City, Idaho			
Backyard cow milk	0.35	1.4	8.7
Local commercial milk	0.24	0.95	4.0
Goat milk	0.95	7.5	42
Regional commercial milk	0.053	0.16	0.54
Arco, Idaho			
Backyard cow milk	0.20	0.92	4.6
Local commercial milk	0.17	0.58	2.0
Goat milk	0.71	4.5	20
Regional commercial milk	0.053	0.16	0.54
Howe, Idaho			
Backyard cow milk	0.45	2.2	11
Local commercial milk	0.41	1.4	4.7
Goat milk	1.6	10	49
Regional commercial milk	0.053	0.16	0.54

* Estimated concentrations in milk are averaged over the three years of release (1957-1959).

† 50^{th} percentile of probability distribution function describing the uncertainty in estimated concentrations.

An example of the estimated concentrations of I-131 in milk over time is shown in Figure B.1. The dependence of the concentrations on time is somewhat different from that of the concentration in air (see Figure 5.4), because it includes other time-dependent parameters, such as the availability of pasture grass. Concentrations of I-131 in eggs, beef, and fresh leafy vegetables averaged over the period of release (1957-1959) are presented in Table B.2.

All concentrations of I-131 in food products reported in this appendix are concentrations at the time of collection of the food product. That is, the concentration in milk is representative of the time of milking, the concentration in beef is representative of the time of slaughter, the concentration in vegetables applies to the time of harvesting, and the concentration in eggs applies to the time of collection. Concentrations at the time of consumption are lower than estimated values at the time of collection due to radioactive decay of I-131 (half-life of 8.04 day) during storage time and reductions of the amount of I-131 by food processing (e.g., washing of vegetables, cooking, etc.).

B.2 Validation of Estimated Concentrations of Iodine-131 in Milk

An environmental monitoring program has been in operation at INEL since the mid-1950s. Concentrations of I-131 have been measured in various environmental media during the days following known atmospheric releases from the ICPP and other facilities or operations at INEL. For example, from May 29 through June 26, 1958, milk samples were collected from farms near Mud Lake and in other areas around INEL to monitor releases of I-131 from RaLa run No. 14 on May 28, 1958 and run No. 15 on June 2, 1958 (DOE 1991, AEC 1959). Such data can be used to test the validity of concentrations of I-131 estimated in the dose reconstruction.

Measured concentrations of I-131 in milk should represent overestimates of the true value because they are based on gross gamma counts. After subtraction of a background count rate, the remainder is assumed to be due to I-131. It is likely that some of the gross gamma count rate was due to shorter-lived radioiodines (e.g., I-132 and I-133) or other radionuclides.

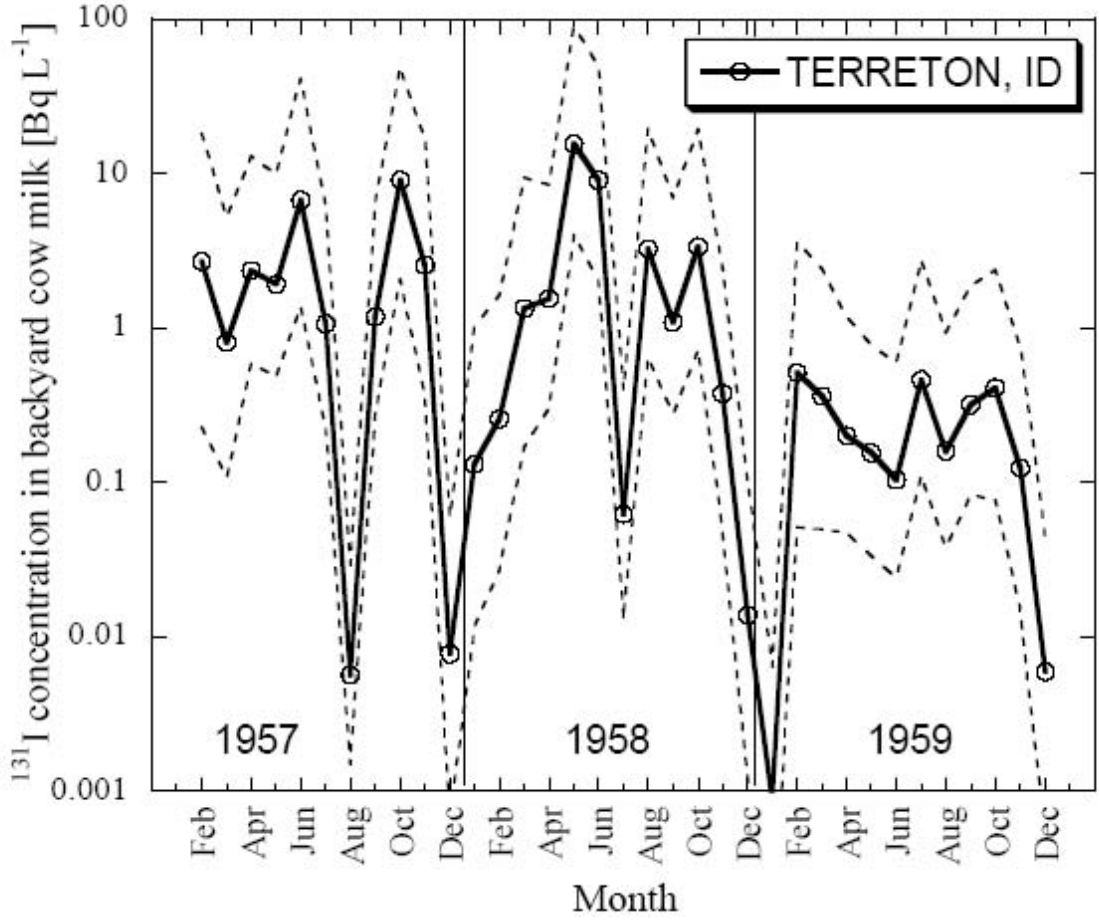

Figure B.1 Monthly averaged concentration of iodine-131 in backyard cow milk at one location as a function of time during the years of releases included in this study (1957-1959)

Table B.2 Average concentrations of iodine-131 in other food products at selected locations around the Idaho National Engineering Laboratory

Community or town	95% confidence interval		
	Lower bound	Central Estimate*	Upper bound
	Average concentrations in eggs† [Bq kg^{-1}]		
Arco	0.024	0.67	10
Atomic City	0.061	1.32	27
Blackfoot	0.0062	0.15	2.5
Howe	0.062	1.4	25
Idaho Falls	0.0096	0.24	4.5
Roberts	0.027	0.63	11
Terreton	0.077	1.56	27
	Average concentrations in beef† [Bq kg^{-1}]		
Arco	0.032	0.27	2.2
Atomic City	0.059	0.46	3.9
Blackfoot	0.0074	0.057	0.44
Howe	0.094	0.63	4.9
Idaho Falls	0.010	0.078	0.57
Roberts	0.026	0.20	1.5
Terreton	0.094	0.61	4.8
	Average concentrations in leafy vegetables† [Bq kg^{-1} fresh mass]		
Arco	0.55	1.5	4.7
Atomic City	0.92	3.1	14
Blackfoot	0.15	0.37	1.1
Howe	1.4	3.8	11
Idaho Falls	0.23	0.58	1.7
Roberts	0.63	1.5	4.3
Terreton	1.6	4.1	12

* 50th percentile of probability distribution function describing the uncertainty in estimated concentrations.

† Estimated concentrations are averaged over the three years of release (1957-1959).

DOE (1991) reports that 12 of the 15 samples collected in May and June of 1958 that contained above-background levels of I-131 were collected on May 29 and 30, after RaLa run No. 14 on May 28. Two samples from the location of highest concentration in milk (presumably Mud Lake) averaged 1,780 pCi L^{-1} (66 Bq L^{-1}).

Since our models provide monthly averaged concentrations in milk, the measurement reported above was adjusted to give an estimated average for the month of May 1958 by assuming that (1) the measurements were taken on May 29, 1958 and the I-131 came from the RaLa releases on May 28, 1958, (2) additional iodine was deposited on grass and transferred to milk due to releases on May 29 and 30 (this iodine was not measured by the above milk sample), and (3) the half-time of I-131 on vegetation is about 5 days (see Appendix A). The average concentration of I-131 in milk for the month of May 1958 estimated this way is 8.9 Bq L^{-1}. This average concentration includes only I-131 from RaLa run No. 14, and it assumes a concentration equal to zero for all days in May before May 29.

The average concentration of I-131 in milk at Mud Lake during May 1958 predicted by our model, including only I-131 from RaLa run No. 14, is 6.6 Bq L^{-1} (95% C.I. = 1.6 – 31 Bq L^{-1}). The estimated concentration in milk based on the measurements given above is 35% larger than the central value of the predicted concentration in milk and lies within the 95% confidence interval. This result shows that estimated concentrations of I-131 in milk produced in this study are not contradicted by relevant measurements.

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APPENDIX C

PRELIMINARY ASSESSMENTS OF DOSES FROM OTHER RADIONUCLIDES AND DOSES FROM ONSITE EXPOSURES

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C. PRELIMINARY ASSESSMENTS OF DOSES FROM OTHER RADIONUCLIDES AND DOSES FROM ONSITE EXPOSURES

C.1 Introduction

The dose reconstruction for releases of radionuclides from the Idaho Chemical Processing Plant (ICPP) presented in this report has focused on I-131 and exposures of the public at locations beyond the INEL site boundary. This appendix presents preliminary assessments of (1) offsite doses to the public from radionuclides released from the ICPP other than I-131, and (2) doses in assumed scenarios for onsite exposure of the public. These assessments are intended to provide bounding estimates of doses, rather than central estimates and their associated uncertainties that would be obtained in more detailed and realistic dose reconstructions. Such bounding estimates can be used to judge the potential importance of offsite doses from radionuclides other than I-131 and exposures of the public on the INEL site.

C.2 Preliminary Assessment of Offsite Doses from Other Radionuclides

The decision to focus the dose reconstruction on I-131 was based on a screening analysis to select radionuclides of potential concern in releases from the ICPP during the years 1957-1959 (Kocher 2005b). That analysis, which is summarized in Section 3, indicated that I-131 is by far the most important radionuclide when it is assumed that members of the public who resided near the INEL site boundary consumed substantial quantities of locally produced milk and other foods. Those types of scenarios result in much higher estimates of dose than other scenarios for exposure beyond the site boundary that do not involve consumption of locally produced foodstuffs.

Twelve additional radionuclides listed in Table 3.1 (see Section 3) were identified in the previous screening analysis (Kocher 2005b) for consideration in a detailed dose reconstruction for releases from the ICPP. In this section, we investigate the potential importance of those radionuclides to offsite exposures of the public by comparing dose estimates for I-131 in an exposure scenario for a rural resident discussed in Section 7.1.1 with calculated screening doses from I-131 and the other radionuclides (Kocher 2005a, 2005b). Such a comparison is valid because the exposure scenario for a rural resident, who is assumed to obtain a large fraction of intakes of milk, meat, and garden vegetables from sources near the INEL site boundary, is essentially the same as the scenario assumed in the screening analysis to select the radionuclides of concern (IAEA 2001).

In the dose reconstruction for I-131, doses to a rural resident were estimated at several locations (see Section 7.1.1, Table 7.1). One of those locations (Atomic City, Idaho) is the same as the receptor location assumed in the previous screening analysis to select radionuclides of concern (Kocher 2005a). Therefore, because the screening methodology is intended to result in substantial overestimates of dose to a rural resident (Kocher 2005a, IAEA 2001), a comparison of doses from I-131 at Atomic City obtained in the two analyses provides an indication of the degree of overestimation of doses obtained in the screening analysis. Such a comparison then can be used to evaluate the potential importance of doses from radionuclides other than I-131.

In the dose reconstruction for I-131 discussed in Section 7.1.1, an upper confidence limit of the effective dose to a resident of Atomic City who was born in 1957 and drank 2 to 4 glasses of goat's milk per day during the period of releases is 0.68 cSv, and the upper confidence limit of the dose to the thyroid is 14 cSv, or a factor of 20 higher than the effective dose. Doses at Atomic City are similar to those at Terreton (see Section 7.1.1, Table 7.4). For a person born in 1957 who drank the same amount of milk from a backyard cow, upper confidence limits of the effective and thyroid doses are about a factor of 4 less. Doses to an individual born in 1957 are the highest of any age group in a scenario in which the milk consumption rate is assumed to be the same throughout the period of releases from the ICPP.

In comparison, the effective dose from I-131 calculated in the previous screening analysis to select radionuclides of concern (Kocher 2005b) is 5.7 cSv (see Table C.1), and the dose to the thyroid is 114 cSv. Because the screening doses for I-131 are values for a young child (IAEA 2001), they can be compared with upper confidence limits of doses obtained in the dose reconstruction, as given above.

Table C.1 Screening effective doses estimated in previous analysis to select radionuclides of potential concern in releases from the Idaho Chemical Processing Plant*

Nuclide	Effective dose (cSv)†	Nuclide	Effective dose (cSv)†
Sr-89	0.10	I-133	0.013
Sr-90	0.033	Ba-140	0.16
Y-91	0.076	Ce-141	0.018
Zr-95	0.045	Ce-144	0.056
Nb-95	0.011	Pr-143	0.026
Ru-103	0.018	Pu-238	0.017
I-131	5.7		

* See Kocher 2005b. Screening doses are estimated on the basis of upper confidence limits of estimated releases from the ICPP (see Section 2.1, Table 2.1, and Section 2.2, Table 2.2) and models of atmospheric transport and exposure pathways that should result in substantial overestimates of dose (Kocher 2005a, IAEA 2001).

† Screening effective doses are obtained from screening risks given in Table 3-1 of Kocher 2005b and assumption for purposes of screening that the lifetime risk of cancer incidence per unit effective dose is 0.1 Sv^{-1} (Kocher 2005a).

We find that the previous screening analysis apparently overestimates doses from I-131 by *at least* a factor of 5.7/0.68, or about a factor of 8, in a scenario involving a high consumption rate of goat's milk by a young child at Atomic City, and the degree of overestimation of doses in the screening analysis apparently is *at least* a factor of about 30 in a scenario involving a high consumption rate of milk from a backyard cow. The latter scenario presumably was a more common occurrence near INEL.

For the other 12 radionuclides of potential concern in releases from the ICPP (see Table C.1), screening estimates of effective doses (Kocher 2005b) are 0.16 cSv or less, and the effective dose from all other radionuclides combined is 0.57 cSv, or a factor of 10 less than the screening effective dose from I-131. By assuming that effective doses that would be obtained in a detailed and more realistic dose reconstruction would be at least a factor of 8 less than calculated screening doses, we find that the effective dose at Atomic City from all radionuclides other than I-131 is highly unlikely to exceed 0.07 cSv. Given that the dose to any organ from all other

radionuclides combined would not exceed the effective dose by more than a factor of 10, the highest dose to any organ would not exceed about 0.7 cSv.[10] Again, these doses are bounding estimates for a worst-case scenario involving a high consumption rate of goat's milk by a young child.

In the dose reconstruction for I-131, upper confidence limits of estimated doses at Mud Lake, Idaho, which is close to Terreton (see Section 4, Figure 4.1), are about 25% higher than the corresponding estimates at Atomic City. Mud Lake is the location beyond the INEL site boundary where the median of the average concentration of I-131 in air is the highest (see Section 5.3, Table 5.1). Thus, in a worst-case scenario involving a high consumption rate of goat's milk, the effective dose at Mud Lake from all radionuclides other than I-131 is highly unlikely to exceed about 0.09 cSv, and the dose to any organ would not exceed about 0.9 cSv.

In the more common scenario involving a high consumption rate of cow's milk by a young child, bounding estimates of the effective dose and the dose to any organ from all radionuclides other than I-131 would be substantially less than bounding estimates of doses from consumption of goat's milk. The reduction in bounding estimates of doses depends on the relative importance of different ingestion, inhalation, and external exposure pathways for the other radionuclides in a rural resident scenario involving consumption of cow's milk, but the reduction should be at least a factor of 2 to 3 for all radionuclides combined, given the likely importance of the milk pathway (e.g., see Section 3, Table 3.1, and Table C.3 in the following section).

In summary, the discussions in this section have led to two conclusions. First, the screening methodology that was used to select radionuclides of potential importance in releases from the ICPP provides considerable overestimates of doses that would be obtained in a detailed and more realistic dose reconstruction. Second, on the basis of estimated doses from I-131 obtained in the dose reconstruction for a rural resident scenario and the apparent degree of overestimation of doses incorporated in the screening methodology, doses from all other radionuclides of concern would be rather low in scenarios for exposure of the public at locations beyond the INEL site boundary. Our preliminary assessment has indicated that the effective dose from all other radionuclides combined almost certainly would be less than 0.1 cSv, and that the dose to any organ almost certainly would be less than 1 cSv.

C.3 Preliminary Assessment of Onsite Exposure Scenarios

All scenarios considered in Section 7.1 involve exposure of the public at locations beyond the INEL site boundary. Those scenarios considered exposure of young children as well as adults, and doses to young children from exposure to I-131 were substantially higher. As described in Section 4, additional scenarios for exposure of the public at locations on the INEL site were considered in the dose reconstruction. Those scenarios, which apply only to adults, involve exposures of the following:

[10] An assumption that the ratio of the highest organ dose to the effective dose from all other radionuclides combined would not exceed 10 is based on considerations that (1) this ratio is substantially less than 20 for all radionuclides except I-133; (2) the screening effective dose from I-133 is less than the value for nearly all other radionuclides of concern (Kocher 2003b), and (3) the organ receiving the highest dose is not the same for all radionuclides.

- An onsite rancher
- A hunter who consumes meat obtained from game that grazed on the site, but who does not come onto the site while hunting
- A one-time or regular visitor to the site

Such scenarios are potentially important because onsite exposure (including exposure of livestock or game within the INEL site boundary) occurred at locations closer to the ICPP than offsite locations, and airborne concentrations of radionuclides over much of the site were higher than at any location beyond the boundary. Adults who are included in the assumed scenarios also could have been exposed as rural or urban residents beyond the site boundary (see Sections 7.1.1 and 7.1.3). However, the scenarios listed above are concerned only with exposure on the INEL site, and they do not include additional exposures that could have occurred at locations beyond the site boundary.

In a previous report (Kocher 2005b), the assumed scenarios for onsite exposure were evaluated to determine whether application of the screening methodology (Kocher 2005a, 2005b) summarized in Section 3 to those scenarios would result in selection of additional radionuclides of potential importance in releases from the ICPP, other than those listed in Table C.1. That evaluation indicated that no additional radionuclides would be selected by screening when relevant exposure pathways and reasonable exposure times in each scenario were taken into account.

This section presents a preliminary assessment of the assumed onsite exposure scenarios to investigate whether doses in those scenarios could be important. This assessment takes into account all radionuclides listed in Table C.1, even though only I-131 is included in the detailed dose reconstruction presented in Section 7.1. The other radionuclides must be included because doses to offsite members of the public from I-131 are dominated by the consumption of contaminated milk, but this pathway does not occur in onsite exposure scenarios. Thus, when relevant pathways in onsite exposure scenarios are considered, doses from other radionuclides could greatly increase in importance relative to doses from I-131 in those scenarios. An assessment of the importance of each onsite exposure scenario is based on the following:

- Upper confidence limits of average airborne concentrations of I-131 at various locations on the INEL site and at Atomic City, Idaho, as obtained in the detailed dose reconstruction (see Section 5.3, Table 5.1)
- An upper confidence limit of the effective dose from I-131 in a rural resident scenario at Atomic City, as obtained in the dose reconstruction discussed in Section 7.1.1
- Average airborne concentrations of I-131 at Atomic City assumed in the screening analysis to select radionuclides of potential concern in releases from the ICPP (Kocher 2005a)
- Screening effective doses given in Table C.1 for the other radionuclides of concern (Kocher 2005b), which were calculated by assuming an exposure scenario at Atomic City

that is essentially the same as a rural resident scenario used in the dose reconstruction for I-131 presented in Section 7.1.1

- Consideration of relevant exposure pathways and reasonable exposure times in each scenario for onsite exposure

The information described above can be used to obtain bounding estimates of doses in onsite exposure scenarios because the screening analysis to select radionuclides of concern incorporates assumptions about releases, atmospheric transport, and exposures from assumed pathways, especially ingestion pathways, that should result in substantial overestimates of dose (Kocher 2005a, IAEA 2001).

The preliminary assessment of onsite exposure scenarios also assumes that airborne concentrations at any location per unit release from the ICPP are the same for all radionuclides. That assumption is supported by calculations using the CALPUFF code (see Section 5) that take into account possible differences in deposition rates of iodine and other radionuclides.

C.3.1 Onsite Rancher

The assumed scenario for exposure of an onsite rancher is based on the consideration that portions of the INEL site were open to controlled grazing of beef cattle and sheep during the period of operations at the ICPP (Apostoaei and Reed 2005). An onsite rancher is assumed to be exposed at locations close to the Big and Little Lost River sink area, which is located about 16-24 km (10-15 miles) north-northeast of the ICPP. Relevant exposure pathways in the scenario include consumption of contaminated beef, inadvertent soil ingestion, inhalation, and external exposure.

For I-131 released from the ICPP, upper confidence limits of average airborne concentrations at the assumed location of an onsite rancher and at Atomic City are 0.015 and 0.011 Bq m^{-3}, respectively (see Section 5.3, Table 5.1). Thus, the upper confidence limit at the location of an onsite rancher is a factor of 1.4 higher than at Atomic City. This increase also can be applied to radionuclides other than I-131, because uncertainties in average airborne concentrations of I-131 are determined primarily by uncertainties in the atmospheric transport model, which are essentially the same for all radionuclides. The uncertainty in the total release of I-131 (see Section 2.1, Table 2.1) is unimportant compared with the uncertainty in the average concentration in air at a given location per unit release.

For a rural resident scenario at Atomic City, the dose reconstruction for I-131 gave an upper confidence limit of the effective dose to an adult male who consumed 2 to 4 glasses of milk per day from a backyard cow of about 2×10^{-2} cSv (see Table C.2).[11] Consumption of cow's milk by a rural resident, rather than goat's milk, is assumed to be consistent with the milk pathway assumed in the screening methodology (IAEA 2001), which is used in this assessment to estimate upper bounds of doses to an onsite rancher from radionuclides other than I-131. It also

[11]The upper confidence limit of the effective dose to a young child is nearly an order of magnitude higher. However, only the dose to an adult is relevant to an assessment of dose to an onsite rancher.

seems likely that consumption of milk from a backyard cow was more common than consumption of goat's milk.

As indicated in Table C.2, most of the effective dose to an adult male rural resident at Atomic City is due to the consumption of milk, and the upper confidence limit of the effective dose from relevant pathways for an onsite rancher (consumption of beef, soil ingestion, inhalation, and external exposure) is less than 2×10^{-3} cSv. About half of that dose is due to inhalation and the remainder is due about equally to the beef and soil ingestion pathways.[12] In addition, the fraction of the time that a rancher spent on the INEL site presumably did not exceed 25%, and this exposure time reduces doses from soil ingestion and inhalation compared with doses to a rural resident from those pathways by about a factor of 4. By taking into account upper confidence limits of doses to an adult male rural resident at Atomic City from the beef, soil ingestion, and inhalation pathways, the bounding estimate of the exposure time for an onsite rancher, and the higher average airborne concentration of I-131 at the onsite receptor location, we obtain an upper bound of the effective dose to an onsite rancher from I-131 of about 6×10^{-4} cSv.

We now consider the effective dose to an onsite rancher from the other radionuclides of concern in releases from the ICPP. As indicated in Table C.1, effective doses to a rural resident scenario at Atomic City estimated using the screening methodology (Kocher 2005a, 2005b) range from 0.011 cSv for Nb-95 to 0.16 cSv for Ba-140. In comparison, the screening effective dose from I-131 is 5.7 cSv, which indicates the dominant importance of I-131 in that scenario, as assumed in the dose reconstruction.

Table C.2 Estimated effective doses from different exposure pathways for an adult male rural resident at Atomic City

	Effective dose (cSv) 95% confidence interval		
	Lower bound	**Central estimate**	**Upper bound**
Backyard cow milk (high consumption rate*)	3.4×10^{-4}	2.2×10^{-3}	1.6×10^{-2}
Inhalation†	2.0×10^{-5}	1.0×10^{-4}	5.5×10^{-4}
Eggs	2.2×10^{-6}	8.5×10^{-5}	2.4×10^{-3}
Vegetables	5.5×10^{-6}	2.9×10^{-5}	1.7×10^{-4}
Inadvertent soil ingestion	7.0×10^{-7}	1.5×10^{-5}	2.4×10^{-4}
Beef	6.5×10^{-7}	1.2×10^{-5}	2.2×10^{-4}

* Two to four 8-oz glasses per day.

† Assumes large amounts of time spent outdoors.

For radionuclides other than I-131, screening effective doses to a rural resident at Atomic City (Kocher 2005b) can be used to estimate upper bounds of effective doses to an onsite rancher by making the following four adjustments.

The first adjustment takes into account the degree of overestimation of airborne concentrations of radionuclides that is incorporated in the atmospheric transport model used in the screening

[12] External dose from I-131 is only about 1% of the dose from inhalation and, thus, is negligible.

methodology (Kocher 2005a). In calculating a screening dose from I-131, the average airborne concentration at Atomic City was assumed to be 0.38 Bq m^{-3} (Kocher 2005a). That concentration was based in part on an assumption that releases from the ICPP occurred during a single year (Kocher 2005a).[13] Given that releases occurred over a 3-year period, the average concentration at Atomic City during the release assumed in the screening analysis becomes 0.13 Bq m^{-3}. In comparison, as noted above, the upper confidence limit of the average concentration at Atomic City obtained in the dose reconstruction is 0.011 Bq m^{-3}. Therefore, screening effective doses for radionuclides other than I-131 given in Table C.1 should be reduced by a factor of 0.13/0.011 = 12 to account for the degree of overestimation of upper confidence limits of airborne concentrations in the screening analysis. The resulting doses are still upper bounds for a rural resident at Atomic City, because they are calculated on the basis of estimated releases from the ICPP that are intended to be upper confidence limits (see Section 2.1, Table 2.1, and Section 2.2, Table 2.2), an upper confidence limit of the average airborne concentration at Atomic City per unit release, and models of exposure pathways that are intended to overestimate doses per unit concentration in air.

The reduction in screening effective doses by a factor of 12 for radionuclides other than I-131 that applies at Atomic City also can be applied at the location of an onsite rancher. That conclusion is based mainly on the consideration that uncertainties in average airborne concentrations of I-131 at Atomic City and at the Big and Little Lost River sink area are about the same (i.e., the upper confidence limit differs from the median by a factor of 2 to 3 at both locations; see Section 5.3, Table 5.1). Given the greater importance of uncertainties in the atmospheric transport model used in the dose reconstruction compared with the uncertainty in the total release of I-131, as noted previously, the similarity in uncertainties in average airborne concentrations of I-131 indicates that uncertainties in the atmospheric transport model are about the same at the two locations. Reduction of the screening doses in Table C.1 by a factor of 12 results in effective doses at the location of an onsite rancher that range from 9×10^{-4} cSv for Nb-95 to 1.3×10^{-2} cSv for Ba-140. Again, these doses are calculated using upper confidence limits of releases from the ICPP and screening models for exposure pathways that are intended to overestimate doses per unit concentration in air (IAEA 2001), and they include all exposure pathways in a rural resident scenario.

The second adjustment takes into account the higher average air concentrations at the location of an onsite rancher compared with the concentrations at Atomic City. As noted previously, the upper confidence limit of average airborne concentrations at the location of an onsite rancher is a factor of 1.4 higher than at Atomic City. This adjustment results in effective doses at the assumed receptor location, again including all exposure pathways in a rural resident scenario, that range from 1.3×10^{-3} cSv for Nb-95 to 1.9×10^{-2} cSv for Ba-140.

The last two adjustments to effective doses calculated in the previous screening analysis involve eliminating contributions from consumption of garden crops and milk, given that those exposure pathways are not relevant to an onsite rancher, and reducing contributions from inhalation and external exposure pathways by a factor of 4 to account for the reduced onsite exposure time. The importance of the two adjustments is radionuclide-specific. On the basis of the percent

[13] An assumption in the screening analysis of releases over a single year, although arbitrary, is unimportant because the total dose in that analysis depends only on the total release and is independent of the assumed duration of the release.

contributions to screening effective doses in a rural resident scenario from each exposure pathway (garden crops, milk, meat, external exposure, and inhalation) given in Table C.3, the adjustments to account for relevant pathways and a reduced exposure time result in reductions in estimated effective doses that range from a factor of 2.2 for Ru-103 to about a factor of 200 for I-133.

Application of all the adjustments described above to screening effective doses in Table C.1 for a rural resident scenario at Atomic City gives the following results for an onsite rancher scenario:

- The effective dose from individual radionuclides ranges from about 8×10^{-6} cSv for ^{133}I to 1.5×10^{-3} cSv for ^{91}Y
- The effective dose from all radionuclides combined, other than I-131, is about 7×10^{-3} cSv.

Table C.3 Radionuclides selected by screening for inclusion in dose reconstruction for releases from the Idaho Chemical Processing Plant and percent contributions from different exposure pathways to calculated screening risks*

Nuclide	Half-life	Contribution [percent]†				
		Crops‡	Milk	Meat	External§	Inhalation
Sr-89	50.5 d	47	41	11	1	0
Sr-90	29.1 y	18	60	20	2	0
Y-91	58.5 d	81	1	17	1	0
Zr-95	64.0 d	44	0	0	55	0
Nb-95	35.1 d	56	0	0	44	0
Ru-103	39.3 d	43	0	41	15	0
I-131	8.04 d	12	80	8	0	0
I-133	20.8 h	0	98	0	1	1
Ba-140	12.74 d	51	41	2	6	0
Ce-141	32.5 d	88	7	0	4	1
Ce-144	284 d	86	7	0	6	1
Pr-143	13.6 d	95	1	3	0	0
Pu-238	87.7 y	58	0	0	0	42

* Obtained from Table I-I of IAEA 2001 or calculated using exposure pathway models and parameter values given in IAEA 2001.

† Percent contributions from different exposure pathways do not necessarily apply to a more realistic dose reconstruction.

‡ Includes ingestion of contaminated garden vegetables and direct ingestion of contaminated soil.

§ Includes external exposure to atmospheric cloud and contaminated ground surface.

These estimates can be considered upper bounds of effective doses to an onsite rancher because they are calculated on the basis of upper confidence limits of estimated releases of each radionuclide, an upper confidence limit of the average airborne concentration at the receptor location per unit release, a screening model for the beef pathway that is intended to overestimate

intakes of radionuclides per unit concentration in air, and an assumed exposure time for inhalation and external exposure pathways that should overestimate actual exposure times.[14] Thus, on the basis of a preliminary assessment described above, an estimated upper bound of the effective dose to an onsite rancher from all radionuclides combined, including I-131, is about 8×10^{-3} cSv. Iodine-131 is unimportant in this scenario compared with the other radionuclides of concern.

C.3.2 Hunter of Onsite Game

Hunting was not permitted on the INEL site during the period of releases from the ICPP. However, a hunter beyond the site boundary could have consumed meat from game, such as prong-horned antelope, that ranged over the site and outside the boundary. In this assessment, we assume that game animals wandered freely over the INEL site and were exposed to average concentrations of airborne radionuclides on the site.

As in the preliminary assessment of effective doses to an onsite rancher, we first consider the dose to a hunter from I-131. On the basis of the upper bound of the effective dose to an onsite rancher from I-131 of about 6×10^{-4} cSv given in the previous section, the similarity in upper confidence limits of average airborne concentrations of I-131 over the INEL site (0.013 Bq m^{-3}) and at the location of an onsite rancher (0.015 Bq m^{-3}) (see Section 5.3, Table 5.1), and the consideration that only the meat pathway occurs in the hunter scenario, we can conclude without further analysis that an upper bound of the effective dose to a hunter from I-131 is somewhat less than the upper-bound estimate for an onsite rancher.

We next consider the dose to a hunter from all other radionuclides of concern. As in the assessment for an onsite rancher described in the previous section, a preliminary assessment of the effective dose to a hunter is based on results of the dose reconstruction for I-131 in a rural resident scenario at Atomic City (see Section 7.1.1), screening effective doses for the other radionuclides of concern in the same scenario at Atomic City (Kocher 2005a, b) given in Table C.1, and consideration of differences in relevant exposure pathways in the rural resident and hunter scenarios. The assessment proceeds as follows:

- As in the assessment for an onsite rancher, screening effective doses given in Table C.1 for a rural resident scenario at Atomic City are reduced by a factor of 12 to account for the degree of overestimation of upper confidence limits of average airborne concentrations at Atomic City and other locations that are incorporated in the atmospheric transport model used in the screening methodology (Kocher 2005a).

[14]Not all parameters included in the screening models for different pathways (IAEA 2001) tend to provide overestimates of doses per unit concentration of radionuclides in air. Dose coefficients for internal or external exposure (doses per unit activity intake or dose rates per unit activity concentration in air or on the ground surface) are intended to be best estimates. In addition, the screening models include inadvertent soil ingestion in the pathway involving consumption of garden crops, and this pathway thus is not included in the relevant pathways for an onsite rancher. Neglect of the soil ingestion pathway has only a small effect on the total dose from all radionuclides of concern combined.

- To account for the higher upper confidence limit of average airborne concentrations over the INEL site compared with Atomic City, screening effective doses in a rural resident scenario (Kocher 2005b) are increased by a factor of 0.013/0.011 = 1.2.

- Only the dose from the meat pathway is taken into account, because no other exposure pathways included in a rural resident scenario apply to the hunter scenario.

By applying these adjustments to screening effective doses in a rural resident scenario at Atomic City (Table C.1) and taking into account contributions from different exposure pathways (Table C.3), an upper bound of the effective dose to a hunter from all radionuclides combined, including I-131 which is unimportant, is about 4×10^{-3} cSv.

An additional factor that could be considered in evaluating doses to a hunter is that upper confidence limits of intake-to-meat transfer coefficients in game animals could be substantially higher than transfer coefficients in beef that were assumed in the screening methodology (IAEA 2001), even though the latter are intended to provide overestimates of transfers to beef. That possibility is indicated by intake-to-meat transfer coefficients for several radionuclides in beef cows compared with more limited data on transfer coefficients in sheep and lambs, goats, and pigs (IAEA 1994). The comparison suggests that transfer coefficients in other farm animals could be as much as an order of magnitude higher than values in beef cattle. However, such increases are not seen in all farm animals or for all radionuclides, and the applicability of transfer coefficients in farm animals to game at INEL is unknown.

If we assume that upper bounds of intake-to-meat transfer coefficients in game that grazed on the INEL site could be an order of magnitude higher than values in beef cattle that were assumed in the screening methodology (IAEA 2001), an upper bound of the effective dose to a hunter would be about 4×10^{-2} cSv. However, it cannot be determined on the basis of available data whether such a higher estimate is credible in a hunter scenario at INEL.

C.3.3 Onsite Visitor

Two types of visitors to the INEL site are considered: a one-time visitor (e.g., an individual who takes a tour of the site) and a regular visitor who comes to the site frequently in performing job duties (see Section 4).

A regular visitor to the site is assumed to be an individual not employed at INEL, who was present at the Central Facilities Area (CFA) for a few hours each week while making deliveries of such goods as office supplies or food and drink. The CFA is located about 3.4 km (2.1 miles) from the ICPP and, thus, is much closer to the source than other locations of public exposure on the site. The time spent at the CFA by a delivery person is assumed to be 500 hours each year (Kocher 2005b), or about 6% of the exposure time for a rural resident outside the site boundary. That exposure time should be a bounding value for regular onsite visitors. Inhalation and external exposure are the only relevant pathways in this scenario.

As in the assessments in the previous two sections, we first consider the dose to a regular site visitor from I-131. The upper confidence limit of average airborne concentrations of I-131 at the CFA is 0.049 Bq m^{-3} (see Section 5.3, Table 5.1). That value is a factor of 4.5 higher than the

upper confidence limit at Atomic City of 0.011 Bq m^{-3}. At the upper limit of confidence, only about 3% of the effective dose to an adult rural resident at Atomic City of about 2×10^{-2} cSv is due to inhalation (see Table C.2). Taking into account the reduced exposure time for an onsite visitor compared with a rural resident, the higher average airborne concentration of I-131 at the CFA, and the contribution to the effective dose to a rural resident from inhalation, we obtain an upper bound of the effective dose to a regular onsite visitor from I-131 of about 2×10^{-4} cSv.

We next consider the dose to a regular onsite visitor from all other radionuclides of concern. Similar to the assessments for an onsite rancher and hunter, a preliminary assessment of the effective dose to a regular onsite visitor is based on results of the dose reconstruction for I-131 in a rural resident scenario at Atomic City (see Section 7.1.1), screening effective doses for the other radionuclides of concern in the same scenario at Atomic City (Kocher 2005a, b) given in Table C.1, and consideration of differences in relevant pathways in the rural resident and onsite visitor scenarios. The assessment proceeds as follows:

- As in the assessments for an onsite rancher or hunter, screening effective doses given in Table C.1 for a rural resident scenario at Atomic City are reduced by a factor of 12 to account for the degree of overestimation of upper confidence limits of average airborne concentrations at Atomic City and other locations that are incorporated in the atmospheric transport model used in the screening methodology (Kocher 2005a).

- To account for the higher upper confidence limit of average airborne concentrations at the CFA compared with Atomic City, screening effective doses in a rural resident scenario (Kocher 2005b) are increased by a factor of 4.5, as noted above.

- Only the doses from inhalation and external exposure pathways are taken into account, because no other exposure pathways included in a rural resident scenario apply to a site visitor, and the exposure time is assumed to be 6% of the exposure time in a rural resident scenario, as noted above.

By applying these adjustments to screening effective doses in a rural resident scenario at Atomic City (Table C.1) and taking into account contributions from different exposure pathways (Table C.3), an upper bound of the effective dose to a regular visitor from all radionuclides combined, including I-131 which is unimportant, is about 1×10^{-3} cSv.

A one-time visitor to the INEL site would be exposed for a much shorter time than a frequent visitor and, therefore, the dose to such an individual would be much lower. If we assume, for example, that a visiting scientist spent two weeks on the site (about 80 hours), the dose would be about 5% of the dose to a regular visitor given above, on average. Actual doses to a one-time visitor could be higher or lower than doses estimated only on the basis of an assumed exposure time, because airborne concentrations of radionuclides during a one-time visit could be higher or lower than average values over the 3-year period of releases from the ICPP.

C.3.4 Interpretation of Estimated Doses in Onsite Exposure Scenarios

Estimated effective doses in different scenarios for onsite exposure of the public obtained in this preliminary assessment are considered to be upper bounds, because they were calculated essentially as the product of upper confidence limits of total releases, airborne concentrations per unit release, and doses per unit concentration in air.[15] An upper confidence limit could be obtained by combining the variances of each quantity in quadrature. However, such an analysis is beyond the scope of a preliminary assessment because uncertainties in doses per unit concentration in air are radionuclide-specific and would need to be evaluated by taking into account uncertainties in dose coefficients (see footnote 5) as well as uncertainties in exposures per unit concentration in air.

C.3.5 Summary

Sections C.3.1 through C.3.3 have presented preliminary assessments of effective doses to members of the public in scenarios involving exposure on the INEL site. The assessments have shown that upper bounds of effective doses in scenarios for an onsite rancher, a hunter who consumes meat obtained from game that grazed on the site, and a one-time or regular visitor to the site are about 1×10^{-2} cSv or less. The estimated upper bounds are less than upper confidence limits of estimated doses from exposure of adult rural residents to I-131 at locations of highest exposure beyond the site boundary, and they are much less than upper confidence limits of doses to children at offsite locations. As noted in Section C.2, doses to any organ should not exceed effective doses by more than a factor of 10 and, therefore, should be less than about 0.1 cSv in all scenarios.

It is possible that the effective dose to a hunter could be as high as about 4×10^{-2} cSv. However, such a dose would be credible only if intake-to-meat transfer coefficients in game for radionuclides other than I-131 were an order of magnitude higher than transfer coefficients in beef cattle assumed in this assessment (IAEA 2001). That possibility is suggested by data on transfer coefficients in farm animals other than beef cattle (IAEA 1994). On the other hand, there are no data on transfer coefficients in game (e.g., prong-horned antelope) that would support an assumption of a large increase in dose, and such an assumption would be largely speculative.

[15]The product of upper confidence limits of independent quantities always results in an overestimate of the upper confidence limit of the product of those quantities that would be obtained when uncertainties in each quantity are propagated correctly.

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I-131 Thyroid Dose/Risk Calculator for NTS Fallout

Welcome to the Individual Dose and Risk Calculator for Nevada Test Site fallout

This calculator estimates the radiation dose from I-131 absorbed by your thyroid gland from nuclear tests conducted at the Nevada Test Site (NTS). The calculator also estimates your risk of thyroid cancer from this exposure.

Testing was conducted at the NTS from 1951 through 1992. However, only the above-ground tests conducted from 1951 through 1962, and some of the underground tests conducted from 1961 through 1970, produced appreciable I-131 fallout in the continental United States and are included in this calculator.

To use this calculator, you will need to supply:

- your date of birth and gender
- the states and counties that you lived in between 1951 and 1971
- the primary type of milk that you drank (cow or goat)

Note: Before you start the calculator, we urge you to read the Summary Information to learn more about the tests, about I-131 exposure, and about thyroid cancer risks.

If you were born after 1971, you were not affected by tests conducted at the Nevada Test Site. Your estimated I-131 dose from these tests is zero. You do not need to use the calculator.

Start Calculator

To ensure that all features of this calculator operate correctly, please use either Netscape Navigator 7.0 (or greater) or Internet Explorer 6.0 (or greater). Click the icon below to download the software:

Technical Documentation

Acknowledgements