APPENDIX F

**UNCERTAINTIES ASSOCIATED WITH ESTIMATING
SPECIFIC ISOTOPE RELEASES BASED
ON NONSPECIFIC MONITORING DATA**

APPENDIX F

UNCERTAINTIES ASSOCIATED WITH ESTIMATING SPECIFIC ISOTOPE RELEASES BASED ON NONSPECIFIC MONITORING DATA

As described in Section 2.4.1, from 1953 to 1973 only long-lived gross alpha activity was routinely monitored in Rocky Flats airborne effluents. Release estimates of specific isotopes for this time period were calculated from the reported long-lived gross alpha activity. Even after 1973, some isotopes such as Am-241 and Pu-241, were not routinely monitored and their annual release quantities have to be derived from the release estimates of Pu-239/240. Because of the limited information available, uncertainty and error might have been introduced into the estimation of source terms of these radionuclides. This appendix evaluates and quantifies this source of uncertainty.

Because different information and sources of uncertainty were involved in the calculation of release estimates of plutonium and uranium isotopes, they are discussed separately in the following sections.

F.1 Uncertainties Associated with Estimating Pu-239/240, Am-241 and Pu-241 Releases Based on Nonspecific Monitoring Data

1953-1973

Before 1973, only long-lived gross alpha activity was routinely monitored in Rocky Flats airborne effluents. As a result, it is necessary to assume that long-lived gross alpha activity sampled in effluents from buildings in which plutonium was handled consisted solely of those alpha-emitting radionuclides associated with weapons grade plutonium. Since gross alpha analysis is not specific to any radionuclide, the plant made attempts during the 1970s to determine the accuracy of this practice. As discussed in Section 2.2.2.1, results of studies carried out in the mid and late 1970s indicate the correlation between total long-lived alpha and plutonium measured was poor. However, if samples with activity below 0.002 pCi m^{-3} were excluded from the study, the correlation was very good. For the purpose of this evaluation, it was estimated that the uncertainty associated with this practice is ± 20 percent. In other words, in order to compensate for this source of uncertainty, an uncertainty factor with an uniform distribution that ranges from 0.8 to 1.2 was applied to the annual release estimates developed in Section 2.8.

As discussed in Section 2.4.1.1, in the development of source terms for Pu-239/240 and Am-241 between 1953 and 1973, long-lived gross alpha activities measured from plutonium buildings were partitioned into Pu-239/240 and Am-241 release estimates. This method is based on the assumptions that there is a constant ratio between the two radionuclides and that the ratio established in 1980s can be applied to long-lived alpha monitored in the earlier years. In order to account for the uncertainty associated with the development of this ratio, other sources of information were consulted. According to Table 2-35, annual airborne Am-241 emissions for each year from 1985 to 1989 were between 13 percent and 31 percent of the plutonium alpha activity release total for the same year. Theoretical calculations based on the initial purity and average age of plutonium handled at Rocky Flats indicate that the americium to plutonium activity ratio ranged from 10 percent to 20 percent.

For the purpose of this evaluation, uncertainty factors of source terms of Pu-239/240 and Am-241 were developed using 0.31 and 0.1 as the upper and lower bounds of the americium to plutonium activity ratio:

When $\quad$ $F_{Am} / F_{Pu} = 0.10$ $\quad$ and $\quad$ $F_{Am} + F_{Pu} = 1$

$\quad\quad\quad\quad\quad$ $F_{Am} = 0.09$ $\quad$ and $\quad$ $F_{Pu} = 0.91$

And when $\quad$ $F_{Am} / F_{Pu} = 0.31$ $\quad$ and $\quad$ $F_{Am} + F_{Pu} = 1$

$\quad\quad\quad\quad\quad$ $F_{Am} = 0.24$ $\quad$ and $\quad$ $F_{Pu} = 0.76$

Based on the calculation shown above, the upper and lower bounds of the uncertainty factor of Pu-239/240 release estimate were calculated to be 1.11 (0.91/0.82) and 0.93 (0.76/0.82), respectively. Similarly, the upper and lower bounds of the uncertainty factor of Am-241 release estimate were determined to be 1.33 (0.24/0.18) and 0.50 (0.09/0.18), respectively. It was assumed that these uncertainty factors are triangularly distributed with best estimates equal to 1.0.

The uncertainty factor of the source term of Pu-241 was calculated by combining the uncertainties associated with the release estimates of total plutonium alpha activity and the ratio of Pu-241 to total plutonium alpha activity. As shown above, the uncertainty factor associated with the source terms of Pu-239/240 was represented by a triangular distribution with an upper bound of 1.11, a best estimate of 1 and a lower bound of 0.93. The range of beta to alpha activity percentage was not available. Since Pu-241 decays into Am-241, it is reasonable to assume that the uncertainty associated with the ratio of Pu-241 to Pu-239/240 is the same as the uncertainty associated with the ratio of Am-241 to Pu-239/240. As described earlier, release ratio of Am-241 to Pu-239/240 ranged from 0.1 to 0.31, with a most likely value of 0.22. Therefore, the uncertainty factor of the ratio was assumed to have a triangular distribution with a best estimate of 1 and upper and lower bounds of 1.4 (0.31/0.22) and 0.45 (0.1/0.22), respectively. The uncertainty associated with the calculation of Pu-241 release estimates based on the long-lived gross alpha data reported between 1953 and 1973 was obtained by combining the two distributions by Monte Carlo simulation. This resulted in an uncertainty factor with a normal distribution with a mean of 0.96 and a standard deviation of 0.2.

1974-1984

During this period, Pu-239/240 was monitored directly by alpha spectral analysis. However, Am-241 and Pu-241 were not routinely measured and release estimates of these two isotopes have to be calculated from the source terms of Pu-239/240. As described in Section 2.4.1.2, this calculation was based on the ratios of Am-241 to Pu-239/240 and Pu-241 to Pu-239/240. The uncertainty factor of the ratio of Am-241 to Pu-239/240 was developed in the section above, and was represented by a triangular distribution with a best estimate of 1 and upper and lower bounds of 1.4 (0.31/0.22) and 0.45 (0.1/0.22), respectively.

Information regarding the range of ratios of Pu-241 to Pu-239/240 is not available. Since Pu-241 decays into Am-241, it was assumed that the uncertainty associated with the Pu-241 to Pu-239/240 ratio is the same as the uncertainty associated with the Am-241 to Pu-239/240 ratio. Therefore, the uncertainty factor associated with the Pu-241 to Pu-239/240 ratio was also represented by a triangular distribution, with a best estimate of 1.0 and upper and lower bounds of 1.4 and 0.45, respectively.

1985-1989

During this period, both Pu-239/240 and Am-241 were monitored by alpha spectral analysis. Only Pu-241 release estimates were derived from the source terms of Pu-239/240. As explained above, this calculation was based on the ratio of Pu-241 to Pu-239/240. The uncertainty factor of this ratio was developed in the section above, and was represented by a triangular distribution, with a best estimate of 1.0 and upper and lower bounds of 1.4 and 0.45, respectively.

F.2   Uncertainties Associated with Estimating Enriched and Depleted Uranium Releases Based on Nonspecific Monitoring Data

1953-1977

Routine isotopic analysis of effluent sample filters did not start until around 1973. However, reporting of long-lived gross alpha activity continued for uranium facilities until approximately 1978. Emission from uranium facilities were "radiochemically determined as U-233, U-234 and U-238" for the first time in the 1978 Rocky Flats Plant annual environmental report (Rockwell, 1979). Before that time, long-lived gross alpha activity sampled in effluents from buildings in which enriched or depleted uranium were handled was assumed to be 100 percent enriched or depleted uranium, respectively. As discussed above, an uncertainty factor represented by a uniform distribution with an upper bound of 1.2 and a lower bound of 0.8 was developed to compensate for the uncertainty introduced by this practice.

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<u>1978-1980 and 1984-1989</u>

From 1978 through 1989, with the exception of 1981-1983, reported airborne uranium emissions were based on alpha spectral measurements of uranium isotopes. Due to the specific nature of these analysis, it was assumed that there was no uncertainty associated with the identity of analytes measured during this period of time.

<u>1981-1983</u>

Uranium emissions for calendar years 1981, 1982, and 1983 were reported only as total uranium emissions—separate depleted and enriched results were not reported. Based on the historical fractions of airborne depleted and enriched uranium emissions reported in 1978-1980 and 1984-1989, release estimates of enriched and depleted uranium were calculated from the total uranium emission. However, because of the fluctuation of the relative importance of enriched and depleted uranium in the total uranium emission, uncertainty was introduced in this translation. Based on the values shown in Figure F-1, the correction factor for enriched uranium was assumed to have a triangular distribution with a best estimate of 1 and upper and lower bounds of 1.95 and 0.42, respectively. Similarly, the correction factor for depleted uranium was assumed to have a triangular distribution with a best estimate of 1 and upper and lower bounds of 1.38 and 0.37, respectively.

Figure F-1 Uranium Emission of 1978-1980 and 1984-1989

APPENDIX G

**OVERALL UNCERTAINTIES ASSOCIATED WITH RELEASE
OF ESTIMATES OF CONTAMINANTS OF CONCERN**

APPENDIX G

OVERALL UNCERTAINTIES ASSOCIATED WITH RELEASE
OF ESTIMATES OF CONTAMINANTS OF CONCERN

As discussed in Section 4, there are many sources of uncertainty that may have an impact on the release estimates of contaminants of concern. Uncertainty factors that can be used to represent uncertainties associated with monitoring programs and the identity of contaminants measured were developed in Section 4 and Appendix F, respectively. In this appendix, the identified uncertainty factors of a particular contaminant are combined where they overlap in time by a statistical technique called Monte Carlo simulation to produce an overall uncertainty factor.

Annual emission estimates were calculated by multiplying the source terms developed in Sections 2.8 and 3.1 by the appropriate overall uncertainty factor distribution. This produced a probability distribution for the annual emissions. It is believed that the use of these emission probability distributions will bound the actual emissions in a year.

In the following section, the calculation of overall uncertainty factors for plutonium and americium isotopes, uranium isotopes, tritium and beryllium are discussed.

G.1 Overall Uncertainties of Release Estimates of Pu-239/240, Am-241 and Pu-241

As discussed in Sections 2.2 and 4, five potential sources of uncertainty were identified in the development of release estimates of plutonium and americium isotopes:

      (1)   Sampling flow rate,
      (2)   Effluent flow rate,
      (3)   Analytical procedure,
      (4)   Identity of long-lived alpha emitters, and
      (5)   Apportioning of plutonium alpha into Pu-239/240 and Am-241.

Because monitoring data available for the calculation of release estimates of plutonium and americium isotopes from 1953 to 1973, from 1974 to 1984, and from 1985 to 1989 are different, different uncertainty factors were developed to account for various sources of uncertainty. They are listed in Tables G-1 through G-3. The overall uncertainty factor of each isotope and time period was calculated by combining all relevant uncertainty factors by Monte Carlo simulation. The results of the simulations are also provided in Tables G-1 through G-3. Since overall uncertainty factors calculated for a specific isotope do not change significantly over the three time periods, it was assumed that they can be represented by a single factor. The overall uncertainty factors of Pu-239/240, Am-241, and Pu-241 that were used to characterize plant emission from 1953 to 1989 are summarized below:

Overall Uncertainty Factor

Pu-239/240   Lognormal Distribution (GM=1.3, GSD=1.6)
Am-241   Lognormal Distribution (GM=1.4, GSD=1.6)
Pu-241   Lognormal Distribution (GM=1.2, GSD=1.6)

TABLE G-1

**INDIVIDUAL AND OVERALL UNCERTAINTY FACTORS ASSOCIATED WITH RELEASE
ESTIMATES OF PU-239/240, AM-241 AND PU-241, 1953-1973**

Nuclides	Source of Uncertainty					
	Partitioning of Total Alpha into Specific Isotopes		Sampling Flow Rate	Effluent Flow Rate	Analytical Procedure	Overall Method
Pu-239/240	Uniform 0.8 - 1.2	Triangular 0.93 - 1 - 1.11	Normal Mean = 0.92 SD = 0.11	Triangular 0.5 - 1 - 2	Normal Mean = 1.1 SD = 0.3	Lognormal GM = 1.3 GSD = 1.6
Am-241	Uniform 0.8 - 1.2	Triangular 0.5 - 1 - 1.33	Normal Mean = 0.92 SD = 0.11	Triangular 0.5 - 1 - 2	Normal Mean = 1.1 SD = 0.3	Lognormal GM = 1.2 GSD = 1.6
Pu-241	Uniform 0.8 - 1.2	Normal Mean = 0.96 SD = 0.2	Normal Mean = 0.92 SD = 0.11	Triangular 0.5 - 1 - 2	Normal Mean = 1.1 SD = 0.3	Lognormal GM = 1.2 GSD = 1.6

Note:
SD   =   Standard deviation
GM   =   Geometric mean
GSD   =   Geometric standard deviation

TABLE G-2

**INDIVIDUAL AND OVERALL UNCERTAINTY FACTORS ASSOCIATED WITH RELEASE
ESTIMATES OF PU-239/240, AM-241 AND PU-241, 1974-1984**

Nuclides	Source of Uncertainty				
	Partitioning of Total Alpha into Specific Isotopes	Sampling Flow Rate	Effluent Flow Rate	Analytical Procedure	Overall Method
Pu-239/240	NA	Normal Mean = 0.92 SD = 0.11	Triangular 0.5 - 1 - 2	Normal Mean = 1.07 SD = 0.14	Lognormal GM = 1.3 GSD = 1.4
Am-241	Triangular 0.45 - 1 - 1.4	Normal Mean = 0.92 SD = 0.11	Triangular 0.5 - 1 - 2	Normal Mean = 1.07 SD = 0.14	Lognormal GM = 1.2 GSD = 1.5
Pu-241	Triangular 0.45 - 1 - 1.4	Normal Mean = 0.92 SD = 0.11	Triangular 0.5 - 1 - 2	Normal Mean = 1.07 SD = 0.14	Lognormal GM = 1.2 GSD = 1.5

TABLE G-3

**INDIVIDUAL AND OVERALL UNCERTAINTY FACTORS ASSOCIATED WITH RELEASE
ESTIMATES OF PU-239/240, AM-241 AND PU-241, 1985-1989**

Nuclides	Source of Uncertainty				
	Partitioning of Total Alpha into Specific Isotopes	Sampling Flow Rate	Effluent Flow Rate	Analytical Procedure	Overall Method
Pu-239/240	NA	Normal Mean = 0.92 SD = 0.11	Triangular 0.5 - 1 - 2	Normal Mean = 1.07 SD = 0.14	Lognormal GM = 1.3 GSD = 1.4
Am-241	NA	Normal Mean = 0.92 SD = 0.11	Triangular 0.5 - 1 - 2	Normal Mean = 1.07 SD = 0.14	Lognormal GM = 1.4 GSD = 1.5
Pu-241	Triangular 0.45 - 1 - 1.4	Normal Mean = 0.92 SD = 0.11	Triangular 0.5 - 1 - 2	Normal Mean = 1.07 SD = 0.14	Lognormal GM = 1.2 GSD = 1.5

Note:
SD   =   Standard deviation
GM   =   Geometric mean
GSD   =   Geometric standard deviation
NA   =   Not Applicable

G.2   Overall Uncertainties Associated with Release Estimates of Enriched and Depleted Uranium

As discussed in Sections 2.2 and 4, four potential sources of uncertainty were identified in the source term development of uranium isotopes:

  (1)   Sampling flow rate,
  (2)   Effluent flow rate,
  (3)   Analytical procedure, and
  (4)   Identity of the long-lived alpha emitter.

To establish the overall uncertainty in the emission estimates of enriched and depleted uranium, the uncertainties described above were combined where they overlap in time. The uncertainty factors used in Monte Carlo modeling and the results obtained are listed in Tables G-4 through G-6. Since overall uncertainty factors calculated for a specific isotope do not change significantly over time, they can be represented by a single factor. The overall uncertainty factors of enriched and depleted uranium over 1953-1989 are as follows:

Overall Uncertainty Factor

Enriched uranium   Lognormal Distribution (GM=1.3, GSD=1.6)
Depleted uranium   Lognormal Distribution (GM=1.3, GSD=1.6)

G.3   Overall Uncertainties Associated with Release Estimates of Tritium

1953-1973

As described in Section 2.2.1.1, sampling for tritium in airborne effluents was not routinely conducted until 1974. Therefore, emission estimates of tritium prior to 1974 are not based on measurements. According to Section 2.5.5, annual tritium emissions for this period will be treated as a uniform distribution with the identified lower and upper bounds of 140 and 390 for the period of 1968 through 1973, and 1 and 800 for the period of 1953 through 1967.

1974-1989

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The approach used to evaluate uncertainties associated with the monitoring of airborne plutonium and uranium can also be applied to the monitoring of airborne tritium. As discussed in Section 2.2, four potential sources of uncertainty were identified in the development of release estimates for tritium:

    (1)   Sampling flow rate,
    (2)   Effluent flow rate,
    (3)   Analytical procedure, and
    (4)   Collection efficiency.

TABLE G-4

INDIVIDUAL AND OVERALL UNCERTAINTY FACTORS ASSOCIATED WITH RELEASE ESTIMATES OF ENRICHED AND DEPLETED URANIUM, 1953-1977

Nuclides	Source of Uncertainty				
	Partitioning of Total Alpha into Specific Isotopes	Sampling Flow Rate	Effluent Flow Rate	Analytical Procedure	Overall Method
Enriched Uranium	Uniform 0.8 - 1.2	Normal Mean = 0.92 SD = 0.11	Triangular 0.5 - 1 - 2	Normal Mean = 1.1 SD = 0.3	Lognormal GM = 1.3 GSD = 1.6
Depleted Uranium	Uniform 0.8 - 1.2	Normal Mean = 0.92 SD = 0.11	Triangular 0.5 - 1 - 2	Normal Mean = 1.1 SD = 0.3	Lognormal GM = 1.3 GSD = 1.6

TABLE G-5

INDIVIDUAL AND OVERALL UNCERTAINTY FACTORS ASSOCIATED WITH RELEASE ESTIMATES OF ENRICHED AND DEPLETED URANIUM, 1978-1980 and 1984-1989

Nuclides	Source of Uncertainty				
	Partitioning of Total Alpha into Specific Isotopes	Sampling Flow Rate	Effluent Flow Rate	Analytical Procedure	Overall Method
Enriched Uranium	NA	Normal Mean = 0.92 SD = 0.11	Triangular 0.5 - 1 - 2	Normal Mean = 1.06 SD = 0.2	Lognormal GM = 1.3 GSD = 1.4
Depleted Uranium	NA	Normal Mean = 0.92 SD = 0.11	Triangular 0.5 - 1 - 2	Normal Mean = 1.06 SD = 0.2	Lognormal GM = 1.3 GSD = 1.4

TABLE G-6

INDIVIDUAL AND OVERALL UNCERTAINTY FACTORS ASSOCIATED WITH RELEASE ESTIMATES OF ENRICHED AND DEPLETED URANIUM, 1981-1983

Nuclides	Source of Uncertainty				
	Partitioning of Total Alpha into Specific Isotopes	Sampling Flow Rate	Effluent Flow Rate	Analytical Procedure	Overall Method
Enriched Uranium	Triangular 0.42 - 1 - 1.95	Normal Mean = 0.92 SD = 0.11	Triangular 0.5 - 1 - 2	Normal Mean = 1.06 SD = 0.2	Lognormal GM = 1.4 GSD = 1.6
Depleted Uranium	Triangular 0.37 - 1 - 1.38	Normal Mean = 0.92 SD = 0.11	Triangular 0.5 - 1 - 2	Normal Mean = 1.06 SD = 0.2	Lognormal GM = 1.1 GSD = 1.6

Note:
SD = Standard deviation
GM = Geometric mean
GSD = Geometric standard deviation
NA = Not Applicable

As described in Section 2.2.1.2, sampling flow rates have historically been set at approximately 50 cm^3 min^{-1} for tritium samplers. However, the actual average sampling flow rate is likely to be larger than this value. It is because water was used as the trapping medium for tritium; as sampled air was bubbled through the medium, a portion of water was lost to evaporation. This resulted in a drop in resistance to air flow and an increase of sampling flow rate. Since data to characterize the variability of the actual sample flow rates were not located, it is assumed that the correction factor of tritium sampling has a triangular distribution with a best estimate of 1.2 and upper and lower bounds of 1.5 and 0.9, respectively.

The uncertainty associated with the estimation of effluent flow rate has been discussed and quantified in Section 4. It was used in the determination of overall uncertainty associated with the tritium monitoring data.

As discussed before, beginning in 1974, the plant began reporting the average relative error associated with tritium analysis in the annual environmental reports. The reported annual errors are summarized in Table 4-1. Investigators were unable to clearly establish whether the plant corrected the reported release estimates for these errors; however, it is believed that they did not. Therefore, an uncertainty factor is developed to compensate for this potential source of error. It is assumed to have a normal distribution with a mean of 0.95 and a standard deviation of 0.08 as indicated in Table G-7.

TABLE G-7

**INDIVIDUAL AND OVERALL UNCERTAINTY FACTORS ASSOCIATED
WITH RELEASE ESTIMATES OF TRITIUM, 1974-1989**

Source of Uncertainty				
Sampling Flow Rate	**Collection Efficiency**	**Effluent Flow Rate**	**Analytical Procedure**	**Overall Methods**
Triangular 0.9 - 1.2 - 1.5	Triangular 1.3 - 3.0 - 4.8	Triangular 0.5 - 1 - 2	Normal Mean = 0.95 SD = 0.08	Lognormal GM = 2.6 GSD = 1.5

Another source of uncertainty that is unique to tritium is the collection efficiency of the analyte. Based on the result of a special tritium study performed by the plant in 1978 (Section 2.2.1.1), it is estimated that the collection efficiency of tritium is 48 ± 27 percent, meaning actual emission to the air ranged from 1.3 to 4.8 times those reported. Using this information, the correction factor for

collection efficiency of tritium was assumed to have a triangular distribution with a best estimate of 3.0 (average of 1.3 and 4.8) and an upper and lower bounds of 1.3 and 4.8, respectively.

Monte Carlo simulation was used to combine the four uncertainty factors described above to estimate the overall uncertainty in the tritium emission data after 1973. The uncertainty factors used in Monte Carlo modeling and the results obtained are listed in Table G-7. The overall uncertainty associated with the determined release estimates between 1974 and 1989 can be represented by a lognormal distribution with a GM of 2.6 and a GSD of 1.5.

G.4   Overall Uncertainties Associated with Release Estimates of Beryllium

As described in Section 3.1.2, a relatively complete record of the annual beryllium emissions was compiled from sample data logbooks for 1960 through 1970 and annual beryllium releases reported in the Annual Environmental Monitoring Reports for 1971 through 1989. No sampling data were located for the period prior to 1960. Based on document review and personnel interviews, it is believed that beryllium was not used in the manufacturing process until 1958 and in the absence of any data, it is assumed that the emissions from 1958 and 1959 were approximately the same as those reported in 1960.

Upon reviewing the beryllium monitoring program used at Rocky Flats, three potential sources of uncertainty were identified in the development of release estimates of beryllium:

> (1)   Sampling flow rate,
> (2)   Effluent flow rate, and
> (3)   Analytical procedure.

The sources of uncertainty related to the collection of samples and the quantification of effluent flow rates discussed for plutonium and uranium measurements also apply to beryllium measurements. Therefore, uncertainty factors associated with sample flow rate and effluent flow rate developed earlier can also be used in this section.

A brief description of different analytical methods used at Rocky Flats to measure beryllium throughout the history of the plant is provided in Section 3.1.1. Listed chronologically, they are: emission spectroscopy with photographic plates as detector, emission spectroscopy with photomultiplier as detector, flame atomic absorption spectroscopy and nonflame atomic absorption spectroscopy. It is believed that detection limit and precision of a new method is generally better than the method it replaced. Annual average relative errors of the latest method, nonflame atomic absorption spectroscopy, were reported from 1974 through 1989 and are reproduced in Table 4-1. Based on the information provided in this table, it is estimated that for the period from 1971 through 1989, the potential error associated with beryllium analysis can be represented by an uncertainty

factor with a normal distribution which has a mean of 1.21 and a standard deviation of 0.49. However, this would lead to an upper bound of approximately 2.7 (mean + 3 standard deviations) and a lower bound of approximately -0.3 (mean - 3 standard deviations). As it is physically meaningless to have a correction factor with a negative value, a surrogate uncertainty factor was created to represent this source of uncertainty. The new uncertainty factor is assumed to have a triangular distribution with a best estimate of 1.2 and an upper and lower bounds of 2.7 and 0, respectively.

Although information about the accuracy of the methods used in the earlier years is not available, it is reasonable to assume that they are larger than those reported after 1973. For the purpose of this evaluation, it is assumed that variability of relative error of earlier methods is about twice as large as those reported for nonflame atomic absorption spectroscopy. In other words, the correction factor would have a normal distribution which has a mean of 1.21 and a standard deviation of 1.0. However, this would give an upper bound of approximately 4.2 (mean + 3 standard deviations) and a lower bound of approximately -1.8 (mean - 3 standard deviations). For the same reason given above, a surrogate uncertainty factor was created to represent this source of uncertainty. The new uncertainty factor is assumed to have a triangular distribution with a best estimate of 1.2 and an upper and lower bounds of 4.2 and 0, respectively.

Like before, Monte Carlo simulation was used to combine the uncertainty factors developed to estimate the overall uncertainties in the beryllium emission data from 1960 to 1970 and from 1971 to 1989. The uncertainty factors used in Monte Carlo modeling and the results obtained are listed in Table G-8. The overall uncertainty factors developed for beryllium emissions are as follows:

<div align="center">Overall Uncertainty Factor</div>

1960-1970	Lognormal Distribution (GM=1.9, GSD=2)
1971-1989	Lognormal Distribution (GM=1.4, GSD=1.9)

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TABLE G-8

**INDIVIDUAL AND OVERALL UNCERTAINTY FACTORS
ASSOCIATED WITH RELEASE ESTIMATES OF BERYLLIUM**

Period	Source of Uncertainty			
	Sampling Flow Rate	Effluent Flow Rate	Analytical Procedure	Overall Method
1960-1970	Normal Mean = 0.92 SD = 0.11	Triangular 0.5 - 1 - 2	Triangular 0 - 1.2 - 4.2	Lognormal GM = 1.9 GSD = 2.0
1971-1989	Normal Mean = 0.92 SD = 0.11	Triangular 0.5 - 1 - 2	Triangular 0 - 1.2 - 2.7	Lognormal GM = 1.4 GSD = 1.9

Note:
SD   =   Standard deviation
GM   =   Geometric mean
GSD   =   Geometric standard deviation

APPENDIX H

**CALCULATION OF OVERALL UNCERTAINTY FACTORS
BY MONTE CARLO SIMULATION**

APPENDIX H

CALCULATION OF OVERALL UNCERTAINTY FACTORS
BY MONTE CARLO SIMULATION

Potential systematic errors and degrees of biases in the estimation of annual release rates of radionuclides and beryllium are identified in Section 4.0 of this report. Because of the uncertainties associated with the identified parameters, they are not defined in terms of a single, discrete number, but instead in terms of a probability distribution of values that we are confident includes the true but unknown value of a particular parameter. When using inputs that are described in terms of probability distributions to perform calculations, there is more than one possible answer, and an equation must be solved many times using discrete input values that are sampled from the probability distributions defined for each of the inputs. The answer that this calculation process produces is itself a probability distribution. One method that is commonly used to perform this type of calculation is known as Monte Carlo simulation. It permits the propagation of the errors identified throughout the analytic process so that they are accurately reflected in the result. The purpose of this appendix is to describe Monte Carlo simulation and its application in calculating the overall uncertainty associated with release estimates of a contaminant.

Monte Carlo Simulation

A simple equation like the one shown below can be used to illustrate how Monte Carlo simulation may be used to propagate uncertainties in the input parameters of an equation.

$$A = B * C$$

When the exact values of parameters B and C are known, A can be calculated by simply multiplying B by C. This is also known as a point estimate or deterministic calculation, because it produces a single value of A. However, when there are uncertainties associated with parameters B and C, A cannot be determined by multiplying B by C. Figure H-1 shows how Monte Carlo simulation can be used to propagate the uncertainties in B and C through the equation and produce a probability distribution of A. The process can be divided into three steps. First, many values of each parameter are selected according to the probability distribution of the parameter. Second, the selected values of parameter B are randomly paired with the selected values of parameter C. Lastly, the paired values are multiplied together consistent with the equation to produce an estimate of A. For example, if 500 pairs of parameter B and C are selected, Monte Carlo simulation would produce 500 estimates of A. These estimates can be arranged numerically to provide a probability distribution of A as shown in Figure H-1.

Figure H-1   Schematic Representation of Monte Carlo Simulation

Calculation of Overall Uncertainty Factors Associated with Contaminant Release Estimates

As described in Section 4 of this task report, potential systematic errors associated with the sampling and analytical processes, estimation of effluent flow rates and lack of information about the identity of the analyte are identified in the estimation of annual release rates of radionuclides and beryllium. The overall uncertainty factors of radionuclides and beryllium were determined by combining the appropriate uncertainty factors with Monte Carlo simulation (Appendix G). Calculation of the overall uncertainty factor associated with release estimates of Pu-239/240 between 1953 and 1973 is used in this appendix to illustrate this process.

As described in Appendix G (Table G-1), there are five sources of uncertainty in the determination of release estimates of Pu-239/240 between 1953 and 1973:

- Assignment of total long-lived alpha to plutonium and americium isotopes, U_a;
- Partition of plutonium and americium isotopes into specific isotopes, U_{Pu};
- Measurement of sampling flow rate, U_s;
- Estimation of effluent flow rate, U_e; and
- Measurement of total long-lived alpha particles, U_m.

If each of these five sources of uncertainty can be represented by an uncertainty factor, the overall uncertainty of the Pu-239/240 release estimates can be determined by the following equation:

$$Overall \ \ Uncertainty \ \ Factor \ \ = \ \ U_a \ * \ U_{Pu} \ * \ U_e \ * \ U_m \ / \ U_s$$

A commercial software package called Crystal Ball® (Decisioneering, 1993) was used to perform the simulation. Probability distributions of the five uncertainty factors defined in Table G-1 of Appendix G were used as inputs to the equation and are presented in Figure H-2. In this example, 3000 runs were performed by the program to generate a probability distribution of the overall uncertainty factor (Figure H-3). Since the distribution appears to be lognormally distributed, it is best characterized by its geometric mean (GM) and geometric standard deviation (GSD).

In order to facilitate the determination of GM and GSD of the overall uncertainty factor, the probability distribution of the overall uncertainty factor in logarithmic scale was also generated (Figure H-4).

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Insert Figure H-2

Insert Figure H-2

Insert Figure H-3

Insert Figure H-4

GM and GSD of the overall uncertainty factor are related to the mean and standard deviation of the overall uncertainty factor in logarithmic scale by the following two equations:

$$GM \ = \ \exp[mean(L)]$$

$$GSD \ = \ \exp[SD(L)]$$

Where:

GM   =   Geometric mean of the probability distribution of the overall uncertainty factor,

GSD =   Geometric standard deviation of the probability distribution of the overall uncertainty factor,

mean(L) =   Mean of the probability distribution of the overall uncertainty factor in logarithmic scale,

SD(L) =   Standard deviation of the probability distribution of the overall uncertainty factor in logarithmic scale.

GM and GSD of the overall uncertainty factor associated with release estimates of Pu-239/240 between 1953 and 1973 are calculated based on the Monte Carlo simulation output and the equations shown above:

GM   =   exp[0.27]

       =   1.3

GSD   =   exp[0.44]

       =   1.6

GMs and GSDs of overall uncertainty factors of other contaminants of concern were determined in a similar manner.

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REFERENCE

Decisioneering (1993). Crystal Ball, version 3.0. User Manual. Decisioneering, Inc., Denver, CO.

APPENDIX I

**COLORADO DEPARTMENT OF HEALTH DRINKING WATER DATA
SUMMARY AND STATISTICAL TESTING**

APPENDIX I

COLORADO DEPARTMENT OF HEALTH DRINKING WATER DATA
SUMMARY AND STATISTICAL TESTING

During the period of 1970 to 1989, the Colorado Department of Health (CDH) sampled drinking waters from the cities of Broomfield and Westminster. These cities draw their water from Great Western Reservoir and Standley Lake, which are potentially impacted by waterborne effluents from the Rocky Flats plant. In addition, CHD also sampled drinking waters from the cities of Arvada, Boulder, and Golden which derive their waters from reservoirs at a considerable distance from Rocky Flats and that do not receive run-off or effluents from Rocky Flats. The drinking water samples from the cities deriving their water from distant water supplies relative to Rocky Flats provide reference points for comparing the levels of radioactivity found in waters possibly impacted by the plant. These comparisons are presented in the main body of the text of this report in Section 5.5. This appendix presents details about the CDH data set in terms of the frequency of detection and the maximum values in Tables I-1 through I-4 and the annual average radionuclide concentrations in Table I-5 through I-8. As indicated in Table I-1 through I-4, a large number of samples were below detection limits. In calculating the annual average values, one-half the applicable detection limit was used whenever non-detect results were reported. As described in the main text of the report, the average is not necessarily the best statistical descriptor to use to characterize the data since the data are not normally (or even log-normally) distributed because of the large number of non-detects. The average, in this case, would tend to provide an over-estimate of the central tendency of the data set. For this reason, other methods of examining the data were explored to determine if they might yield additional information.

Statistical Testing

The objective of the analysis is to construct a statistical test that can be used to show if the radionuclide concentrations found in the drinking water from Broomfield and Westminster are the same as those from other cities (Arvada, Boulder, and Golden) which do not derive their water from reservoirs that were likely impacted by the Rocky Flats Plant.

The available water sample data cover an approximately 10-year period, with over 50 percent of the sample results below detection limits. The detection limits varied over time.

There are no known "direct" statistical tests that can be used when so many non-detects are involved. However, for large sample sizes, the generalized likelihood-ratio test (Mood *et al.*, 1963; pages 440-442) can be used to give an approximate solution to this problem. Like all likelihood tests, both "point" and interval data can be directly incorporated into the analyses without resorting to using "mid-value" or any other artificial value for data that are below a level of detection. Besides being restricted to large sample sizes, the only other constraint is that the underlying distribution of the data must be assumed. However, the test can be repeated by first assuming a log-normal distribution, then a normal, Weibull, etc. A more detailed description of the underlying theory is given by Sverdrup

(1967; pages 133-140), Wald (1943) and Wilks (1938). Measurements of contaminant concentrations, as in this application, are typically well-approximated by a lognormal distribution.

The generalized likelihood-ratio test is performed by defining a null hypothesis and its alternative, computing the parameters of the assumed distribution using the maximum likelihood algorithm, and then taking the ratio, λ, of the two likelihood functions that have different assumptions concerning the equality of the parameters:

$$\lambda = \sup\ L(\mu_n\ ,\ \sigma_{1n}\ ,\ \sigma_{2n})\ /\ \sup\ L(\mu_{1d}\ ,\ \mu_{2d}\ ,\ \sigma_{1d}\ ,\ \sigma_{2d})$$

where sup means to find the largest value; L(.) is the likelihood function for the joint log-normal distributions of two data samples containing both point and interval data; μ is the true mean of the assumed normal distribution of the logarithms of the original measurements (i.e, $\mu = \text{Mean}[y]$ and $y = \text{Ln(Conc)}$); σ is the true standard deviation of the same (i.e., $\sigma = \text{Std Dev}[y]$ and $y = \text{Ln(Conc)}$). The subscripts 1 and 2 refer to the two data sets or cities from which the chemical concentrations are being compared and the subscripts "n" and "d" refer to the fact that the parameters for the numerator and may be different. In the numerator likelihood function, the mean of the two data distributions are assumed to be equal and the variances to be unequal. In the denominator likelihood function, both the mean and the standard deviations of the two data distributions are assumed to be unequal. The λ ratio will always be less than one because the numerator term has one degree of freedom less than that of the denominator term (the numerator term will always be more restrictive than the denominator, and therefore also less likely).

The exact distribution of the λ ratio is intractable (Mood *et al.*, 1963; page 440), but an approximate size-σ test can be obtained. The asymptotic distribution of the generalized likelihood-ratio reduces to that of the chi-square distribution with 1 degree of freedom for large sample sizes. If the null hypothesis is defined such that the means of the data distributions are assumed to be equal, then the null hypothesis should be rejected when the following condition holds:

$$-2Ln(\lambda) > \chi^2(1-\alpha,1)$$

when Ln(.) is the natural logarithm and $\chi^2(.)$ is the chi-square function evaluated at 1 degree of freedom and at the quantile 1-α.

The above test was performed with the radionuclide concentration data from Broomfield and Westminster compared to that of three surrounding cities (Arvada, Boulder, and Golden). A computer code was written to solve the maximum likelihood function for the joint distribution

parameters, i.e., the set $(\mu_n, \sigma_{1n}, \sigma_{2n})$ and $(\mu_{1d}, \mu_{2d}, \sigma_{1d}, \sigma_{2d})$ and then perform the above chi-square test. Solving the maximum likelihood problem for joint distributions can be very difficult. A brute force method of systematically guessing the parameters over various ranges was first used to get a rough estimate of the maximum likelihood parameters and then a Newton-Raphson iteration scheme was used to refine the solution to a high degree of accuracy.

The p-value or "size" of the test was found by setting $-2Ln(\lambda) = \chi^2(1-\alpha,1)$ and solving for the resultant α that just satisfied this condition. Summaries of the results are given in Tables I-9 and I-10. The analysis shows that over the ten year period, the mean values of the Broomfield and Westminster water concentrations of plutonium, tritium, and uranium were statistically no different than those of the other cities. The only exceptions were those for uranium concentrations compared for Boulder and Broomfield and Westminster and Arvada.

Discussion

The likelihood-ratio test comparing means is reasonably reliable, provided the sample sized are not too small. Some of the sample sizes under consideration are rather small, particularly those for plutonium-238, and this problem is exacerbated by the fact that the non-detects contain less information than the detected concentrations. A more serious problem for the tests involving the smaller sample sized is lack of power to detect (that is, declare statistically significant) a difference between means when the difference between the true means is large enough to be considered important. Given that there are no better alternatives to the likelihood-ratio test for these data, the best that can be done is to exercise caution in interpreting the results of this analysis and emphasize that these analyses are not definitive given the limitations of the data.

Caution should also be exercised in the interpretation of the two p-values that are marginally less than 0.05 (i.e., uranium for Broomfield—Boulder and Westminster—Arvada). The probability of getting at least one p-value less than 0.05 when performing 12 independent tests, if there are no true differences between the 12 pairs of means, is 0.46. These tests are not completely independent, but clearly the chance of one "false positive" in the set is substantial. In addition, any imprecision in the tests due to small sample sizes and deviations from the distributional assumption is likely to result in p-values that are too small rather than too large. Therefore, the isolated significant differences provide, at most, weakly suggestive evidence of difference between the true means.

The bottom-line is that, as was concluded from the qualitative evaluation of the data in the main text of the report, there is little or no evidence based on the drinking water sampling conducted by CDH that waters from Broomfield and Westminster were any different with respect to the long-term (1970-1989) concentrations of the radionuclides sampled from those of Arvada, Boulder, or Golden.

TABLE I-1

**DETECTION OF FREQUENCY OF PU-238
IN DRINKING WATER, 1970-1989**

Location	Number of Samples	Frequency of Detection	Percent Detects	Maximum Concentration (pCi L^{-1})	Date of Maximum
Broomfield	253	33/253	13	0.16	3/19/74
Westminster	100	10/100	10	0.62	10/25/73
Arvada	28	5/28	18	0.58	2/14/75
Golden	30	3/30	10	0.19	2/14/75
Boulder	22	4/22	18	0.21	4/7/75

Source: Calculated from data reported by the Colorado Department of Health in Monthly Environmental Surveillance Reports.

TABLE I-2

**DETECTION FREQUENCY OF PU-239/240
IN DRINKING WATER, 1970-1989**

Location	Number of Samples	Frequency of Detection	Percent Detects	Maximum Concentration (pCi L^{-1})	Date of Maximum
Broomfield	343	84/343	24	4.52	4/26/73
Westminster	150	47/150	31	0.75	7/5/72
Arvada	47	8/47	17	1.15	3/1/75
Golden	51	10/51	20	0.09	2/14/75
Boulder	49	6/49	12	0.48	4/7/75

Source: Calculated from data reported by the Colorado Department of Health in Monthly Environmental Surveillance Reports.

TABLE I-3

**DETECTION FREQUENCY OF NATURAL URANIUM
IN DRINKING WATER, 1970-1989**

Location	Number of Samples	Frequency of Detection	Percent Detects	Maximum Concentration (pCi L^{-1})	Date of Maximum
Broomfield	356	188/356	53	346.4	12/13/76
Westminster	280	94/280	34	29.15	7/10/74
Arvada	91	31/91	34	35	2/1/80
Golden	45	14/45	31	13.89	3/29/74
Boulder	42	8/42	19	15.46	4/16/73

Source: Calculated from data reported by the Colorado Department of Health in Monthly Environmental Surveillance Reports.

TABLE I-4

**DETECTION FREQUENCY OF TRITIUM
IN DRINKING WATER, 1970-1989**

Location	Number of Samples	Frequency of Detection	Percent Detects	Maximum Concentration (pCi L^{-1})	Date of Maximum
Broomfield	854	345/854	41	23293	6/1/73
Westminster	280	131/280	47	3450	6/2/75
Arvada	124	29/124	23	1291	11/13/91
Golden	59	16/59	27	1776	10/6/71
Boulder	65	12/65	19	1101	9/26/72

Source: Calculated from data reported by the Colorado Department of Health in Monthly Environmental Surveillance Reports.

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TABLE I-5

ANNUAL AVERAGE PU-238 CONCENTRATIONS
IN DRINKING WATER (pCi L^{-1}), 1970-1989

Year	Arvada	Boulder	Golden	Broomfield	Westminster
1970	0.015	no samples	0.015	0.015	0.015
1971	0.015	no samples	no samples	0.015	0.015
1972	0.015	0.015	0.015	0.0155	0.015
1973	0.015	0.053	0.015	0.023	0.051
1974	0.015	0.015	0.0125	0.039	0.021
1975	0.32	0.1125	0.0775	0.017	0.017
1976	no samples	no samples	no samples	0.013	0.015
1977	no samples	no samples	no samples	0.014	no samples
1978	0.015	0.015	0.015	0.015	0.015
1979	0.015	no samples	0.015	0.018	0.015
1980	0.015	0.01375	0.015	0.015	0.015
1981	0.015	0.01375	0.01375	0.015	0.015
1982	0.015	0.015	0.015	0.018	0.02
1983	0.015	no samples	0.015	0.015	0.015
1984-1989	No Analyses for Pu-238				

Source: Calculated from data reported by the Colorado Department of Health in Monthly Environmental Surveillance
   Reports.

TABLE I-6

**ANNUAL AVERAGE PU-239/240 CONCENTRATIONS
IN DRINKING WATER (pCi L^{-1}), 1970-1989**

Year	Arvada	Boulder	Golden	Broomfield	Westminster
1970	0.015	no samples	0.02	0.027	0.11
1971	0.01	no samples	no samples	0.022	0.014
1972	0.01	0.025	0.027	0.073	0.07
1973	0.013	0.01	0.013	0.174	0.055
1974	0.01	0.01	0.01	0.063	0.019
1975	0.43	0.245	0.045	0.016	0.018
1976	no samples	no samples	no samples	0.015	0.01
1977	no samples	no samples	no samples	0.018	no samples
1978	0.03	0.01	0.03	0.01	0.01
1979	0.01	no samples	0.01	0.084	0.015
1980	0.01	0.01	0.01	0.01	0.01
1981	0.01	0.01	0.011	0.01	0.011
1982	0.01	0.0175	0.015	0.011	0.17
1983	0.0275	0.041	0.01	0.021	0.029
1984	0.46	0.256	0.0275	0.0275	0.073
1985	0.35	0.03	0.02	0.017	0.02
1986	0.0075	0.014	0.005	0.0055	0.006
1987	0.004	0.004	0.004	0.006	0.006
1988	0.002	0.0013	0.003	0.0023	0.0063
1989	no samples	no samples	no samples	0.0004	0.0015

Source: Calculated from data reported by the Colorado Department of Health in Monthly Environmental Surveillance
Reports.

TABLE I-7

**ANNUAL AVERAGE NATURAL URANIUM CONCENTRATIONS
IN DRINKING WATER (pCi L^{-1}), 1970-1989**

Year	Arvada	Boulder	Golden	Broomfield	Westminster
1970	no samples	no samples	no samples	no samples	no samples
1971	no samples	no samples	no samples	3.52	3.78
1972	6.41	15	1.69	3.29	4.53
1973	5.32	8.48	5.64	5.48	5.84
1974	3.86	6.53	11.68	5.19	6.15
1975	1.5	1.5	1.5	4.52	3.6
1976	no samples	no samples	no samples	16.74	0.852
1977	no samples	no samples	no samples	1	1.5
1978	3.52	1.5	1.21	1.29	0.99
1979	0.69	0.35	0.69	0.85	0.69
1980	14.5	1.5	1.5	2.21	1.75
1981	4.64	1.5	1.5	2.86	1.72
1982	2.53	2.375	2.25	2.925	2.01
1983	1.52	1.8	1.375	1.125	1.27
1984	1.27	1	1	1.23	1.18
1985	1.12	1	1.3	1	1
1986	1	1	1	1.18	1.05
1987	1	1	1.6	1.75	1
1988	1	1	2.1	1	1
1989	no samples	no samples	no samples	1	1

Source: Calculated from data reported by the Colorado Department of Health in Monthly Environmental Surveillance Reports.

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TABLE I-8

**ANNUAL AVERAGE TRITIUM CONCENTRATIONS
IN DRINKING WATER (pCi L^{-1}), 1970-1989**

Year	Arvada	Boulder	Golden	Broomfield	Westminster
1970	no samples	no samples	no samples	1051.8	1151
1971	982.5	735	1776	836	698
1972	no samples	533.7	650	995.4	892.75
1973	454.67	250	741.25	8555.2	678.3
1974	381.33	469	250	5432.1	648.2
1975	250	443	405.5	1899.6	724.1
1976	317	329.5	351	754	540.8
1977	495.5	250	330	446	370.04
1978	670.5	524	307.5	362.1	352.1
1979	280.25	212.5	212.5	302.8	281
1980	258	175	329.7	282.5	252.6
1981	412.3	262	332.3	439	275.4
1982	329	355.25	463.25	295.8	330.48
1983	218.2	231.6	175	265.3	284.4
1984	175	175	175	223.5	235.6
1985	175	175	175	208.9	209.9
1986	189.6	175	235.5	188.9	190.2
1987	233.9	175	175	194	202.24
1988	175	175	175	187	187.13
1989	no samples	no samples	no samples	102.1	100

Source: Calculated from data reported by the Colorado Department of Health in Monthly Environmental Surveillance
   Reports.

TABLE I-9

**RESULTS OF PAIRED STATISTICAL COMPARISON TEST OF
RADIOACTIVITY CONCENTRATIONS IN BROOMFIELD DRINKING WATER WITH DRINKING
WATER FROM ARVADA, BOULDER, AND GOLDEN**

City Paired with Broomfield	Resulting p-Value with [Data Set Size]			
	^{238}Pu	^{239}Pu	Tritium	Uranium
Broomfield Sample Size	[253]	[343]	[854]	[356]
Arvada	0.202 [28]	0.683 [47]	0.878 [124]	0.211 [91]
Boulder	0.541 [22]	0.156 [49]	0.393 [65]	0.023* [42]
Golden	0.139 [30]	0.402 [51]	0.849 [59]	0.522 [45]
Westminster	0.146 [100]	0.643 [197]	0.605 [418]	0.124 [279]

*   You must reject the null hypotheses that the means of the log-normal distributions are the same. Test
         also rejected the null hypotheses for normal and Weibull distributions.

NOTE: Statistical test of the hypotheses that the concentrations of plutonium, tritium, and uranium in the
         water supply from Broomfield are the same as that from the cities of Arvada, Boulder, and Golden.
         The resultant p-value of the generalized likelihood-ratio test is given. The null hypothesis assumes
         that the statistical distribution of the data is log-normal and that the mean of the assumed normal
         distribution of the logarithms of the original measurements for radionuclide concentrations in
         Broomfield drinking water and that of any other city is the same but unknown and that the variance of
         the same are different and unknown. The alternative hypothesis assumes that the means of the
         distributions are different. The null hypotheses is accepted for any p-value greater than 0.05.

TABLE I-10

**RESULTS OF PAIRED STATISTICAL COMPARISON TEST OF
RADIOACTIVITY CONCENTRATIONS IN WESTMINSTER DRINKING WATER
WITH DRINKING WATER FROM ARVADA, BOULDER, AND GOLDEN**

City Paired with Westminster	Resulting p-value with [Data Set Size]			
	²³⁸Pu	²³⁹Pu	Tritium	Uranium
Westminster Sample Size	[100]	[197]	[418]	[279]
Arvada	0.700 [28]	0.853 [47]	0.569 [124]	0.036* [91]
Boulder	0.817 [22]	0.264 [49]	0.250 [65]	0.116 [42]
Golden	0.568 [30]	0.290 [51]	0.939 [59]	0.919 [45]

*   You must reject the null hypotheses that the means of the log-normal distributions are the same. Test also rejected the null hypotheses for normal and Weibull distributions.

NOTE: Statistical test of the hypotheses that the concentrations of plutonium, tritium, and uranium in the water supply from Westminster are the same as that from the cities of Arvada, Boulder, and Golden. The resultant p-value of the generalized likelihood-ratio test is given. The null hypothesis assumes that the statistical distribution of the data is log-normal and that the mean of the assumed normal distribution of the logarithms of the original measurements for radionuclide concentrations in Westminster drinking water and that of any other city is the same but unknown and that the variance of the same are different and unknown. The alternative hypothesis assumes that the means of the distributions are different. The null hypotheses is accepted for any p-value greater than 0.05.

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APPENDIX J

**RESOLUTION OF COMMENTS
ON THE DRAFT TASK 5 REPORT**

METRIC FRACTIONS

Multiple	Decimal Equivalent	Prefix	Symbol
10^6	1,000,000	mega-	M
10^3	1,000	kilo-	k
10^2	100	hecto-	h
10	10	deka-	da
10^{-1}	0.1	deci-	d
10^{-2}	0.01	centi-	c
10^{-3}	0.001	milli-	m
10^{-6}	0.000001	micro-	µ
10^{-9}	0.000000001	nano-	n
10^{-12}	0.000000000001	pico-	p
10^{-15}	0.000000000000001	femto-	f
10^{-18}	0.000000000000000001	atto-	a

METRIC CONVERSION TABLE

Multiply	By	Equals	Multiply	By	Equals
in.	2.54	cm	cm	0.394	in.
ft	0.305	m	m	3.28	ft
ac	0.404	ha	ha	2.47	ac
mi	1.61	km	km	0.621	mi
lb	0.4536	kg	kg	2.205	lb
liq. qt.-U.S.	0.946	l	l	1.057	liq. qt.-U.S.
ft^2	0.093	m^2	m^2	10.764	ft^2
mi^2	2.59	km^2	km^2	0.386	mi^2
ft^3	0.028	m^3	m^3	35.31	ft^3
d/m	0.450	pCi	pCi	2.22	d/m
pCi/l(water)	10^{-9}	µCi/ml(water)	µCi/ml(water)	10^9	pCi/l(water)
pCi/m^3(air)	10^{-12}	µCi/cc(air)	µCi/cc(air)	10^{12}	pCi/m^3(air

TRADITIONAL AND INTERNATIONAL SYSTEMS OF RADIOLOGICAL UNITS

(Traditional units are in parentheses.)

Quantity	Name	Symbol	Expression in Terms of Other Units
absorbed dose	Gray	Gy	J/Kg^{-1}
	(rad)	rad	10^{-2} Gy
activity	Becquerel	Bq	1 dps
	(curie)	Ci	3.7 x 12^{10} Bq
dose equivalent	Sievert	Sv	J/Kg^{-1}
	(rem)	rem	10^{-2} Sv
exposure	Coulomb per kilogram		C/Kg^{-1}
	(roentgen)	R	2.58 x 10^{-4} C/Kg^{-1}