FINAL REPORT

Estimated Exposure and Lifetime Cancer Incidence Risk from Beryllium Released to the Air from the Rocky Flats Plant

Part of Task 3: Independent Analysis of Exposure, Dose, and Health Risk to Offsite Individuals

August 1999

Submitted to the Colorado Department of Public Health and Environment, Disease Control and Environmental Epidemiology Division, Rocky Flats Health Studies in partial fulfillment of Contract No. 100APPRCODE 391

"Setting the standard in environmental health"

RAC

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Authors

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EXECUTIVE SUMMARY

The Rocky Flats Environmental Technology Site is owned by the U.S. Department of Energy (DOE) and is currently contractor-operated by Kaiser-Hill Company. For most of its history, the site was called the Rocky Flats Plant (RFP) and was operated by Dow Chemical Company as a nuclear weapons research, development, and production complex. The RFP is located about 8–10 km from the cities of Arvada, Westminster, and Broomfield, Colorado and 26 km (16 mi). northwest of downtown Denver, Colorado.

Through a 1989 Agreement in Principle between the DOE and the State of Colorado, DOE provided the State with funding and technical support for health-related studies. The purpose of the Historical Public Exposures Studies on Rocky Flats is to identify potential health effects in residents in nearby communities who may have been exposed to past toxic and radioactive releases.

This report documents risk calculations for inhalation of beryllium in air resulting from routine operational releases at the RFP. The report evaluates environmental monitoring data, discusses evidence of health effects, and describes environmental transport modeling. Estimates of airborne concentrations with uncertainty are provided along with lifetime carcinogenic incidence risk resulting from inhalation of beryllium for generic receptor scenarios.

Source Term. The source term for beryllium was derived from work done in Phase I of the Historical Public Exposures Studies. The monthly and annual average beryllium concentrations for each stack were calculated from the building effluent data. Releases that occurred during three fires were also included in the source term estimates. Uncertainty associated with the beryllium source term estimates was characterized using the same approach applied to Phase I plutonium and uranium sample measurements and release estimates. Uncertainties in exhaust and sample flow rate estimates, and in analytical results were combined, and the total uncertainty was estimated using Monte Carlo methods (ChemRisk 1994a, Appendix G). Releases were typically around 10–30 g y^{-1} for the years 1958–1971 and generally less than 10 g y^{-1} after 1971.

Environmental Monitoring. Beryllium has been monitored in water effluent since 1980 (ChemRisk 1994a). Routine surface water monitoring for beryllium has always shown less than 0.05 mg per liter of water, which is the analytical detection limit. The beryllium compounds of concern are not very water soluble and would be expected to bind to sediments and soils. Beryllium concentrations in the sediments of Great Western Reservoir and Standley Lake are similar to background levels and concentrations found in soil and sediment samples from other Rocky Mountain regions (ChemRisk 1994a; EPA 1975; DOE 1995a).

Barrick (1983) studied beryllium in soil near the RFP and estimated beryllium releases from the plant. This study measured beryllium in 241 soil and rock samples from the site and from nearby areas. The RFP-originated beryllium could not be distinguished from geological, naturally occurring beryllium taken on lands outside plant property. Natural and imported gravels at RFP had the highest and most variable beryllium concentrations. The mean concentration in these gravels was 1.1 $\pm$ 1.4 µg g^{-1} of soil. The background beryllium concentrations in Rocky Flats alluvium averaged 0.64 $\pm$ 0.07 µg g^{-1}. In an early draft of these results, Barrick (1982) suggested that atmospheric transport of beryllium to soils surrounding the plant had not occurred because no surficial soils near the plant were found to have elevated beryllium concentrations. The mean level in soils in the plant area was reported to be 0.6 µg g^{-1} and ranged from 0.2 to 1.1 µg g^{-1}. Higher levels found near roads and buildings were attributed to surficial gravel aggregates, which

had the highest background or natural beryllium levels. Additional studies were performed for the Colorado Department of Public Health and Environment in 1971, 1989, and 1992. Spatial variation of beryllium in soil did not indicate a plume of beryllium from the plant operations. Similar conclusions were reached in Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) reports in 1994 and 1995 (DOE 1995a; Allen and Litaor 1995).

The Dow Chemical Company site survey monthly reports from the 1950s contain some qualitative statements and a few quantitative measurements of beryllium in ambient air. Routine monitoring was conducted from 1970 to 1976 and reported in the Dow Chemical Company Monthly Environmental Reports. The RFP beryllium releases were less than the U.S. Environmental Protection Agency's (EPA's) discharge limit of 10 g per stationary source for a 24-hour period (EPA 1973) in the 1970s. It is likely that the ambient air monitoring results were not reported in the annual environmental reports because the results were thought to be low and the site was in compliance with EPA standards (Rope et al. 1999).

Health Hazards of Beryllium. Numerous studies have shown that beryllium compounds are carcinogenic in experimental animals by several routes of exposure, including inhalation; however, there has been considerable debate as to whether beryllium can cause cancer in humans. Currently, beryllium is classified by the EPA as a B2 probable human carcinogen, primarily on the basis of sufficient evidence from animal experiments. Evidence in humans is considered inadequate or limited (EPA 1998a). Beryllium risk values were scheduled to be reevaluated by the EPA in 1997.

Chronic beryllium disease or berylliosis is also a health endpoint of concern. Chronic beryllium disease is a progressive granulomatous disease. Although the lung is primarily involved, it is a systemic disease and granulomatous inflammation may involve other organs. No dose response relationship has been established for chronic beryllium disease, which is interpreted as involving a delayed-type hypersensitivity so that even very low exposures may be sufficient to induce it. Chronic beryllium disease can develop in people with relatively low exposures, whereas nonsensitized people experiencing high exposures may not develop the disease (Deodhar and Barna 1991; Wagoner et al. 1980; Mancuso 1980). The occurrence of beryllium disease in those with inadvertent or seemingly trivial exposure has been reported in secretaries and security guards at the RFP (Kriess et al. 1993b) and other facilities (Lang 1994). Although Kriess et al. (1993b) reported that the degree of beryllium exposure was associated with disease rates, they found that sensitization occurred in workers with exposures as short as a 1 month or in people with unrecognized exposure. In light of the complexity and apparent immunological (no dose-response) nature of chronic beryllium disease, the Health Advisory Panel for the Rocky Flats Health Studies chose lung cancer, rather than chronic beryllium disease, as an end point for risk assessment.

Carcinogenic potency factors (slope factors) were obtained from EPA (1987b). A distribution was assigned to these values based on the relative risk estimates. The slope factor was assigned a triangular distribution with a maximum of 25 kg-d mg^{-1}, a minimum of 0.56 kg d mg^{-1}, and a most likely value of 8.4 kg d mg^{-1}.

Environmental Transport Modeling. Five atmospheric transport models ranging from a simple straight-line Gaussian plume model to a complex terrain model were evaluated for use in this study (Rood 1997). Models were compared to tracer measurements taken in the winter of 1991 at Rocky Flats. The results of this evaluation indicated no one model clearly outperformed the others. However, the puff trajectory models, RATCHET, TRIAD, and INPUFF2 generally

had lower variability and higher correlation to observed values compared to the other models. The RATCHET model was chosen for these calculations because it was particularly well suited for long-term annual-average dispersion estimates and it incorporates spatially varying meteorological and environmental parameters.

The model domain encompassed a 2,200 km^2 area (50 km north-south by 44 km east-west). The domain extended 28 km south, 12 km west, 22 km north, and 32 km east from the RFP. Most of the Denver metropolitan area and the city of Boulder were included in the domain. Reliable meteorological data from RFP is lacking before 1984. For this reason, a recent 5-year (1989–94) meteorological data set was run to determine annual average *X*/Q (concentration divided by release rate) values for 2300 receptor locations in the model domain. Meteorological data taken at the Denver Stapleton International Airport during the same period was also incorporated into the simulations. Annual average concentrations for each year were them determined by multiplying the annual release rate by the appropriate *X*/Q value.

Model prediction uncertainty was accounted for through the use of several multiplicative stochastic correction factors that accounted for uncertainty in the dispersion estimate, the meteorology, and deposition and plume depletion. Dispersion uncertainty was based on distributions on predicted-to-observed ratios from field tracer experiments using the Gaussian plume and other models including RATCHET. These values were derived from literature reviews and results from studies specific to this project. Meteorological uncertainty arises because we are using 5 years of meteorological data spanning a recent time period (1989–1993) to define an annual average *X/Q* value that will be applied to all previous years of the assessment period (1952–1989). This correction factor was derived from studies performed for the Fernald Dosimetry Reconstruction Project (Killough et al. 1996) and comparisons made at Rocky Flats. Deposition and plume depletion uncertainty factors were calculated using the Monte Carlo sampling features of RATCHET. All correction factors were distributed lognormally and were combined with the source term uncertainty to yield distributions of predicted concentrations at selected receptor locations. Monte Carlo techniques were used to propagate model prediction uncertainty through to the final risk calculations.

Predicted Concentrations. Median value predicted concentrations at the location of highest concentration outside the buffer zone (east of the plant along Indiana Street) ranged from 1.3×10^{-6} ng m^{-3} in 1986 to 7.3×10^{-4} ng m^{-3} in 1968, the year of highest release. The maximum concentration in the model domain for the year of highest release (1968) was calculated within the plant buffer zone and ranged from 2.5×10^{-3} ng m^{-3} (5^{th} percentile) to 6.8×10^{-2} ng m^{-3} (95^{th} percentile). This can be compared with an annual average natural background range of 0.03 to 0.3 ng m^{-3}, (median of 1×10^{-1} ng m^{-3}) estimated in Rope et al. (1999). Note that the predicted offsite concentrations would be indistinguishable from background.

Exposure Scenarios. The risk that a person receives depends upon a number of factors, such as

- Lifestyle (that is, did the person spend a great deal of time outdoors or doing heavy work on a farm)
- When and how long that person lived near the RFP (for example, during the key release events in 1957 and late 1960s or in the 1970s when releases were less)
- Age and gender of the person
- Where the person lived and worked in relation to the RFP.

To consider these features of a person's life, we developed profiles, or exposure scenarios, of hypothetical, but typical residents of the RFP area for which representative risk estimates could be made. Risks were calculated for nine hypothetical exposure scenarios. These scenarios incorporate typical lifestyles, ages, genders, and lengths of time in the area. and can help individuals determine risk ranges for themselves by finding a lifestyle profile that most closely matches their background. The scenarios are not designed to include all conceivable lifestyles of residents who lived in this region during the time of the RFP operations. Rather, they provide a range of potential profiles of people in the area.

The nine exposure scenarios include a rancher located outside the east cattle fence along Indiana Avenue, a housewife who lived in Broomfield, a child who grows up in Broomfield during the operational period of the RFP (1953–89), and several receptors (retiree and office worker) who move into the Denver Metropolitan area in the 1970's.

Each receptor scenario incorporates inhalation rates that reflect their lifestyle. For example, the rancher's breathing rate reflects one who performs manual labor for part of the day. Uncertainty was not incorporated into the exposure scenarios; that is, the physical attributes and behavior of the receptors were assumed to be fixed. The calculated risks were not intended to represent a population of receptors who exhibit a given behavior.

Risk Estimates. Geometric mean incremental lifetime cancer incidence risk estimates for beryllium inhalation were greatest for the rancher scenario (3.9×10^{-10}) and least for the retiree scenario (7.5×10^{-13}). Using the rancher scenario as an example, these risks may be interpreted as follows:

- *There is a 90% probability that incremental lifetime cancer incidence risk to the lung for the rancher was between 7.5×10^{-11} (5% value) and 1.8×10^{-9} (95% value).*
- *There is a 5% probability that incremental lifetime cancer incidence risk to the lung for the rancher greater than 1.8×10^{-9}*
- *There is also a 5% probability the risk was less than 7.5×10^{-11}*

All risk estimates were well below the EPA point of departure for acceptable risks (10^{-6} to 10^{-4}).

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ACRONYMS

ACGIH	American Conference of Governmental Industrial Hygienists
ASOS	Automatic Surface Observation Site
CDPHE	Colorado Department of Public Health and Environment
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
DOE	U.S. Department of Energy
EPA	U.S. Environmental Protection Agency
GM	geometric mean
GSD	geometric standard deviation
HAP	Health Advisory Panel
INPUFF	INtegrated PUFF dispersion code
IRIS	Integrated Risk Information System
NIOSH	National Institute for Occupational Safety and Health
OSHA	Occupational Safety and Health Administration
OU	Operable Unit
RAC	*Radiological Assessments Corporation*[1]
RATCHET	Regional Atmospheric Transport Code for Hanford Emission Tracking
RfC	reference concentration
RFP	Rocky Flats Plant
SF	slope factor
TRAC	Terrain Responsive Atmospheric Code
TIC	time-integrated concentration
USGS	U.S. Geological Survey
UTM	universal transverse mercator

[1] In 1998 *Radiological Assessments Corporation* changed its name to *Risk Assessment Corporation*. For consistency throughout the project, all reports were published by *Radiological Assessments Corporation*.

INTRODUCTION

The Rocky Flats Environmental Technology Site is owned by the U.S. Department of Energy (DOE) and is currently contractor-operated by Kaiser-Hill Company. For most of its history, the site was called the Rocky Flats Plant (RFP) and was operated by Dow Chemical Company as a nuclear weapons research, development, and production complex (Figure 1). The RFP is located on approximately 2,650 ha (6,500 acres) of Federal property, about 8–10 km from the cities of Arvada, Westminster, and Broomfield, Colorado and 26 km (16 mi). northwest of downtown Denver, Colorado. The original 156 ha (385 acre) main production area is surrounded by a 2,490-ha (6,150-acre) buffer zone that now delineates the RFP boundary.

Through a 1989 Agreement in Principle between the DOE and the State of Colorado, DOE provided the State with funding and technical support for health-related studies. The purpose of the Historical Public Exposures Studies on Rocky Flats is to identify potential health effects in residents in nearby communities who may have been exposed to past toxic and radioactive releases. The Colorado Department of Public Health and Environment (CDPHE) first invited a national panel of experts to help design the health studies. Because of intense public concern about Rocky Flats contamination among Denver metropolitan area residents following a Federal Bureau of Investigation raid of Rocky Flats in June 1989, the panel decided to stress public involvement and to separate the research into two major phases conducted by two different contractors to enhance accountability and credibility.

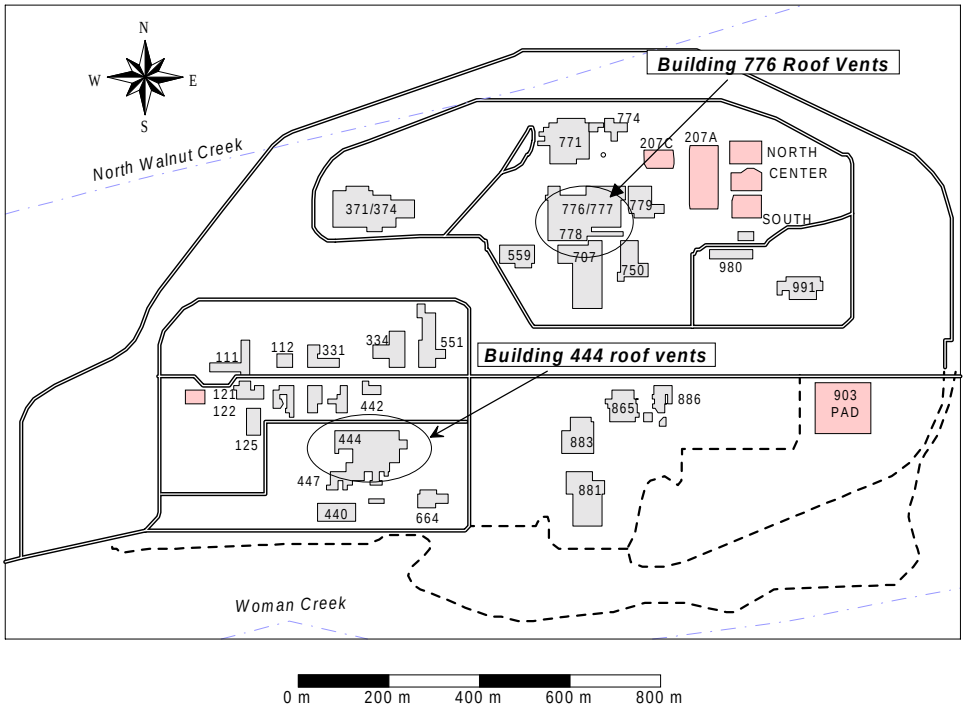

Figure 1. Main production area of the Rocky Flats Plant. Originally, the buildings were identified with two-digit numbers. Later, a third digit was added. The production area, now sometimes called the industrial area, is surrounded by a security perimeter fence. The area between the perimeter fence and Indiana Street to the east is the buffer zone. The buffer zone was expanded to Indiana Street in the 1970s. Major beryllium release points are identified.

Phase I of the study was performed by ChemRisk (a division of McLaren/Hart, Environmental Engineering). In Phase I, ChemRisk conducted an extensive investigation of past operations and releases from the RFP. The Phase I effort identified the primary materials of concern, release points and events, quantities released, transport pathways, and preliminary estimates of dose and risk to offsite individuals. The conclusions from Phase I were released in a public summary document (HAP 1993), a series of task reports by ChemRisk, and several articles in the journal *Health Physics*.

Radiological Assessments Corporation (*RAC*) was awarded the contract to conduct Phase II of the study, which is an in-depth investigation of the potential doses and risks to the public from historical releases from Rocky Flats. Recommendations for work to be performed in Phase II are outlined in the Phase I summary document HAP (1993).

This report documents risk calculations for inhalation of beryllium in air resulting from normal operational releases at the RFP. A brief review of the Phase I work and beryllium source terms is provided. This report evaluates soil and sediment monitoring data for beryllium and discusses regulatory guidelines, evidence of carcinogenicity and chronic beryllium disease. It also describes environmental transport modeling, provides estimates of uncertainty in the model predictions, and presents distributions of carcinogenic risk resulting from inhalation of beryllium for several generic receptor scenarios.

BERYLLIUM SOURCE TERM ESTIMATES

Beryllium Use at the Rocky Flats Plant

Although, initial research and developmental work with beryllium began in 1953 at the RFP, foundry operations with beryllium became significant from 1958 to 1975. Details of beryllium component manufacturing, machining, cutting, heat treating, rolling and other operations and ventilation systems used to control beryllium emissions over the years are described in the Task 3 & 4 report for Phase I of the Rocky Flats Dose Reconstruction Project (ChemRisk 1992) and in a letter written by Campbell (1986). Most of the beryllium work was done in Buildings 444 and 883. The airborne emission points for beryllium are listed in Table 5-1, pages 169–176, of the Phase I Task 3 & 4 report (ChemRisk 1992). With the possible exception of effluent from Building 441 in the early 1960s, all air exhaust discharged from plant facilities that processed beryllium was subjected to high-efficiency particulate air (HEPA) filtration designed for controlling radioactive effluents (ChemRisk 1994a).

Beryllium Releases Estimated from Effluent Monitoring Data

Beryllium has been monitored in the plant air exhaust effluent since at least 1963 (ChemRisk 1992, 1994a; Hammond 1963). The Phase I Task 5 report describes the monitoring program and summarizes the release data generated as a result of the monitoring program (ChemRisk 1994a).

The monitoring program data for routine airborne emissions of beryllium served as the basis for the Phase I release estimates (Tables 1 and 2 and Figure 2). ChemRisk compiled a record of beryllium emissions using sample data logbooks for 1960 through 1970 and annual beryllium releases reported in the annual Environmental Monitoring Reports for 1971 through 1989. The logbooks contain daily sample results for workroom air and building effluents. The monthly and

annual average beryllium concentrations for each stack were calculated from the building effluent data. Data on exhaust flow rates and total exhaust volume were lacking for some facilities and had to be estimated using facilities of similar size. No sampling data from before 1960 were located. It was assumed that emissions in 1958 and 1959 were the same as those reported in 1960.

Air exhaust samples were taken from filter plenum exhausts after the air passed through HEPA filters but before it exited the stack. The sampling practices, sampling system design, sample line losses, calculations of flow rates, and exhaust volume and uncertainties discussed previously for radioactive particles (ChemRisk 1994a), were applied to the beryllium sampling data. The Task 5 report for Phase I describes the different analytical techniques used over time (ChemRisk 1994a).

The Phase I report also discusses beryllium released during three fires that occurred in Building 444 in 1962, 1964, and 1978 (West 1978; Werkema 1978a). Any releases as a result of the fires would have been monitored by the stack sampling equipment, so they were thought to be included in the Phase I release estimates (ChemRisk 1994a). The most significant fire occurred on February 23, 1978. A release estimate of 14.5 g from the fire was incorporated into the Phase I release estimate of <17 g total for 1978. This estimate was based on monitoring results from the plenum sampler; ambient air sampling, and water samples from water used to fight the fire that drained into and was sampled from ponds, ditches and temporary impoundments (Rockwell 1978; ChemRisk 1994a; Hawes 1978; Werkema 1978b). Other estimates of a maximum beryllium source term from the fire were 10 kg (Rockwell 1978), and 8-10 kg (West 1978; Werkema 1978a). These release estimates were used to calculate a maximum air concentration of 8 to 10 µg m^{-3} at Highway 93, using Gaussian plume calculations. One calculation led to an estimate of 91.7 kg of beryllium (West 1978), which was later recognized as "so gross an overestimate" as to be discounted (Werkema 1978a).

Beryllium release summaries in the Task 5 report (ChemRisk 1994a) suggest low release values that average tens of grams or less annually. Documentation suggests that beryllium measurement data handling practices may have led to reporting annual emissions that were greater than actual releases (ChemRisk 1994a).

Table 1. Total Annual Release Estimates for Beryllium from Table 3-2 of the Phase I Task 5 Report[a]

Year	Annual release estimate (g)	Year	Annual release estimate (g)
1958	13.0 [b]	1965	31.0
1959	13.0 [b]	1966	33.0
1960	13.0	1967	33.0
1961	11.0	1968	38.0
1962	6.7	1969	24.0
1963	12.0	1970	14.0
1964	12.0	Total (1958–1970)	253.7

a Source: ChemRisk (1994a)
b No monitoring data were found for 1958 and 1959 and releases in those years were assumed to be the same as 1960.

Beryllium releases from 1971 to 1989 were obtained from the Annual Environmental Monitoring Reports issued by the RFP. These reports often reported beryllium release totals for the year as less than values. The 1975 report explained that samples with concentrations less than the minimum detectable concentration were considered to be at the minimum detectable concentration for averaging purposes. Averages calculated using results below the minimum detectable concentration level were identified with a less than (<) sign. ChemRisk included the < sign in reporting their compilation of annual average results from the Environmental Monitoring Reports.

Table 2. Reported Annual Beryllium Releases from the Rocky Flats Annual Environmental Reports Compiled in Table 3-3 of the Phase I Task 5 Report[a]

Year	Reported release (g)	Year	Reported release (g)
1971	16	1981	0.2
1972	<2.0	1982	0.1
1973	<7.1	1983	–0.1[a]
1974	<10	1984	0.3
1975	<5.2	1985	0.5
1976	<3.7	1986	0.1
1977	<4.9	1987	0.2
1978	<17	1988	0.1
1979	<1.5	1989	0.6
1980	<1.1	Total 1971-1989	70.5

a Source: ChemRisk 1994a

b The Annual Environmental Monitoring Report estimated an annual emission total for 1983 as a negative number.

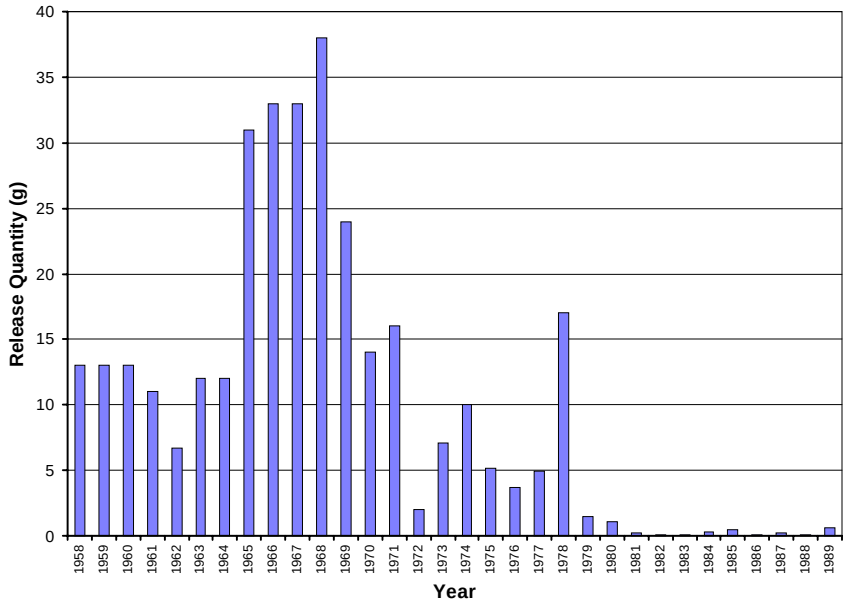

Figure 2. Annual release estimates for beryllium as estimated by ChemRisk (1994a).

The annual emission total for 1983 is a negative value of –0.1 g. The explanation in the annual report suggested that the level in air sampled this year could not be distinguished from background levels (ChemRisk 1994a). Explanation for differences from year to year are not offered in the Phase I Task 5 report. Documented changes in production and upgrades in cyclone separators and exhaust filtration are described but do not appear to correspond to the release estimates (ChemRisk 1994a). Releases in the 1960s are probably related to peak production of beryllium parts. Changes in exhaust ventilation were made in 1964, 1974, and several times in the 1980s (Holwager 1996). The variability in sample measurements was high, largely because of averaging measurements of relatively low concentrations in large volumes of exhaust air. Changes in analytical methods, ways of averaging effluent measurements, adjustments for background levels, and changes in methods that altered the solubility of beryllium in the samples are examples of reasons for the variability of the release estimates (Barrick 1997; Daugherty 1997). Quality assurance for effluent measurements are further discussed in the Phase II Task 4 Report (Rope et al. 1999).

Uncertainty in the Source Term Estimates

Uncertainty associated with the beryllium source term estimates was characterized using the same approach applied to plutonium and uranium sample measurements and release estimates. Uncertainties in exhaust and sample flow rate estimates, and in analytical results were combined, and the total uncertainty was estimated using Monte Carlo methods (ChemRisk 1994a, Appendix G). The Task 5 report (ChemRisk 1994a) also contained a description of another source of uncertainty, as follows:

> The beryllium release summaries suggest extremely low environmental emissions of beryllium averaging in the tens of grams or less annually. A 1980 plant internal letter indicated that, based on an evaluation by the plant's General Service Laboratory, use of the minimum detectable amount value for beryllium at each effluent measurement location would result in a calculated minimum beryllium discharge per month of 0.4 gram (Hornbacher 1980). This would lead to a reported yearly minimum discharge of about 4 to 5 grams even if none of the samples had a positive result. The information that was reviewed suggests that the beryllium data handling practices may have led to the reporting of annual emissions that were higher than the actual releases. However, given the low magnitude of the reported emissions, the uncertainty introduced by this practice has not been characterized.

An uncertainty correction factor was applied to the source term estimates based on the year of the release. The uncertainty correction factor is described in the Source Characterization section in the Environmental Transport Modeling section.

ENVIRONMENTAL MONITORING FOR BERYLLIUM

Beryllium in Surface Water

Beryllium had the potential to be transferred offsite in surface water, and this pathway was investigated in Phase I. Beryllium has been monitored in water effluent since 1980 (ChemRisk 1994a). Routine surface water monitoring for beryllium has always shown less than 0.05 mg per liter of water, which is the analytical detection limit.

ChemRisk reported that available data on surface waterborne releases from the RFP were not sufficient to develop direct estimates of release for beryllium. They concluded that, "the only information available for addressing past releases from the plant would be measurements of beryllium in reservoir sediments" (ChemRisk 1994a). The beryllium compounds of concern are not very water soluble and would be expected to bind to sediments and soils. Beryllium concentrations in the sediments of Great Western Reservoir and Standley Lake are similar to background levels and concentrations found in soil and sediment samples from other Rocky Mountain regions (ChemRisk 1994a; EPA 1975; DOE 1995a).

Historically, inhalation of beryllium has been a much greater human health concern than ingestion, in part because less than 1% of ingested beryllium is absorbed through the gastrointestinal tract (EPA 1991). Beryllium does not bioaccumulate in fish. Releases of beryllium to surface water were not evaluated further because of a lack of source term and effluent and environmental monitoring data, insufficient evidence of accumulation in soils and sediments, and the low solubility and gastrointestinal absorption of beryllium.

Beryllium in Soil

Beryllium concentrations in soil are of interest because resuspension of beryllium in soil is of potential concern and a pattern of beryllium contamination in soil could reveal information about discharge from the plant. Beryllium sources that might affect concentrations in soils at the RFP include

- Operations at the plant
- A beryllium ore industry located 2 km east of the plant
- A beryllium ceramics industry 15 km south of the plant
- Coal burning and other combustion sources near the plant
- Beryllium in gravels brought into the site
- Naturally occurring beryllium (Barrick 1982; Kray 1992).

A study to characterize sources of beryllium and to quantify environmental beryllium contamination in soil near the RFP was conducted in 1981 (Barrick 1983). This study reported an estimate of 196 g for the total amount of beryllium exhausted from all buildings that processed beryllium during the 24-year period 1958 to 1982. This estimate included releases from two reported filter-fire accidents in February 1978, which released 14.5 g of beryllium from the main beryllium production building, (Building 444). In this study, 241 soil and rock samples from the site and from nearby areas were obtained. Deeper samples were taken at 5 to 10 cm to establish geological background levels of beryllium. The study concluded that RFP-originated beryllium could not be distinguished from geological, naturally occurring beryllium taken on lands outside plant property. The survey found that natural gravels and an estimated 36 million kg of gravels

brought in and added to RFP surfaces have the highest and most variable beryllium concentrations. The mean concentration in these gravels is 1.1 $\pm$ 1.4 μg g^{-1} of soil (parts per million or milligrams per kilogram of soil). The background beryllium concentrations in soil (Rocky Flats alluvium) averaged 0.64 $\pm$ 0.07 μg g^{-1}.

In what appears to be an earlier draft of these results, Barrick (1982) suggested that atmospheric transport of beryllium to soils surrounding the plant had not occurred because no surficial soils near the plant were found to have elevated beryllium concentrations. The mean level in soils in the plant area was reported to be 0.6 μg g^{-1} and ranged from 0.2 to 1.1 μg g^{-1}. Higher levels found near roads and buildings were attributed to surficial gravel aggregates, which had the highest background or natural beryllium levels.

One accumulation of beryllium in soil that likely originated from RFP operations was found 30 m from the stack of a plant that processed beryllium. The samples at this location were 44 to 69 μg g^{-1} of soil above background. Subsequent to this study, more samples were taken at various depths to try to determine when the accumulation in soil had occurred. A high beryllium soil concentration was found in a 10 × 10 m^2, adjacent to door 10 of Building 444. The beryllium contamination was found within the top 5 cm of soil and ranged from 1 to 114 μg g^{-1}. The pattern of contamination suggested the source of the beryllium release was door 10 and not the nearby stack for Building 444 or the filter plenum room. The study's authors recommended removal of 1 m^3 of contaminated soil (Barrick 1983). Before 1970, chlorinated hydrocarbon solvent that had been used to rinse beryllium parts was disposed of by pouring it on soil outside door 10 on the south side of Building 444. A special study to sample beryllium in air near this solvent disposal site was conducted in the summer of 1977. Filters were collected weekly from an air monitor mounted 3 ft above ground level. The detection limit was reported to be approximately 7.5×10^{-9} μg m^{-3}. The maximum level reported at this location was 2.3×10^{-3} μg m^{-3} and the average was 9×10^{-4} μg m^{-3}. Because the air concentrations averaged about 9% of the ambient air standard of 0.01 μg m^{-3}, soil removal was not recommended at that time (Barker 1978).

The CDPHE conducted studies on beryllium in soil in 1971 and 1989. In 1992, at the request of Bob Quillan, (a Health Advisory Panel member representing the CDPHE), a discrepancy in the 1971 and 1989 beryllium soil sampling results was evaluated by CDPHE personnel (Quillan 1992). The study done in 1989 reported 21 results, all less than the analytical detection limit of 2.7 μg g^{-1}. The 1971 data consisted of 13 results, ranging from 2.0 to 60 μg g^{-1} with no analytical detection limit reported. The pattern of positive values seen in 1971 was not consistent with what would be expected if the beryllium in the soils had been deposited because of atmospheric dispersion from the RFP. Spatial variations did not indicate a plume of beryllium from the plant operations. The 1989 dataset was judged to be more credible because of better documentation of analytical procedures, more rigorous quality assurance, and improved analytical methods and equipment (Kray 1992).

To investigate potential contamination of surface soils from windborne dispersal in Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) Operable Unit (OU) 3 (offsite), the distribution of metals in OU 2 (onsite areas) was evaluated. If contamination of soils onsite was due to activities at the plant, then the soil sampling results were expected to show a distinct spatial distribution trend of decreasing concentrations with increasing distances from areas where beryllium was used. The CERCLA program personnel reasoned that if metal contamination of soil in onsite areas (OU 2) was at background concentrations or

appeared to be a result of localized incidents of contamination, and if no spatial trends could be identified, then contamination in offsite (OU3) soils was unlikely and sampling of OU 3 soils (at distances further out) would not be warranted (DOE 1994). Samples were compared with results from two studies of background concentrations: the Rock Creek and the Background Soils Characterization Project. The study found a mean beryllium concentration in OU2 soils of 0.68 $\mu g\ g^{-1}$ a standard deviation of 0.21 $\mu g\ g^{-1}$, and a coefficient of variation of 0.31. One extreme value of 1.50 $\mu g\ g^{-1}$ was determined to be an outlier. The beryllium concentrations in OU 2 soils were similar to those for Rock Creek soil samples, which had a mean value of 0.68 $\mu g\ g^{-1}$ and a maximum concentration of 0.96 $\mu g\ g^{-1}$. The Background Soils Characterization Project study showed a similar mean of 0.66 $\mu g\ g^{-1}$. The U.S. Geological Survey (USGS) geometric mean for beryllium concentrations in the Front Range soil was 1.2 $\mu g\ g^{-1}$. Beryllium concentrations in the OU2 soil did not appear to be above background. No spatial trends in the data or recognizable plume were apparent and no preferred direction of beryllium in soils was observed. This study did not provide evidence of airborne contamination (DOE 1995a; Allen and Litaor 1995).

The results of soil monitoring conducted as a part of investigations of CERCLA OU 5 (Woman Creek Drainage) and OU 6 (Walnut Creek Drainage) do not suggest a windborne deposition pattern (DOE 1995b, 1996).

Samples of cottonwood leaves, collected from trees growing on soils with 0.1 to 1.0 $\mu g\ g^{-1}$ beryllium, contained beryllium concentrations that were only slightly correlated ($r = 0.25$) with the concentration in the soil. This observation led researchers to decide that leaf surveys would not be useful as indicators of soil contamination (Barrick 1983), although analysis of cottonwood trees has been used to locate beryllium ore deposits in former Soviet Union.

Taken together, the soil data suggest that beryllium deposited on soil from RFP releases is not distinguishable from beryllium in the soil from natural and other sources.

Beryllium in Ambient Air

Historical ambient air monitoring for beryllium in the vicinity of the RFP is reviewed and put into perspective in the Phase II Task 4 report which evaluates historical monitoring data (Rope et al. 1999). The beryllium ambient air data were not considered as a part of Phase I.

The Dow Chemical Company site survey monthly reports from the 1950s contain some qualitative statements and a few quantitative measurements of beryllium in ambient air. Routine monitoring was conducted from 1970 to 1976 and reported in the Dow Chemical Company Monthly Environmental Reports. The RFP beryllium releases were less than the U.S. Environmental Protection Agency's (EPA) discharge limit of 10 g per stationary source for a 24-hour period (EPA 1973) in the 1970s. It is likely that the ambient air monitoring results were not reported in the annual environmental reports because the results were thought to be low and the site was in compliance with EPA standards (Rope et al. 1999).

Monthly average concentrations measured onsite from January 1972 to February 1973 ranged from 0.7 to 1.5 ng m^{-3}. Offsite samplers over the same period had a monthly average concentrations of 0.2 to 1.5 ng m^{-3}. Both onsite and offsite measurements were similar for this period. The long term average concentration in onsite air from June 1973 to October 1976 was about a factor of 3 greater than the median estimated natural background concentration of 0.1 ng m^{-3} reported in Rope et al. (1999). Maximum short-term (1 week) onsite air concentrations ranged from 0.3 to 2.3 ng m^{-3}. It was suspected that resuspension of a nearby contaminated soil

plot influenced these measurements and they were not a result of emissions from plant operations. The resuspension of contaminated soil did not appear to contribute significantly to offsite air concentrations

Time trend analysis suggests that the concentrations in onsite air appear to be unrelated to the amount of beryllium released from the plant. The monitoring data support the model predictions that offsite air concentrations of beryllium are well below background concentrations (Rope et al. 1999).

Beryllium in Waste

Beryllium was also present in waste, some of which was discharged into the solar evaporation ponds. ChemRisk described the disposal of waste from the Coors Porcelain Company at the RFP (ChemRisk 1992). Resuspension or leaching of beryllium in waste has not occurred in the past to an extent to warrant inclusion in this study.

THE HEALTH HAZARD OF BERYLLIUM

This section describes the regulatory standards for beryllium in air, summarizes evidence of carcinogenicity, and reviews the literature on chronic beryllium disease.

Regulatory Guidelines for Beryllium

Because of beryllium's use in the nuclear weapons industry, the Atomic Energy Commission recommended occupational and community ambient air standards for beryllium in 1949 (Eisenbud et al. 1949) that greatly reduced beryllium exposures in and around beryllium plants. The community air standard became the first ambient air quality standard in the United States. It preceded all others by about 25 years, and the standard remains unchanged to this day (Lang 1994). The ambient air standard, also called the 'neighborhood' air standard, limits beryllium concentrations in air surrounding factories to 0.01 $\mu g\ m^{-3}$, averaged over a 30 day period (EPA 1987a). The occupational limit is 0.002 $mg\ m^{-3}$ (Meyer 1994).

Beryllium Carcinogenicity

Numerous studies have shown that beryllium compounds are carcinogenic in experimental animals by several routes of exposure, including inhalation; however, there has been considerable debate as to whether beryllium can cause cancer in humans.

A number of epidemiological studies have reported an increased risk of lung cancer in beryllium workers, but deficiencies in the studies have not allowed unequivocal conclusions to be made (Meyer 1994; IARC 1980; EPA 1987b). Criticisms include little or no consideration of smoking history, exposure to other potential lung carcinogens, and underestimating expected cancer deaths in control populations (Smith 1981; Meyer 1994).

In a review of the U.S. Beryllium Case Registry Data, Hardy (1980) reported there was no evidence to support beryllium as a human carcinogen, but the author recommended workers be studied.

Four epidemiological studies conducted before 1970 did not clearly demonstrate a relationship between exposure to beryllium compounds and the occurrence of human cancer, but excess risk is suggested by the results of all of the studies (Wagoner et al. 1980; Mancuso 1980; IARC 1980; EPA 1998a).

Additional studies in the 1990s found excess risk of lung cancer in workers enrolled in the Beryllium Case Registry (Steenland and Ward 1991). Occupational exposure to beryllium compounds was said to be the most plausible explanation for the increased risk of lung cancer observed in these studies (Ward et al. 1992).

Four International Agency for Research on Cancer (IARC) working groups (in 1972, 1980, 1987, and 1993) reviewed the animal and epidemiological data on beryllium carcinogenicity. The first working group considered the epidemiological studies available at that time as inadequate to evaluate the human carcinogenicity. In 1980 and 1987, the working group concluded that beryllium was carcinogenic to animals but that epidemiological evidence was limited. They classified beryllium as a suspect human carcinogen. Epidemiological evidence was again carefully scrutinized by the IARC working group convened in 1993. The proceedings of the 1993 conference, in IARC Monograph Volume 58, states that compounds of beryllium are carcinogenic in animals by a number of routes, and several beryllium compounds produce lung tumors in rats exposed by inhalation. The working group concluded that there was sufficient evidence in experimental animals for the carcinogenicity of beryllium and beryllium compounds. After a review of all available epidemiological studies, the working group concluded that there was also sufficient evidence in humans for the carcinogenicity of beryllium and beryllium compounds. However, controversy about the classification of beryllium as a human carcinogen continues.

Studies implicating beryllium as an occupational carcinogen have examined lung cancer in cohorts exposed in the 1930s and 1940s, before industrial hygiene controls were in place, when concentrations were orders of magnitude higher than permitted today. Statistically significant increases have been difficult to demonstrate in workers exposed to lower levels (Jameson 1996).

Currently, beryllium is classified by the EPA as a B1 probable human carcinogen. When this risk assessment was presented to the Health Advisory Panel, the EPA classification was B2, on the basis of sufficient evidence from animal experiments. Evidence in humans was considered inadequate or limited. Beryllium risk values were scheduled to be reevaluated by the EPA in 1997. The evaluation was completed and the weight-of-evidence classification changed from B2 to B1 in April of 1998 (EPA 1998a).

Chronic Beryllium Disease

The potential for historical releases of beryllium from the RFP to have caused chronic beryllium disease in offsite individuals was not addressed in Phase I.

Chronic beryllium disease, also called berylliosis, is a progressive granulomatous disease. Although the lung is primarily involved, it is a systemic disease and granulomatous inflammation may involve other organs. A delayed-type hypersensitivity reaction is thought to play a central role in the pathogenesis of chronic beryllium disease. Sensitization to beryllium can be detected by measuring in vitro proliferative responses of bronchioalveolar lavage lymphocytes or peripheral blood lymphocytes to beryllium. Clinical and experimental animal data on chronic beryllium disease support an immunologic, hypersensitivity mechanism for chromic beryllium

disease. Information consistent with such a mechanism includes (a) the insidious nature of the disease, (b) a long latency between exposure and onset, (c) the granulomatous nature of the lung lesions that develop, (d) berylliosis patients show delayed skin hypersensitivity reactions to beryllium compounds, (e) peripheral blood lymphocytes and bronchoalveolar lymphocytes in people with chronic beryllium disease undergo blast transformation and release of migration inhibition factor after exposure to beryllium in vitro, and (f) the lack of a dose-response relationship (Deodhar and Barna 1991; Hardy 1980; Kriess et al. 1993a; Aller 1990; Clarke et al. 1993).

Susceptibility to sensitization is likely to have a genetic basis. Recently, a genetic marker was identified in people with sensitivity to beryllium (Richeldi et al. 1993). It was concluded that people with this genetic marker have a significantly increased probability of developing sensitization than those without it (Newman 1993). However, it appears that about 30% of the population has the genetic marker and at most, only about 2–15% of exposed workers become sensitized (Lang 1994).

Most commonly, researchers estimate that 1 to 5% of beryllium-exposed workers develop chronic beryllium disease (Eisenbud and Lisson 1983; Meyer 1994; EG&G Rocky Flats 1991). Sensitization rates may be higher. Kriess et al. (1993b) reported 2.9 to 15.8% for beryllium-exposed persons.

Most cases of chronic beryllium disease have occurred in people working in industries processing or using beryllium; however, cases of chronic beryllium disease have been reported in people living near processing plants and in families of beryllium workers, perhaps from exposure to airborne beryllium carried from a plant or from handling contaminated worker's clothing. Chronic beryllium disease has also developed in peripheral workers, people in nonprocessing areas of factories, who were likely exposed to very small amounts of beryllium (Drury et al. 1978; Hardy 1980; Hasan and Kazemi 1974; Meyer 1994). Although Kriess et al. (1993b) reported that the degree of beryllium exposure was associated with disease rates, they found that sensitization occurred in workers with exposures as short as a 1 month or in people with unrecognized exposure.

The occurrence of beryllium disease in those with inadvertent or seemingly trivial exposure has been reported in secretaries and security guards at the RFP (Kriess et al. 1993b) and other facilities (Lang 1994), a janitor in a ceramics company (Lang 1994), and in members of worker's households and neighbors around beryllium extraction plants (Eisenbud et al. 1949; Eisenbud and Lisson 1983; EPA 1998a). The latter are termed neighborhood cases, which are cases of chronic beryllium disease that occur in people living in the vicinity of the beryllium plants (EPA 1987b).

In a report summarizing the relationship between the incidence of nonoccupational related cases of chronic beryllium disease and the levels of atmospheric contamination, Eisenbud et al. (1949) observed that the distribution of the cases with respect to the plant indicated that the incidence of disease was a function of the concentration to which the residents were exposed. The incidence of disease within one-quarter of a mile was about 1% or 5 of 500 people (Eisenbud et al. 1949). The cases of chronic beryllium disease in the 1930s and 1940s in Salem, Massachusetts, occurred almost entirely in fluorescent lamp manufacturing workers except for three neighborhood cases: a night watchman, a near neighbor, and a housewife with two young women living in her home who were fluorescent lamp workers. Protection was minimal, and workers were exposed to high levels of beryllium phosphors (Hardy 1980).

Chronic beryllium disease occurred as an epidemic in the 1940s, leading to the establishment of the Beryllium Case Registry in 1951. The Case Registry is a file for cases of acute and chronic beryllium disease, now maintained by the National Institute of Occupational Safety and Health in Cincinnati (Lang 1994; EPA 1987b). In 1983, Eisenbud and Lisson reviewed the Beryllium Case Registry's 224 acute and 622 chronic cases. These cases included 577 occupational and 65 chronic beryllium disease cases attributed to ambient air pollution; 42 were attributed to ambient air exposure in areas in the vicinity of beryllium plants and 23 to exposure to dust brought home on work clothes. They reported no new cases for individuals exposed after 1972 and believed that control measures implemented in the 1950s had reduced chronic beryllium disease despite a marked increase in the use of beryllium. However, results of more recent research and clinical work have led to questions about the effectiveness of beryllium control measures and standards on reducing the incidence of chronic beryllium disease.

Although many researchers have praised the apparent effectiveness of the air standards for beryllium and have asserted that no new cases of beryllium disease have occurred since observance of these limits (Hurlbut 1974), others believe that the occupational standards may not be protective for sensitization (Newman 1993; Lang 1994) and the limit designed to protect the general public may not be low enough (Clarke et al. 1993). The EPA considers the ambient air standard protective for the public with ample margin of safety (EPA 1987b)

Lee Newman, of the Occupational and Environmental Medicine Division of the National Jewish Center for Immunology and Respiratory Medicine in Denver, was reported to believe that with increasing use of beryllium in industry, the absolute number of cases can be expected to increase. He said, "even with careful ventilatory controls and monitoring, cases of chronic beryllium disease will continue to occur such hypersensitivity can develop in some individuals following even low-level exposures, well within permissible exposure limits" (Lang 1994). Evidence exists for biological responses and possible sensitization occurring after exposure to levels far below the current threshold limit values (Clarke et al. 1993). In their 1998 Toxicological Review the EPA states that several studies have observed chronic beryllium disease in people chronically exposed in modern plants, that are generally in compliance with the workplace standard for beryllium (the permissible exposure limit) of 2 $\mu g\ m^{-3}$ (EPA 1998b).

A clear dose-response relationship or duration of exposure-response relationship has not been established for this chronic beryllium disease, which is interpreted as involving a delayed-type hypersensitivity so that even very low exposures may be sufficient to induce it. Chronic beryllium disease can develop in people with relatively low exposures, whereas nonsensitized people experiencing high exposures may not develop the disease (Deodhar and Barna 1991; Wagoner et al. 1980; Mancuso 1980). Even slightly exposed individuals, such as neighborhood cases, sometimes show severe clinical forms of the disease (Hardy 1980; Eisenbud et al. 1949).

Recent studies published in Kriess et al. (1993b) suggest that both individual sensitivity and degree of exposure or exposure circumstances are important in determining the risk of developing chronic beryllium disease. Although chronic beryllium disease cases have been associated with trivial or unrecognized beryllium exposure, chronic beryllium disease rates were higher in workers with presumed greater beryllium exposure, seeming to challenge the immunological dogma of no dose-response in chronic beryllium disease.

In their 1988 reevaluation of beryllium for IRIS, the EPA derived a reference concentration (RfC). The EPA defines the RfC as an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily exposure to the human population (including sensitive subgroups) that is

likely to be without appreciable risk of deleterious effects during a lifetime. For a good understanding the evidence for an exposure-response and the EPA's derivation of the RfC, see the EPA's Toxicological Review, available online through the IRIS database. The EPA cited the occupational study done by Kriess et al. (1996) as the basis for a lowest observable adverse effect level. The effect was beryllium sensitization, measured by the lymphocyte transformation test, a very sensitive endpoint. The results of Kriess et al.'s (1996) study, taken together with other work studies and studies of community residents living near a beryllium plant (Eisenbud et al. 1949) a RfC of 0.02 μg m^{-3} was determined. The uncertainty in the RfC is very large. Uncertainty Factors for human variability, the less-than-chronic exposure duration in the epidemiological study used, the very sensitive nature of the endpoint and the poor quality of the exposure monitoring in the study were included in the EPA's evaluation (EPA 1998b).

In general, the most appropriate end point for risk assessment is the effect that occurs at the lowest exposure. Because chronic beryllium disease can develop with very low-level exposure, it may be a better end point than lung cancer for assessing risk to low-level exposures. However, chronic beryllium disease may not be dose related, and the percentage of an exposed population that might be expected to develop the disease at a given exposure level is not known (Jameson 1996). In 1997, when this risk assessment was first completed, the Health Advisory Panel agreed that information on the dose-response for chronic beryllium disease was too uncertain to use for a risk assessment. Conducting a quantitative risk assessment was not feasible because of the lack of dose-response. Dr. Kriess presented her research to the Health Advisory Panel. She believes that the lymphocyte transformation test has allowed detection of an exposure-response relationship for beryllium sensitization.

Future research and studies now in progress may answer the question of whether or not a positive lymphocyte transformation test always corresponds to a case of chronic beryllium disease, and at what exposure does the sensitization occur. The EPA acknowledged that varying and generally low prevalence of chronic beryllium disease have been observed in workers, even when exposure concentrations were high (EPA 1998b).

In light of the complexity and apparent immunological (no dose-response) nature of chronic beryllium disease, lung cancer was chosen, rather than chronic beryllium disease, as an end point for this risk assessment. However, in 1998, when this report was undergoing final revision, the EPA's RfC was considered.

PHASE I EXPOSURE AND RISK CALCULATIONS

For Phase I, beryllium releases and transport in air were modeled using assumptions that are described in detail in the Task 6 report (ChemRisk 1994b). The predicted annual air concentrations were presented as concentration isopleths on maps of the site and surrounding area. In calculating deposition of airborne contaminants, it was assumed that the particles released were submicron in size because of HEPA filtration. Small deposition velocities were used, and dry and wet deposition were considered. The calculations and uncertainties are described in Section 3 of the Task 6 report (ChemRisk 1994b). Nine potentially important exposure pathways for beryllium were identified and listed in the Phase I Task 6 report. Using the predicted air and soil concentrations for Sector 12 (southeast of RFP), the exposure equations provided in Appendix I, and parameters in Appendix J, pathway specific doses were estimated in milligrams per kilogram per day using Monte Carlo simulation. The exposed

individual was assumed to be an adult consuming a typical amount of air, food, water, vegetables, and soil. Assumptions about time spent outdoors; consumption of local produce, beef, and milk; and exposure are described in the Task 8 report (ChemRisk 1994c).

Doses of beryllium in units of microgram per year were calculated and reported in the Task 8 Report (ChemRisk 1994c). The highest geometric mean (GM) dose reported in Appendix L for Sector 12 for 1968 was 1.1×10^{-4} μg y^{-1} for inhalation and 9.7×10^{-6} μg y^{-1} for all ingestion pathways combined. The resulting risk for 1 year of exposure reported in Appendix N had a GM of 1×10^{-12} and a geometric standard deviation (GSD) of 2.6 for inhalation risk and a GM of 2×10^{-12} and a GSD of 2.8 for ingestion risk.

PHASE II EXPOSURE AND RISK CALCULATIONS

Annual release estimates, release points, and the percentage contribution to the total releases from the site, reported in Phase I (ChemRisk 1992, 1994a) were used for Phase II calculations. The greatest release occurred in the year 1968. The greatest source of beryllium was operations in Building 444. The annual beryllium emission estimates for 1960–1970 were calculated from data compiled from sample data logbooks and using exhaust volume estimates made in ChemRisk (1992). Because emissions data were lacking before 1960, estimates for 1958 and 1959 were made from those calculated for 1960. Estimates for 1971–1989 were taken from the RFP Annual Environmental Monitoring Reports. ChemRisk independently calculated releases for 1 year during 1984 and found good agreement between their calculation and the value reported in the 1984 Annual Environmental Monitoring Report. The source term estimates consider releases during fires that occurred in Building 444.

For this assessment, inhalation of air is the exposure pathway of concern. Beryllium is not well absorbed after ingestion. Beryllium is relatively immobile in surface water, tending to absorb to soils and sediments, and would not be predicted to transport offsite to a great extent. Beryllium intake from ingestion of vegetation subject to deposition from the air, livestock inhaling air, surface water, soil and sediment, livestock ingesting soil or sediment, vegetation grown on soil, and livestock ingesting vegetation grown on soil could be evaluated but would be expected to contribute only a small amount to overall risk.

In Phase I, a suggestion was made to include the exposure route of dermal contact and wound entry. The response ChemRisk gave to this suggestion in Appendix O of the Task 8 Report (ChemRisk 1994c), which asserts that for the offsite population at large, dermal absorption and wound entry are not likely, is appropriate. Although these routes are important for occupational exposure, absorption through the skin or through wounds would be an extremely rare occurrence for individuals offsite.

Cancer Potency Determination

The EPA weight-of-evidence classification for beryllium is B2, a probable human carcinogen. B2 carcinogens are defined by the EPA as chemicals with sufficient evidence of carcinogenicity in animals with inadequate or a lack of human data.

The estimate of the excess lifetime cancer risk is the product of the dose and the carcinogenic potency slope factor (SF).

$$\text{Excess cancer risk} = \text{beryllium exposure concentrations} \times \text{SF}$$

Cancer SFs are usually estimated from animal studies using mathematical models, most commonly the linearized multistage model, for estimating the largest possible linear slope (within the 95% confidence limit) at extrapolated low doses that are consistent with the data. The SF is expressed in units of the inverse of milligram intake per kilogram body weight per day (kg d mg^{-1}). It represents the 95% upper confidence limit of the probability of a carcinogenic response per daily unit intake of a chemical over 70 years. The SF (and risk) is characterized as an upper-bound estimate. The true risk to humans, while not identifiable, is not likely to exceed the upper bound estimate.

The inhalation unit risk factor is the risk per unit concentration in air, calculated by dividing the SF by 70 kg and multiplying by the inhalation rate (20 m^3 d^{-1}) (EPA 1995).

$$UR = \frac{SF \ BR}{BW \ CF} \quad (1)$$

where

UR = unit risk (m^3 μg^{-1})
SF = slope factor (kg d mg^{-1})
BR = breathing rate (m^3 d^{-1})
CF = correction from mg to µg (1×10^3).

Using this relationship, a SF of 8.4 kg d mg^{-1} was calculated from the mean of the unit risk values published in Integrated Risk Information System (IRIS) (EPA 1998a).

For the quantitative estimate of the carcinogenic risk from inhalation exposure, the inhalation unit risk value was calculated using human occupational epidemiological data of Wagoner et al. (1980). Justification for this approach included the fact that humans are most likely to be exposed by inhalation to beryllium oxide rather than other beryllium salts, and animal studies of beryllium oxide have involved intratracheal instillation rather than inhalation.

Relative risk estimates were derived from the smoking-adjusted lung cancer death data. The relative risk estimates ranged from 1.36 to 1.44, and the 95% confidence limits of these estimates, 1.98 and 2.09, were used to estimate the lifetime cancer risk. The estimates were based on one dataset using a range of estimated exposures and exposure durations. The effective dose was calculated by adjusting for durations of daily (8/24 hours) and annual (240/365 days) exposure and the fraction of the lifetime at risk (duration of employment). Because of uncertainties in the beryllium exposure levels and exposure times, unit risks were derived using two estimates each of concentration: fraction of lifetime exposed and relative risk. These data are summarized in Table 3. The recommended value for use in risk assessment published in IRIS is 2.4×10^{-3}, the arithmetic mean of the eight derived unit risks. The values are conservative, calculated using the 95% confidence limit of the relative risk estimates. Absorption of beryllium is taken into account in developing unit risk levels. Although based on human data, which generally provide for more confidence than animal data, the quality of the study on which the estimates are based is considered poor because the study was confounded by several variables. A quantitative assessment based on animal studies was reported to have resulted in a similar estimate of risk (EPA 1998a, 1987b).

Table 3. Values from Human Inhalation, Occupational Exposure Studies Used to Calculate Unit Risk Values[a]

Workplace beryllium concentration ($\mu g\ m^{-3}$)	Fraction of the lifetime	Exposure concentration ($\mu g\ m^{-3}$)	95% upperbound estimate of relative risk	Unit risk ($m^3\ \mu g^{-1}$)
100	1.0	21.92	1.98	1.61×10^{-3}
100	1.0	21.92	2.09	1.79×10^{-3}
100	0.25	5.48	1.98	6.44×10^{-3}
100	0.25	5.48	2.09	7.16×10^{-3}
1000	1.0	219.18	1.98	1.61×10^{-4}
1000	1.0	219.18	2.09	1.79×10^{-4}
1000	0.25	54.79	1.98	6.44×10^{-4}
1000	0.25	54.79	2.09	7.16×10^{-4}

[a] Source: EPA 1987b, 1998a.

The *Health Assessment Document for Beryllium* (EPA 1987b) describes deficiencies of the epidemiological data, efforts by the EPA's Carcinogen Assessment Group to adjust the data for use in calculating cancer potency, and assumptions and models used to extrapolate from high occupational exposures to low-level exposures.

According to the information in IRIS, an EPA workgroup last assessed the beryllium risk values in 1988. The EPA is currently reevaluating beryllium cancer risk as a part of the IRIS Pilot Project. The reevaluation is not undergoing internal peer review. A preliminary draft may be available to the public in June 1997 (Bayliss 1997; Bruce 1997).

Uncertainties in the Slope Factors

Slope factors are uncertain. The values used for this assessment are those recommended by the EPA in the IRIS Database (EPA 1998a). They were derived from a range of epidemiological data, which is summarized in Table 3. There are obvious limitations to developing values from the results of a single worker epidemiological study with confounding factors and limitations of its own. Uncertainties associated with the concentrations of beryllium in the workplace, duration of exposure, dosimetry, and other assumptions used in determining the unit risk values are discussed in EPA (1987b) but were not quantified.

The relative risk estimates were used to provide a probable range and central value rather than just a 95% confidence limit value. In the occupational epidemiological study on which the cancer potency determination was based, a range for median exposure of 100 to 1000 $\mu g\ m^{-3}$ was determined. Furthermore, an assumption was made that the ratio of exposure duration to years at risk ranged from 0.25 to 1.0. The mean of the potency factors derived using these assumptions was reported in IRIS (EPA 1998a). The maximum and minimum values (EPA 1987b) can be used to calculate a minimum and maximum SF. The maximum risk per microgram per cubic meter value of 7.16×10^{-3} corresponds to a SF of 25 kg-d mg^{-1} and the minimum risk per microgram per cubic meter value of 1.61×10^{-4} corresponds to a slope factor of 0.56 kg d

mg^{-1}. These values were used to approximate an uncertainty distribution for the SF assuming a triangular distribution with the most likely value being 8.4 kg d mg^{-1}.

It is important to note that EPA SF values are not used to determine true carcinogenic risk to an individual. Traditionally, these values have been used to screen contaminants, determine cleanup levels, or used in prospective assessments to show no impact. The risk values calculated in this report are not to be interpreted as actual carcinogenic risk to the selected receptors. Rather, the calculated risks represent upper bound estimates that are not expected to be exceeded for a given intake of beryllium.

ENVIRONMENTAL TRANSPORT MODELING

Offsite exposure to beryllium originating from RFP was investigated in Phase I and summarized in the previous sections. Airborne releases were considered to be the major pathway of exposure. There were insufficient data to estimate a source term for surface water releases, and environmental monitoring of reservoir sediments in Great Western Reservoir and Standley Lake showed beryllium concentrations similar to background levels found in other parts of the Rocky Mountain region. For these reasons only atmospheric transport of beryllium was considered.

Atmospheric releases of beryllium during routine operations at the RFP primarily occurred from two release points; roof vents on Buildings 776 and 444. Other minor release points were also identified in Phase I reports. In this section, we describe our approach to estimating atmospheric dispersion of beryllium for the years 1953–1988 and the uncertainty associated with concentration estimates in the model domain. Our approach to this calculation involves first estimating an annual average X/Q (concentration divided by source term [s m^{-3}]) for each receptor in the model domain. Concentrations for specific years of the assessment period are calculated by multiplying the annual quantity of beryllium released to the atmosphere by the X/Q value for a given receptor located in the model domain. Uncertainties in dispersion estimates are accounted for through multiplicative correction factors. Airborne concentrations are then used with exposure scenarios and the SFs to calculate risk for selected receptors in the model domain.

Atmospheric Model Selection

Five atmospheric transport models considered for use in this study were evaluated in Rood (1997): (1) the Terrain-Responsive Atmospheric Code (TRAC) (Hodgin 1991), (2) the Industrial Source Complex Short Term Version 2 (ISC) (EPA 1992), (3) Regional Atmospheric Transport Code for Hanford Emission Tracking (RATCHET) (Ramsdell et al. 1994), (4) TRIAD (Hicks et al. 1989), (5) and INPUFF2 (Petersen and Lavdas 1986). The purpose of the model comparison study was to determine what models, if any, performed best in the Rocky Flats environs for a given set of modeling objectives. These data along with other studies were used to established the uncertainty one might expect from a model prediction.

Model evaluations were based on how well predictions compared with measured tracer concentrations taken during the Winter Validation Tracer Study (Brown 1991) conducted in February 1991 at the RFP. The study consisted of 12 separate tests; 6 tests were conducted during nighttime hours, 4 during daytime hours, and 2 during day-night transition hours. For each test, an inert tracer (sulfur hexafluoride) was released in an open area near the southern RFP

boundary. The tracer was released at a constant rate for 11 hours from a 10 m high stack. Two sampling arcs, 8 and 16 km from the release point, measured tracer concentrations every hour for the last 9 hours of each test period. Seventy-two samplers were located on the 8-km arc, and 68 samplers were located on the 16-km arc. Predicted concentrations were then compared to the observed tracer concentrations at each of the samplers.

Modeling objectives for the comparison study were based on the premise that identifying locations of individual receptors on an hour-by-hour basis was unlikely. Instead, it was more likely to identify receptors (hypothetical or real) who were present at a fixed location for the duration of a release event. The minimum time scale of historical release events at RFP ranged from one to several days. Release events modeled for the Winter Validation Tracer Study were 9 hours in duration. If we assume the receptor is fixed for a time period of at least 9-hours, then the time-averaged concentration (9-hour average) is an appropriate modeling objective rather that comparing hourly average concentrations. Therefore, models were evaluated based on their performance in predicting time-averaged concentrations at fixed sampler locations in the model domain (9-hour average concentration at each sampler paired with the corresponding predicted value). We also considered the arc-integrated concentration. The arc-integrated concentration was the 9-hour average ground-level concentration integrated across the 8 and 16-km sampling arc. The latter performance objective provides a measure of the vertical dispersion component of the models and the ground-level tracer mass, 8 and 16 km from the release point. Data sets for the time-averaged concentration were limited to only those points where the predicted (C_p) and observed (C_o) concentration pair were greater than the time-averaged minimum detectable concentration.

Fifty percent of the time-averaged model predictions were within a factor of 4 of the observations. Predicted-to-observed ratios (C_p/C_o) ranged from 0.001 to 100 and tended to be higher at the 16-km arc than the 8-km arc. Geometric mean C_p/C_o ratios ranged from 0.64 (TRAC) to 1.5 (ISC), and geometric standard deviations ranged 4.4 (RATCHET) to 6.5 (ISC). The RATCHET model had the highest correlation coefficient for the 8-km (0.67) and 16-km (0.58) sampling arc followed by TRIAD and INPUFF2.

Arc-integrated results (Figure 3) showed INPUFF and TRIAD had the highest correlation coefficients, but correlation coefficients were not significantly different (at the 95% level) from the other models. Qualitatively, the predictions made by the RATCHET model appear to best match the observations. The slope of the regression line was closest to that of the perfect correlation line (solid line in Figure 3). The ISC model tended to overpredict arc-integrated concentration, and the TRAC model showed the greatest variability.

The results reported in Rood (1997) indicated no one model clearly outperformed the others. However, the RATCHET, TRIAD, and INPUFF2 models generally had lower variability (indicated by lower geometric standard deviations of C_p/C_o ratios) and higher correlation coefficients compared to those of ISC and TRAC models. It is desirable in a study such as this to choose a model that has the least amount of variability when comparing model predictions to observations. In addition, the model selected should have a level of complexity that is consistent with available data. The TRAC model is the most complex in terms of its treatment of the atmospheric dispersion process in complex terrain, but the study showed model performance was no better than the other models. In addition, the availability of meteorological data needed to fully use the capabilities of the TRAC model are lacking. The straight-line Gaussian plume model, ISC tended to overpredict concentrations and was also limited to only one meteorological

recording station in the model domain. Available meteorological data for this study period may include two meteorological recording stations; one at the RFP and the other at Denver Stapleton International Airport. Therefore, a model that may include multiple meteorological recording stations in the model domain is desirable. The use of multiple meteorological recording stations will allow for a spatially varying wind field in the model domain.

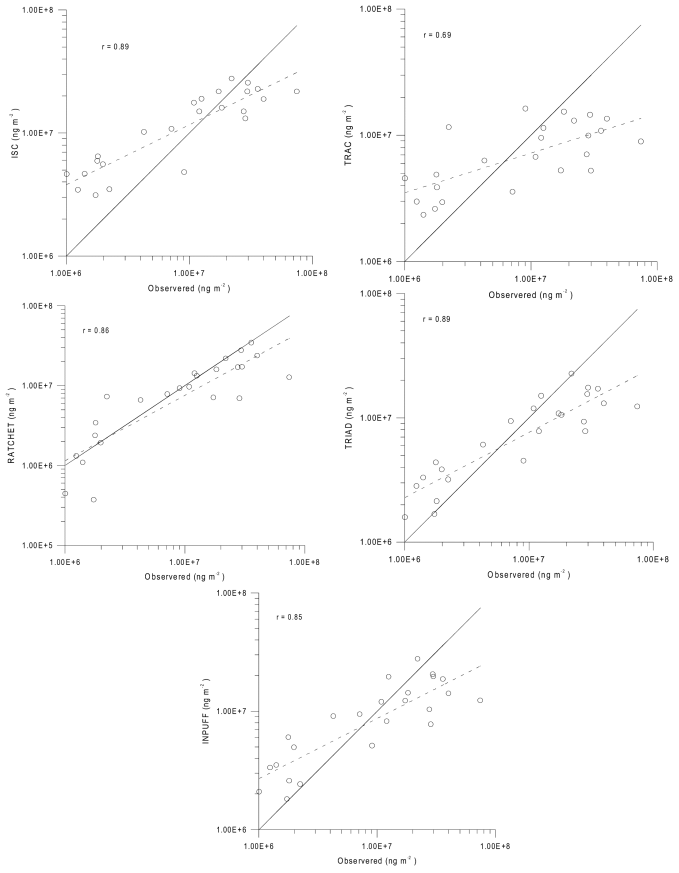

Figure 3. Observed arc-integrated concentration as a function of predicted values for the five models compared using the Winter Validation Data Set. Correlation coefficients were for the log-transformed data. The solid line represents perfect correlation between predicted and observed values. The dashed line represents the log-transformed regression fit.

The three models RATCHET, INPUFF2, and TRIAD performed comparably and were considered viable candidates for atmospheric dispersion estimates. We chose the RATCHET model for modeling routine releases of beryllium for the following reasons:

- The model was easily configured for long-term (annual average) dispersion estimates
- Spatial differences within the model domain are accounted for (i.e., surface roughness meteorology)
- Algorithms to compute plume depletion and deposition for fine particles are included (deposition must be computed outside the TRIAD and INPUFF2 codes)

- The model requires meteorological data in 1-hour increments, which are the same as those given for typical airport observations.

Corrections for model bias were made in the uncertainty analysis. Features of the RATCHET model are summarized in Table 4.

Table 4. Features of the RATCHET Model

Feature	Representation in RATCHET
Domain area[a]	2100 km^2
Node spacing[a]	2000 m
Source term	Hourly release rates
Meteorological data	Hourly
Surface roughness	Spatially varying
Wind fields	$1/r^2$ interpolation
Topographical effects	None explicit[b]
Wind profile	Diabatic
Stability	Spatially varying based on wind, cloud cover, and time of day
Precipitation	Spatially varying, three precipitation regimes with different precipitation rate distributions
Mixing layer	Spatially varying, based on calculated values for each meteorological station
Plume rise	Briggs' equations (Briggs 1969, 1975, 1984)
Diffusion coefficients	Based on travel time and turbulence levels
Dry deposition	Calculated using resistance model
Wet deposition	Reversible scavenging of gases, irreversible washout of particles
Model time step	15 minute maximum, 15 second minimum
Output frequency[c]	Daily
Uncertainty	Options available for Monte Carlo simulation within the code

[a] Modified from the original RATCHET specification for use at Rocky Flats.
[b] The model does not account for terrain elevation changes relative to the plume height explicitly. However, topographical influence on the wind field may be accounted for by incorporating multiple meteorological stations in the model domain.
[c] Modified to output annual average concentrations at user specified grid nodes.

Model Domain and Receptor Grid

The model domain (Figure 4) encompasses a 2200 km^2 area (50 km north-south by 44 km east-west). The domain extends 28 km south, 12 km west, 22 km north, and 32 km east from the RFP. Most of the Denver metropolitan area and the city of Boulder are included in the domain. The domain was limited in its western extent because few receptors are present there and most of the contaminant plumes traveled east and southeast of the plant.

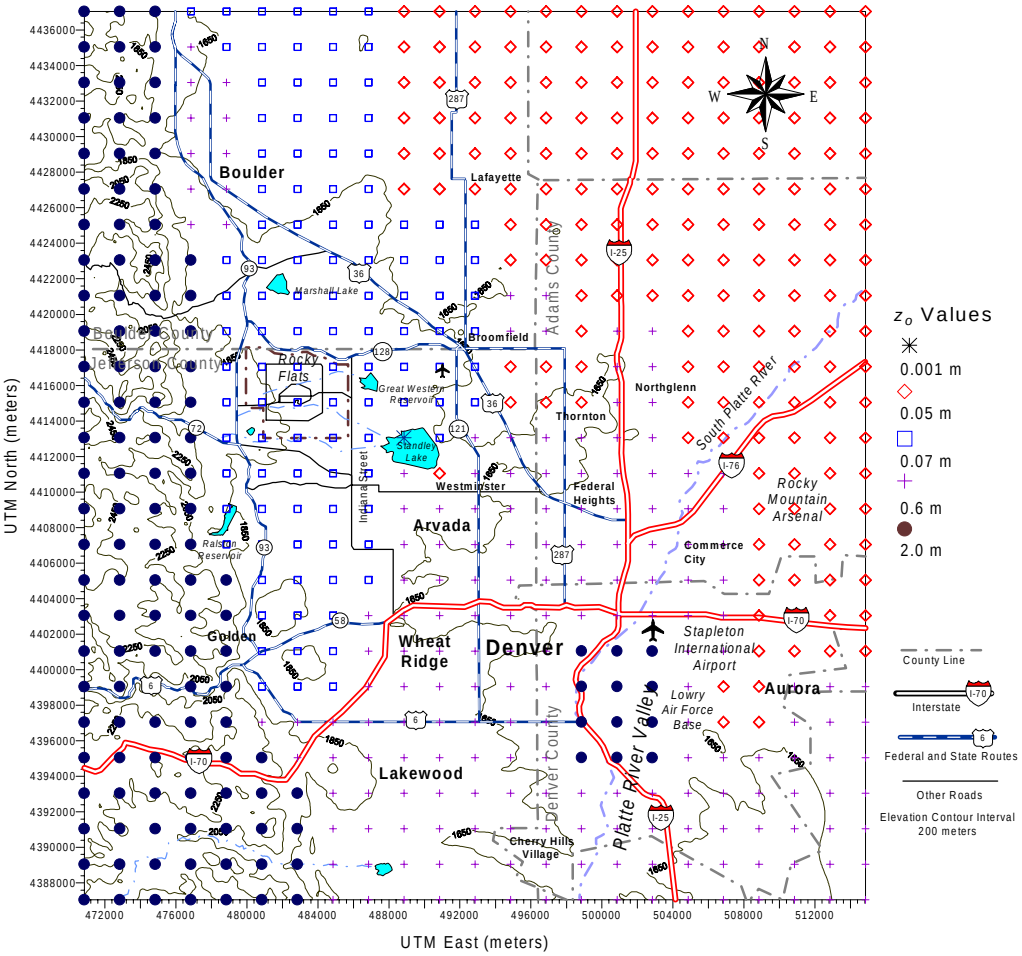

Figure 4. RATCHET environmental modeling grid and roughness length values (z_o). Symbols represent grid nodes and the z_o value assigned to the node.

RATCHET uses two modeling grids. Hourly meteorological records are used to estimate wind speed and direction, stability, and precipitation on the environmental grid in addition to surface roughness features. The concentration grid has spacing one-half that of the environmental grid. Ground-level concentrations and deposition are output at each of these grid nodes. The environmental grid was set at 23 nodes east-west and 26 nodes north south with a grid spacing of 2000 m. The concentration grid has 45 nodes east-west and 51 nodes north-south with a spacing of 1000 m. The southwest corner of the model domain has the universal transverse mercator (UTM) coordinates 470850 E and 4387050 N. Release points are defined by distances (in kilometers) from a reference node. The reference node for the environmental grid was (7,15) and (13,29) for the concentration grid and both have the UTM coordinates of 482850 E and 4415050 N.

Figure 4 was generated using USGS 7.5-minute digital elevation models. Topographic contours were based on an elevation grid spacing of 100 m. Major roadways were digitized from USGS 1:100,000 digital line graphs.

Meteorology

Meteorological data for the operational period of Rocky Flats (1952–1988) are sporadic, incomplete, and of questionable integrity. Requests for meteorological data from the RFP were initially made by ChemRisk during Phase I of the project. ChemRisk was able to locate two letters from Dow Chemical to Dr. Roy Cleare, Executive Director of the Colorado Department of Health, dated March 20, 1970, that contained wind speed and direction for varying time increments during the 1957 and 1969 fire incidents. Computer diskettes containing wind speed, wind direction, and precipitation measurements from October 1968 to May 1969 were also obtained. These data were hourly observations taken approximately 15 minutes before the top of the hour and do not represent hourly average readings. Although these data appeared to be climatologically reasonable, no records of instrument calibration or audits of the information were found. Parameter resolution was very coarse (for example, wind direction resolution was 45 degrees). Five years (1987–1991) of high quality meteorological data taken at the 61-m tower at RFP were obtained and used by ChemRisk in Phase I of this project for predicting annual average concentrations from routine releases.

An extensive data search was initiated in 1994 by *Radiological Assessments Corporation (RAC)* researchers to locate missing data and interview personnel who were involved with measurements at the site. No new data were recovered, but several personnel reported problems with the recording instrumentation at the RFP, such as the measured wind direction being off by 180 degrees. Other data recorded from nearby Jefferson County Airport (about 8 km east of the plant) were obtained for the years 1968–1971. These data were only reported for the hours while the airport was open (06:00–23:00 local standard time) and were instantaneous measurements and not hourly averages as was typical of all airport data before the Automatic Surface Observation Site (ASOS) system was installed at most major airports. In 1994, the RFP hired a subcontractor to compile, screen, validate, and analyze historical climatological data (DOE 1995c). A draft report was issued in February 1995; the report contained monthly and annual summaries of wind speeds, wind directions, precipitation, temperature, and other parameters for the years 1953–1993. While these data are of interest and may be important for some aspects of modeling, they lacked the resolution required for detailed atmospheric transport modeling.

We concluded that meteorological data taken during the time the RFP was operating were incomplete, unreliable, and unsuitable for atmospheric transport modeling during the period 1952–1988. However, surrogate data spanning a different time period can be used to make annual average dispersion estimates for past releases. We used this approach in our modeling effort.

For our modeling effort, we used meteorological data spanning a 5-year period (1989–1993) taken at two recording stations located at the RFP and Denver Stapleton International Airport. How representative this 5-year data set is for earlier time periods is discussed in the uncertainty section. Federal regulations have stated that a 5-year database is adequate for predicting annual-average air quality impacts at a site (CFR 1996). Meteorological data from RFP were taken at the 10-m level from the 61-m tower located on the south side of the plant complex at UTM coordinates 482064 E 4414963 N. Data recorded at this station included wind speed, wind direction, temperature, and other parameters (heat flux and standard deviation of wind direction) that were not used in these simulations. The Denver Stapleton International Airport meteorological station was located 24 km east and 14 km south from the center of the model

domain (RFP). These data included hourly measurements of wind speed, wind direction, cloud cover, and precipitation. It is known that meteorological conditions in the Denver metropolitan area can differ significantly from those at Rocky Flats (DOE 1980). Therefore, it is unreasonable to use meteorological data from Denver alone for simulations involving releases from Rocky Flats. In these simulations, initial plume trajectories are primarily influenced by the wind direction at Rocky Flats. Only after plume elements are transported to the Denver metropolitan area are trajectories and dispersion influenced by meteorological conditions present there.

Data Processing

Meteorological data from 1989–1993 were obtained in electronic format from the Rocky Flats meteorologist. These data were measured at a height of 10 and 61-m from a 61-m tower located at RFP. Only data from the 10-m level were used in the simulations. Each record represented the average over a 15-minute recording period and included wind speed and direction, temperature, heat flux, and standard deviations of these parameters. Processed data suitable for use in EPA's ISC code were also obtained for the same time frame. These data included stability class estimated by the lateral turbulence and wind speed method (standard deviation of the horizontal wind direction fluctuations) as described in EPA (1987c) and mixing height estimates. The mixing heights were derived from linear interpolation for each 15-minute period from the rawinsonde data furnished routinely every 12 hours by the National Weather Service for Denver Stapleton International Airport. These data were used as default mixing-layer depths in RATCHET. Mixing-layer depths are calculated hourly within RATCHET at each active meteorological recording station using a methodology described by Zilitinkevich (1972). The calculated or default value is selected on the basis of the relative magnitude of the calculated and default values, the stability, season, and time of day. The larger of the two is selected for the meteorological recording station for the given hour. A multiple linear regression technique is then used to provide a smooth spatial variation in mixing-layer depth across the model domain.

Stability classes were calculated separately for the RFP and Denver Stapleton International Airport meteorological recording stations using the general classification scheme discussed in Pasquill (1961), Gifford (1961), and Turner (1964). This typing scheme employs seven stability categories ranging from A (extremely unstable) to G (extremely stable) and requires estimates of sky cover and ceiling height. Cloud cover and ceiling height data for both stations were assumed to be the same and were obtained from the Denver Stapleton International Airport data.

Hourly average wind speed and direction also were calculated from the raw RFP meteorological data using the protocol described in EPA (1987c). An arithmetic average of the wind direction was computed first, and it was then segregated into 1 of 36, 10-degree increments as required by RATCHET. The average wind speed for the hour was computed by taking the average of the four, 15-minute data segments. Hourly precipitation records from Denver Stapleton International Airport were assumed to be consistent over the entire model domain and were segregated into integer values as required by RATCHET (see Table 6).

Atmospheric Transport Model Parameters

This section describes the input parameters that were selected for the RATCHET model for simulations involving normal operational releases. These parameters include surface roughness length, topography, dry and wet deposition, diffusion coefficients, release parameters (location and height of release), and model control parameters (number of puffs per hour and computational options).

Surface Roughness Length

Roughness elements (such as trees and buildings) and small-scale topographic features (such as rolling hills) have a frictional effect on the wind speed nearest the surface. The height and spacing of these elements will determine the frictional effects on the wind. These effects are directly related to transport and diffusion and affect atmospheric stability, wind profiles, diffusion coefficients, and the mixing-layer depth. The surface roughness length parameter is used to describe these roughness elements and is a characteristic length associated with surface roughness elements (Table 5). In RATCHET, estimates of the surface roughness length are defined for each node on the environmental grid (Figure 4). In our simulations, we selected a value of 0.6 m to represent residential and urban environs. Farmland, which is predominant in the northeast part of the model domain, was assigned a value of 0.05 m. Range and open land consisting of rolling grass hills were assigned a value of 0.07 m. Nodes that encompass the range and farmland designation were selected based on the topographic contours and land use maps. The foothills and downtown Denver were assigned a value of 2.0 m and open water (Standley Lake) was assigned a value of 0.001 m.

Table 5. Typical Surface Roughness Lengths for Different Land Use, Vegetation, and Topographic Characteristics[a]

Land use, vegetation, and topographic characteristics	Surface roughness length, z_o (m)
Level grass plain	0.007–0.02
Farmland	0.02–0.1
Uncut grass, airport runways	0.02
Many trees/hedges, a few buildings	0.1–0.5
Average, North America	0.15
Average, U. S. Plains	0.5
Dense forest	0.3–0.6
Small towns/cities	0.6–2.5
Very hilly/mountainous regions	1.5+

[a] Source: Stull (1988), Figure 9.6

Topography

The RATCHET model does not explicitly address terrain differences within the model domain. Instead, topography and topographic effects on transport and diffusion are reflected in the surface roughness lengths and observed wind velocity data that are affected by topographical features. Topography in the model domain (Figure 4) can be characterized by three major features: the north-south trending Colorado front range foothills in the western part of the model domain, the southwest to northeast trending Platte River Valley located in the southeast part of the model domain, and rolling hills and flat farmland that is predominant in the central and northeastern part of the model domain. The surface roughness lengths reflect these features as stated in the previous section. Observed meteorological data are lacking in most of the model domain and are woefully inadequate to characterize wind fields in the foothills region. However, meteorological observations at Denver Stapleton International Airport do capture the air movement within the Platte River Valley, which is noticeably different than that at the RFP (DOE 1980). Therefore, to a limited extent, topography is accounted for the model simulation. The use of a complex terrain model would also suffer from the lack of meteorological data, especially in the foothills region. This region may be of little importance because few receptors were present in the foothills when the plant was operating.

Dry and Wet Deposition

The rate of deposition of small particles on surfaces in the absence of precipitation is proportional to the concentration of material near the surface. The proportionality constant between the concentration in air and the flux to the ground surface is the dry deposition velocity. The current generation of applied models estimates deposition using an analogy with electrical systems as described by Seinfeld (1986). The deposition is assumed to be controlled by a network of resistances, and the deposition velocity is the inverse of the total resistance. Resistances are associated with atmospheric conditions; physical characteristics of the material; and the physical, chemical, and biological properties of the surface. The total resistance in RATCHET is made up of three components: aerodynamic resistance, surface-layer resistance, and transfer resistance. Thus, the dry deposition velocity (v_d, m s^{-1}) is calculated using

$$v_d = (r_s + r_a + r_t)^{-1} \quad (2)$$

where

r_s = surface layer resistance (s m^{-1})
r_a = aerodynamic resistance (s m^{-1})
r_t = transfer resistance (s m^{-1}).

Surface layer resistance and aerodynamic resistance are given by

$$r_a = U(z)/u_*^2 \quad (3)$$

$$r_s = 2.6/(0.4\ u_*) \quad (4)$$

respectively where u_* = frictional velocity (m s^{-1}), and $U(z)$ = wind speed (m s^{-1}) measured at height z (m) above the ground. The frictional velocity is given by

$$u_* = \frac{U(z)\,k}{\ln\left(z / z_o\right) - \psi(z / L)} \tag{5}$$

where k = the von Karman constant (0.4), z_o = surface roughness length, ψ = stability correction factor, and L = the Monin-Obukhov length (m). The transfer resistance is associated with the characteristics of the depositing material and surface type. In RATCHET, the transfer resistance is used as a mathematical means to place a lower limit on the total resistance. As the wind speed increases, r_s and r_a become small resulting in unreasonably high deposition velocities. For small particles (<1.0 μm), a transfer resistance of 100 s m^{-1} is suggested in RATCHET, and it results in calculated deposition velocities that are consistent with measured data. Harper et al. (1995) estimates deposition velocities for 1 μm particles and 5 m s^{-1} wind speed to range from 1.0×10^{-2} (5th percentile) to 4.1 cm s^{-1} (95th percentile). The RATCHET calculated values assuming a roughness length of 0.05 m and a transfer resistance of 100 s m^{-1} ranged from 0.66 to 0.75 cm s^{-1}, which is in the range of measured values. Effluent containing beryllium was reported to pass through HEPA filtration resulting in release of particle less than 1 μm in diameter. Median particle size for plutonium effluent subject to the same HEPA filtration has been estimated to be 0.3 μm (Voillequé 1997). We have assumed the beryllium effluent had the same particle size distribution as the plutonium effluent.

Gravitational settling (v_t) is not included in Equation (2) but may be added. However, for small particles (~1.0 μm), gravitational settling is negligible compared to r_s and r_a. Stokes law gives the gravitational settling velocity for particles less than 20 μm as

$$v_t = \frac{C_c\ d^2\ g\rho}{18\mu_{air}} \tag{6}$$

where

C_c = the Cunningham slip correction factor (dimensionless)
d = particle diameter (cm)
g = gravitational acceleration constant (980 cm s^{-2})
μ_{air} = dynamic viscosity of air (1.78×10^{-4} g s^{-1} cm^{-2})
ρ = particle density (1.85 g cm^{-3} for beryllium).

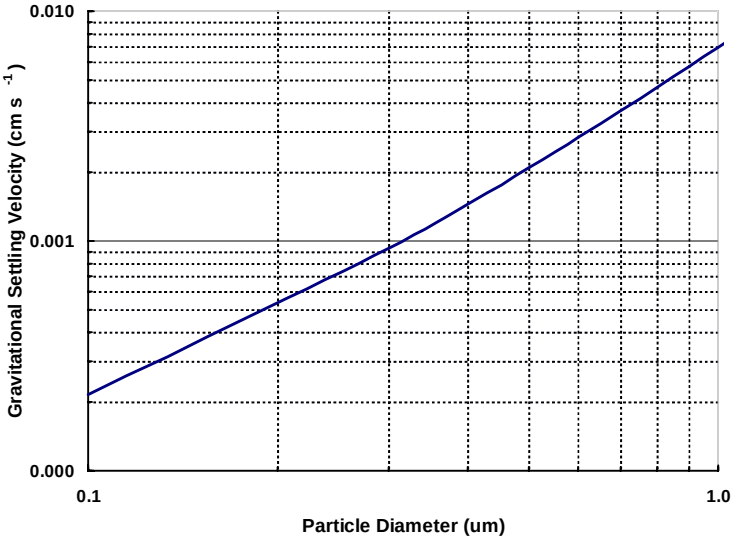

Figure 5. Gravitational settling velocity as a function of particle diameter for beryllium (ρ = 1.85 g cm^{-3}).

For particle sizes less than several microns, the Cunningham Slip correction factor is approximately 1.0. Figure 5 presents gravitational settling velocity as a function of particle size. Whicker and Schultz (1982) report gravitational settling velocities for particles less than 1 μm are insignificant compared to the other components of deposition. Deposition velocities calculated using Equation (2) ranged from 0.3 to 1.0 cm s^{-1}, for wind speeds ranging from 2.5 to 20 m s^{-1}, roughness lengths from 0.001 to 2 m, and a transfer resistance of 100 s m^{-1}. Note that the gravitational settling velocity for 0.3 μm particles ($\approx$0.001 cm s^{-1}) is insignificant compared to the deposition velocity calculated with Equation (2). For our simulations, gravitational settling was ignored and a transfer resistance of 100 s^{-1} m was used.

Wet deposition of small particles in RATCHET is modeled using a washout coefficient and assuming irreversible collection of particles as the precipitation falls through the puffs. The following expression discussed in Slinn (1984) is used to compute the washout coefficient in RATCHET:

$$\Lambda = \frac{C\,E\,P_r}{0.35\,P_n^{1/4}} \qquad (7)$$

where

Λ = washout coefficient (hr^{-1})
C = empirical constant assumed to have a value of 0.5
E = average collision efficiency assumed to be 1.0
P_r = precipitation rate (mm hr^{-1})
P_n = normalized precipitation rate (P_r)/ [1 mm hr^{-1}]).

The normalized precipitation rate is a dimensionless quantity that represents the precipitation rate normalized to 1 mm h^{-1}. During periods of snow, the washout coefficient for particles is computed by

$$\Lambda = 0.2\ P_r. \tag{8}$$

Precipitation rates in RATCHET are separated into six classes: three for liquid and three for frozen precipitation (Table 6). These classes are the similar to those reported by most airport meteorological recording stations.

Table 6. Precipitation Rates and Washout Coefficients Used in RATCHET

Precipitation type	Precipitation rate (mm hr^{-1})	RATCHET precipitation code	Washout coefficient (hr^{-1})
No precipitation	0.0	0	0.00
Light rain	0.1	1	0.254
Moderate rain	3.0	2	3.26
Heavy rain	5.0	3	4.78
Light snow	0.03	4	0.006
Moderate snow	1.5	5	0.3
Heavy snow	3.3	6	0.66

Diffusion Coefficients

The RATCHET model estimates the diffusion coefficients directly from statistics for atmospheric turbulence. In most cases, the statistics describing atmospheric turbulence (i.e., standard deviation of the horizontal and vertical wind direction fluctuations) are not routinely measured at most meteorological recording stations. However, RATCHET makes use of atmospheric conditions that are either measured or calculated from routine meteorological data to estimate the turbulence statistics. The parameters wind speed, atmospheric stability, and surface roughness are used to estimate the turbulence statistics. The general form of the equation used in RATCHET for estimating the horizontal diffusion coefficient (σ_r) for the first hour following release is

$$\sigma_r = 0.5\sigma_v\ t \tag{9}$$

where σ_v = crosswind component of turbulence (m s^{-1}) and t is the travel time. After the first hour, the horizontal diffusion coefficient is given by $\sigma_r = c_{sy}$ t where c_{sy} is a proportionality constant with dimensions of meters per second. Gifford (1983) has shown the value of c_{sy} distributed between 0.14 to 1.4 with a median value of 0.5. For our simulations, we used the median value of 0.5.

The general form of the equation for estimating the vertical diffusion coefficient (σ_z), near the source is

$$\sigma_z = \sigma_w\ t\ f_z(t) \tag{10}$$

where

σ_w = standard deviation of the vertical component of the wind (m s^{-1})

$f_z(t)$ = nondimensional function related to the travel time and turbulence time scale.

As a practical matter, diffusion coefficients in RATCHET are calculated in increments to avoid problems associated with spatial and temporal changes in conditions.

Source Characterization

Estimated releases of beryllium to the atmosphere were provided by ChemRisk (1994a) and are summarized in a previous section. Twenty-five percent of the beryllium released to the atmosphere was attributed to Building 444 and 19% was attributed to Building 776 (ChemRisk 1992). Building 444 contained the beryllium foundry where machining, casting, and milling of beryllium occurred. Beryllium milling and machining did not occur in Building 776, but some materials containing beryllium were processed. Therefore, beryllium was monitored in the plenum exhaust. Plenum exhaust was passed through HEPA filtration before being released to the atmosphere. The remaining 54% of the atmospheric beryllium releases originated from 11 other buildings surrounding Building 444 and Building 776. In Phase I, ChemRisk modeled the combined release from all buildings using a virtual stack located approximately in the center of the plant. We have modeled the combined releases to originate from two points: Building 776 and Building 444. Combined releases were proportioned between the two buildings based on the relative contribution each building had to their combined total. Therefore, the proportion from Building 444 was $0.25/(0.25 + 0.19) \approx 0.6$ or 60% and the remainder (40%) was proportioned to Building 776 (Table 7). Releases from Building 776 were reported to originate from five roof vents. The roof vents were hooked shaped and directed flow down toward the top of the roof. Therefore, the modeled release height was the height of the building. The building height was 11.6 m and the horizontal dimensions were 61 × 104 m. The vents were assumed to be distributed across the roof resulting in an area source geometry. The area source was simulated by modifying the initial diffusion coefficients using a procedure described by Petersen and Lavdas (1986). The initial horizontal diffusion coefficient (σ_r) is the horizontal dimension of the source divided by 4.3, and the initial vertical diffusion coefficient (σ_v) is the height of the source divided by 2.15. For these simulations, we used the 61-m length as the horizontal source dimension.

Atmospheric releases of beryllium from Building 444 originated from vent number 122 after passing through two stages of HEPA filtration (ChemRisk 1992). Flow rates, stack heights, and release velocities were not characterized by ChemRisk. For this analysis, we have assumed the release to occur from a point source on the roof (4.5 m) with no buoyant or momentum driven plume rise.

Table 7. Release Parameters for Building 776 and Building 444

Release point	Parameter	Value
Building 444	Stack height	5 m
	Stack diameter	2.0 m
	Flow rate	1.6 m^3 s^{-1}
	UTM east	482372 m
	UTM north	4414850 m
Building 776 roof vents	Release height	11.6 m
	Initial σ_r	14.1 m
	Initial σ_v	5.4 m
	UTM East	482938 m
	UTM North	4415879 m

Stack tip downwash is also modeled in RATCHET; however, building wake is ignored. At distances of about 2 km, building wake has been shown to have little effect on measured atmospheric concentrations (Start et al. 1980). Ramsdell (1990) showed that for ground-level releases, modeled air concentrations greater than 1 km from the source are relatively unaffected by building wakes. Note the nearest receptor is >3 km from Building 776.

Uncertainty associated with the source term estimates were estimated by ChemRisk (1994a, Appendix G) and were used without modification in this report. Uncertainty was represented by a multiplicative correction factor. For releases that occurred before to 1971, a lognormally distributed correction factor having a GM of 1.9 and a GSD of 2.0 was applied to the source term. For releases during the 1971–1988 time frame, a lognormally distributed correction factor having a GM of 1.4 and GSD of 1.9 was applied to the source term.

Other Parameters

Several other parameters in RATCHET influence the accuracy of output and computer runtime. These parameters include the number of puffs per hour, minimum time step, puff consolidation, maximum puff radius, and minimum puff concentration at center. We chose the suggested RATCHET default values for all these parameters except minimum time step and minimum concentration at puff centers (Table 8). Accuracy of the simulation can be improved by using a smaller time step. The RATCHET default was 20 minutes, which we reduced to 10 minutes. The minimum concentration at puff centers was reduced from 1×10^{-13} to 1×10^{-15} to allow for plume tracking throughout the model domain. The puff consolidation parameter value combines puffs from the same source when ratio of the puff centers to the average σ_r is less than the user-input value. The puff consolidation ratio and maximum puff radius (in units of σ_r) were set at RATCHET default values of 1.5 and 3.72, respectively.

Prediction Uncertainty

We are interested in defining the expected uncertainty in the annual average dispersion estimates within the model domain for each year of the assessment period (1952–1988). The approach used in this assessment to define prediction uncertainty was to develop distributions of

multiplicative correction factors that were applied to each deterministic concentration in the model domain. These multiplicative correction factors were developed from field validation data, joint frequency distribution comparisons, and parametric uncertainty analysis. Three components of uncertainty were evaluated:

1. Dispersion uncertainty
2. Meteorology uncertainty
3. Plume depletion uncertainty.

Table 8. RATCHET Model Control Parameters

Model parameter	Value
Number of puffs per hour	4
Minimum time step	10 minutes
Puff consolidation	1.5
Maximum puff radius (in units of σ_r)	3.72
Minimum concentration at puff centers	1×10^{-15}

Dispersion uncertainty considers the uncertainty in predicting the annual average concentration of an inert, non-reactive tracer for a specific year, assuming we have the meteorological data for that year. Meteorology uncertainty arises because we are using 5-years of meteorological data spanning a recent time period (1989–1993) to calculate an annual average *X/Q* value (concentration divided by release rate) that will be applied to all previous years (1952–1988) of the assessment period. Uncertainty in plume depletion via dry deposition was considered separately because dispersion uncertainty was based on tracer studies that typically employ inert, non-reactive tracers that have dry deposition velocities that are small and inconsequential. Uncertainty in plume depletion from wet deposition was not considered.

Dispersion Uncertainty. Dispersion uncertainty includes two sources: (1) errors in model input and (2) errors in model formulation or in the model itself (i.e., does the model adequately represent the physical process and phenomena it attempts to simulate). For example, suppose we select a location in the model domain and measure the concentration of tracer released from the site for an entire year. Let us assume the uncertainty associated with the measurement is small and inconsequential. Using the meteorological data recorded for that year, we calculate a concentration at the same receptor location using an appropriate atmospheric dispersion model. Assuming our model adequately represents the physical process and phenomena (i.e., if we had the correct inputs to the model, the output would match the observations), the uncertainty associated with the model prediction results from a lack of knowledge about the correct inputs to our model. Propagating these of uncertainties through the model calculation provides a distribution of model output. This is termed parameter uncertainty. The output distribution may be compared with measured data to see if model predictions encompass the measurements. Generally, agreement between predictions and observations is achieved when the model adequately represents the processes it attempts to simulate and choices regarding input parameter values have been made correctly.

Model uncertainty arises from the fact that perfect models cannot be constructed, and models often fail to adequately represent the physical process they attempt to simulate. In atmospheric dispersion models, the advection-dispersion process is often oversimplified and meteorological

data required to characterize turbulence in the environment are lacking. In our previous example, the parameter uncertainty may not account for all differences between model predictions with observations if our model does not perfectly represent the physical process. Field validation exercises provide some information as to the overall performance of a model and in turn, model uncertainty. However, these are only partially relevant because field tests are generally not conducted under the same conditions that actual releases occurred.

The RATCHET model incorporates modules to explicitly assess parameter uncertainty. These parameters include wind direction, wind speed, atmospheric stability class, Monin-Obukhov length, precipitation rate, and mixing-layer depths. Other parameters may be assessed by simply varying the input according to some predefined distribution and repeating the simulation a number of times until an adequate output distribution is achieved. These methods are both time consuming and computationally intensive and fail to capture model uncertainty. In our approach, we ignored the built-in parameter uncertainty in RATCHET and focused our efforts on defining the distribution of a correction factor that will be applied to model output. (Parameter uncertainty was only used to evaluate uncertainty in plume depletion and deposition.) The correction factor was based on field experiments, considering the relevance of the experiment to actual release conditions and model domain environs. In this approach, we have ignored the mass balance features of RATCHET and have instead, treated model output like that of a straight-line Gaussian Plume model. The only difference being that plume trajectories are not limited to straight lines.

We begin the process of defining the distribution of the correction factor for dispersion uncertainty by reviewing some field studies considered relevant to the assessment question (Table 9) which is what is the annual average concentration for each year of the assessment period. The correction factor is defined as the inverse of the distribution of predicted-to-observed ratios $[1/(C_p/C_o)]$. Relevant field studies included a model evaluation using the Rocky Flats Winter Validation Tracer Study data set (Rood 1997), validation exercises for RATCHET performed at the Hanford Reservation (Ramsdell et al. 1994), summaries of model validations performed for the Gaussian plume model (Miller and Hively 1987), and other studies reported in the literature. No one study is entirely relevant. Averaging times, release conditions, meteorological conditions, and terrain conditions are different than what we are attempting to simulate in this study. Nevertheless, these are the data we have chosen to work with and it is unlikely we will find a field validation experiment that was conducted under the exact conditions of past releases at Rocky Flats. Uncertainty bounds may be expanded to compensate for our lack of knowledge.

An additional study (Carhart et al. 1989) not reported in Table 9 included puff dispersion models that were similar to RATCHET (MESOPUFF, MESOPLUME). Evaluations were performed using tracer data bases from Oklahoma and the Savannah River Site. Oklahoma data consisted of two experiments measured at 100 and 600-km arcs downwind of a 3-hour perfluorocarbon release. The Savannah River data involved 15 separate experiments, 2 to 5 days in duration, where ^{85}Kr was released from a 61-m stack and measured at points 28 to 144-km downwind from the source. The ratio of the *average* predicted concentration to the *average* observed concentration was between 0.5 and 2. Note that this measure is different than the distribution of individual predicted-to-observed ratios reported in Table 9. There was also a tendency for models to overpredict concentrations in both data sets.

Table 9. Geometric Mean and Geometric Standard Deviation of Predicted-to-Observed Ratios for Field Studies Relevant to Defining the Correction Factor for Annual Average Concentrations

Model	Averaging time	Receptor distance	Release height	Environment	GM	GSD	Comments
RATCHET[a]	9-hour	8 km	10 m	complex terrain	0.86	4.4	Rocky Flats Winter Validation Study
RATCHET[a]	9-hour	16 km	10 m	complex terrain	1.1	4.3	Rocky Flats Winter Validation Study
RATCHET[b]	28-day	20–80 km	61 m	flat	1.4	2.2	Conducted at the Hanford Reservation
Gaussian Plume[c]	short-term	10 km	ground level	flat - highly instrumented		1.1	P/O ratios ranged from 0.8 to 1.2
Gaussian Plume[c]	short-term	10 km	elevated	flat - highly instrumented		1.2	P/O ratios ranged from 0.65 to 1.4
Gaussian Plume[c]	short -term	—	—	complex terrain		14	P/O ratios ranged from 0.01 to 100
Gaussian Plume[c]	annual average	—	—	complex terrain		3.8	P/O ratios ranged from 0.1 to 10
Gaussian Plume[c]	annual average	10 km	ground-level	flat		1.5	P/O ratios ranged from 0.5 to 2
Gaussian Plume[c]	annual average	10–50 km	ground-level	flat		2.2	P/O Ratios ranged from 0.25 to 4
Gaussian Plume[d]	12-hour	1–5 km	60 m	relatively flat	0.82	3.4	Terrain heights varied by about 50 m
Gaussian Plume[d]	72-hour	1–5 km	60 m	relatively flat	0.67	2.1	Terrain heights varied by about 50 m
Eulerian and Gaussian Plume[e]	annual average	1–1000 km	0–60 m	relatively flat	0.75	1.5	Gaussian model used for receptors out to 50 km
CTDMPLUS[f]	12 to 72 hour	1 km	—	complex terrain	1.6	2.5	EPA complex terrain model

a Rood (1997).

b Ramsdell et al. (1994).

c Miller and Hively (1987).

d Robertson and Barry (1989).

e Simpson et al. (1990).

f Genikhovich and Schiermeier (1995).

The study considered most relevant to the assessment question involved the RATCHET model using the Winter Validation Tracer Study data set. While it is true the release conditions for this study differed from those modeled (i.e., point source and area source) and the averaging time differed (i.e., annual average as opposed to 9-hour average), these data were obtained in the same environs that we are attempting to simulate. In addition, impacts on predicted and observed concentrations because of specific release conditions tend to diminish with increasing receptor distance. Release heights are not that much different from the Winter Validation Tracer Study in which the tracer was released at 10 m above ground level. Abbott and Rood (1996) also showed

that the difference between a point and a 100-m diameter area source (represented by a series of point sources distributed in a circular area) released from a height of 0–19 m is at most 5% along the plume centerline at a distance of 2 km or greater for all combinations of wind speed and stability. We conclude that the major difference between the Winter Validation Data set and our current situation resides with the averaging time.

The largest range of predicted-to-observed ratios reported in Table 9 involved complex terrain, which suggests models are more sensitive to the local meteorological and terrain conditions than other factors such as release height. For example, note the GSD for short-term estimates using the Gaussian plume model at a highly instrumented site for elevated source increases by about 9 % from its ground-level counterpart but the difference between the GSD for flat and complex terrain is almost an order of magnitude.

With the distribution of predicted-to-observed ratios for RATCHET from the Winter Validation Tracer Study as our starting point, our approach was to modify this distribution based (a) on the differences between the study conditions and those of past releases and (b) our assessment question (i.e. What is the annual average concentration for each year of the assessment period?). We combined data points at the 8 and 16-km distance into a composite set and justified this action based on the evaluations in Rood (1997) that showed similar GM and GSD values for 8 and 16-km data. In addition, the confidence intervals on the geometric mean and variance of the observed-to-predicted ratio overlapped. The composite distribution had a GM of 0.95 and GSD of 4.4. Predicted to observed ratios are plotted as a function of the number of standard deviations from the mean (normalized to the standard normal distribution) in Figure 6. Note that most of the data points ($\pm 2\sigma$) lie along the line representing the lognormal fit to the data, with the exception of the tails. We, therefore, represent the distribution of predicted-to-observed ratios as a lognormal distribution with a GM and GSD as defined above. Points on the tails, particularly those with predicted-to-observed ratios less than 0.01, were associated with Test 5 (February 9, 1991) at the 8-km arc in the east northeast–NE sector for the hours 16:00 to 18:00. All models performed poorly for this test. Concentrations in east northeast sector were grossly underestimated (greater than a factor of 10 difference) and the ground-level contaminant mass at 8 km was also underestimated. Models appeared to have difficulty responding to the transition from daytime to nighttime stability conditions. During the latter hours of the test and under predominately nighttime conditions (18:00–23:00), predicted concentrations showed better agreement with the observations.

As stated previously, the major difference between the Winter Validation Tracer Study data and the assessment question is the averaging time. Averaging time appears to have a large impact on the range of predicted-to-observed ratios encountered. For example, Simpson et al. (1990) reports the GSD of the predicted-to-observed ratio is reduced 38% with an increase in averaging time from 12 to 72 hours (Table 9). Also note the GSD for the annual average and short-term predicted-to-observed ratio for the Gaussian plume model under complex terrain conditions increases from 3.8 to 14. Validation exercises performed with RATCHET at the Hanford Reservation for an elevated release at distances greater than 20 km showed a slight overprediction by the model (GM = 1.4) and a GSD value of 2.2, which is about 50% smaller than the GSD for the Winter Validation Tracer Study data. It is not clear whether these differences are due to averaging time, release height, terrain conditions, or receptor distance, but based on the other studies reviewed in Table 9, it is likely that the smaller GSD is primarily due to increased averaging time.

Key observations relevant to defining the distribution of the correction factor are summarized as follows:

- GSD of predicted-to-observed ratios decrease with increasing averaging time
- GSD of predicted-to-observed ratios increase with increasing terrain complexity
- GSD of predicted-to-observed ratios increase for receptor distances >10-km
- GM of predicted-to-observed ratios are greater than 1.0 for receptor distances >20 km.

The GSD is expected to fall somewhere between 1.2 and 4.8 based on the data in Table 9. Noting the key observations stated above and the data in Table 9, the following values for GM and GSD were assigned to the predicted-to-observed ratio:

- GSD=2.2 and GM=0.95 for receptors <8 km
- GSD=2.0 and GM=0.95 for receptors >8 km and <16 km
- GSD=2.2 and GM=1.0 for receptors >16 km.

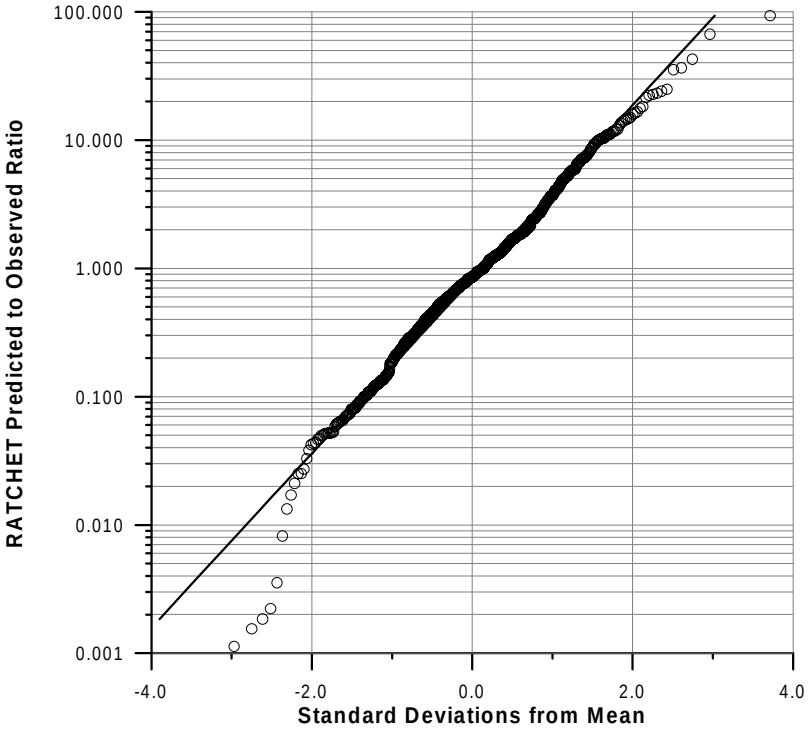

Figure 6. Predicted-to-observed ratios for the RATCHET model as a function of standard deviation from the mean (normalized to a mean of 0 and standard deviation of 1). The solid line represents the lognormal fit to the distribution. Circles represent individual data points.

The distribution of predicted-to-observed ratios translate into dispersion correction factors listed in Table 11 in the summary section. The GSD value of 2.2 was the same value calculated for monthly averages using RATCHET at the Hanford Reservation. It may be argued that a lower value is more appropriate because the averaging time is longer. We have chosen this value because the GSD of monthly average predicted-to-observed ratios will likely be higher for Rocky Flats compared to Hanford because of terrain complexities. In addition, no annual average

predicted-to-observed ratios exist for the Rocky Flats environs. Therefore, uncertainty bounds should be kept large to account for our lack of knowledge. Adjustments in the GSD and GM were also made to account for receptor distance. The GSD was reduced from 2.2 to 2.0 for receptors 8 to 16-km from RFP because the Winter Validation Tracer Study measurements were made at these distances and the lower value reflects our greater confidence in uncertainty at these distances. The GM was held at the same value calculated with the Winter Validation Tracer Study data for receptor distances <16 km and increased to 1.0 for receptor distances >16 km. The GM value was increased to reflect the tendency for models to overpredict at greater distances. Validation studies indicate predicted-to-observed ratios greater than 1.0 (reflecting model overprediction) at distances greater than 20 km. While this may be true, we have no site-specific data to verify this observation for our model domain. The GM predicted-to-observed value of 1.0 will potentially result in model overprediction and, thereby, provide at least a conservative estimate of concentrations at these distances. Correction factor distributions were truncated by a minimum value of 0.01 and a maximum of 1000.

Application of this factor on a year-by-year basis assumes year-to-year annual average concentrations are independent from one another. Analysis of the annual average *X/Q* values for each year in the 5-year meteorological data set indicated annual average concentrations at some locations are correlated (to some degree) from year-to-year. Ideally, we would like to have meteorological data from the entire assessment period in order to estimate the year-to-year correlations, but these data are lacking. In order to account for the unknown year-to-year correlation, we have assumed a correlation coefficient of 1.0. This assumption will tend to overestimate uncertainty in time-integrated concentration (*TIC*), but is justified based on our lack of knowledge about year-to-year correlations. Details concerning incorporation of this factor in the Monte Carlo uncertainty analysis are discussed in the Risk Calculation section of this report.

Meteorology Uncertainty. Meteorology uncertainty arises because we are using 5 years of meteorological data spanning a recent time period (1989–1993) to define an annual average *X/Q* value (concentration divided by release rate) that will be applied to all previous years for the assessment period (1952–1989). The question is, how well does this 5-year period represent the past? Comparisons of annual average *X/Q* values computed with a 5-year data set to the annual average *X/Q* values computed using the meteorological data for each specific year was recently performed for the Fernald Dosimetry Reconstruction Project (Killough et al. 1996). Meteorological data from the Cincinnati Airport from 1987 to 1991 composed the 5-year composite meteorological data set. Annual average *X/Q* values computed with these data were then compared with the annual average *X/Q* value computed for each specific year using the meteorological data for that specific year. The years spanned from 1951 to 1991. Concentrations were calculated at 160 receptors ranging in distance from 1000 to 10,000 m from the release point. A straight line Gaussian plume model for a 10-m release height was used to generate the *X/Q* values. The 5-year composite *X/Q* divided by the *X/Q* for the specific year (P/O ratio) forms the basis of Figure 7 (upper graph). A similar procedure was applied to the *X/Q* values generated for this study and is depicted in the lower graph in Figure 7. However, only the composite period is shown because meteorological data from previous years were not obtained. The lower graph in Figure 7 was generated using the RATCHET model and Building 776 *X/Q* values for 2,300 receptors in the model domain Figure 7 depicts the 5th, 50th and 95th percentile of the cumulative frequency distribution for all points in the model domain. Note that for the composite period, the spread of the data is similar for both data sets.

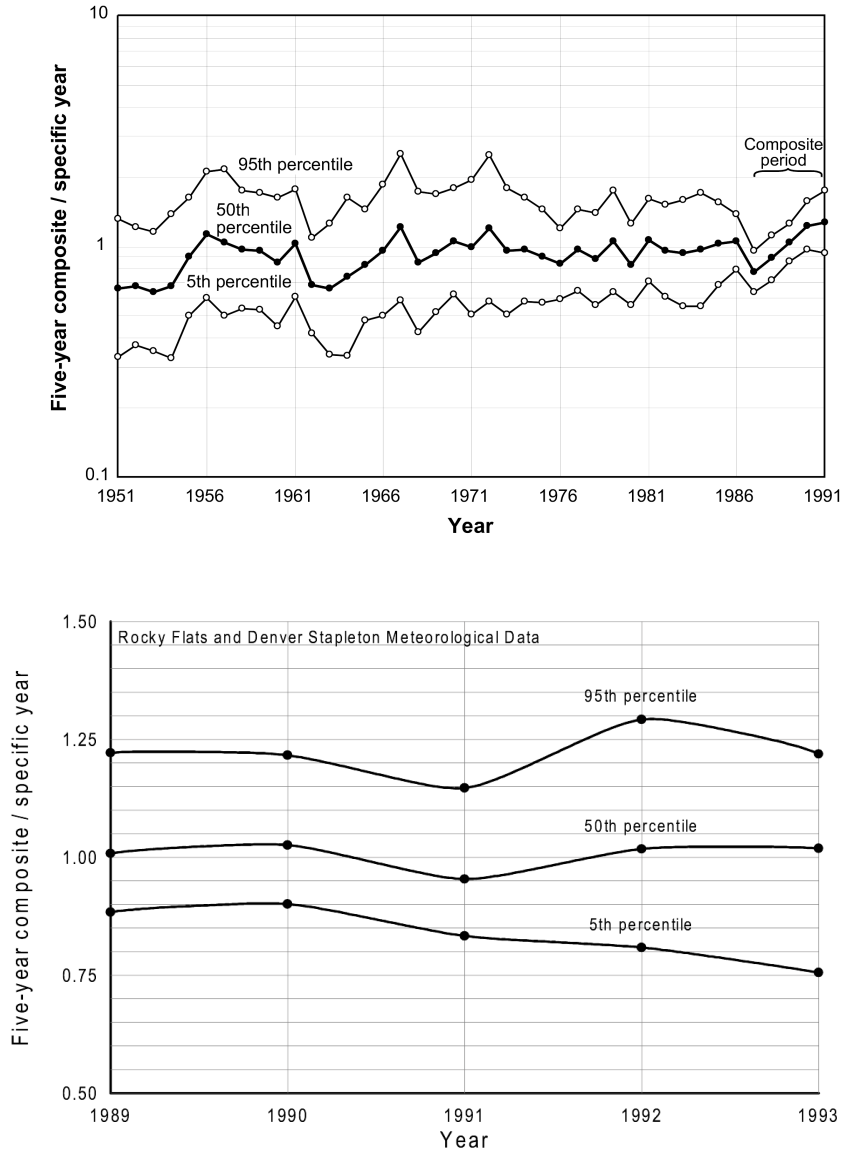

Figure 7. Distributions of P/O ratios for *X/Q* calculated with the Cincinnati meteorological data (upper graph) and RFP–Denver Stapleton International Airport meteorological data (lower graph). Predicted (P) corresponds to *X/Q* values for a five-year composite; observed (O) corresponds to the *X/Q* values for a specific year (from Killough et al. 1996).

As one would expect, the spread is much larger for those years that do not include the 5-year composite data. The long-term trend of these data may not depend strongly on location. If this procedure is applied to the RFP environs using Denver Stapleton International Airport data for instance, the locus of the 50th percentiles is likely to look somewhat different, although the amplitudes may be similar. Obtaining meteorological data from past years (1952–1989) for

Denver Stapleton International Airport and performing the calculations is not a trivial task, and the overall impact on the results may be similar to what is observed at Cincinnati based on a similar spread of these data for the composite period at both locations. For this reason, we have chosen instead to adapt these data to our analysis.

The Fernald data were represented by a multiplicative correction factor having a GM of 1.0 and GSD of 1.7. This distribution was developed using the following sampling scheme:

1. Noting from Figure 7 that the maximum range in the GMs is a factor of two, a GM was randomly selected from a log-uniform distribution with a minimum $2^{-1/2}$ and maximum $2^{1/2}$.
2. Using the GM from step (1) and GSD = 1.61 (the maximum GSD calculated from the ratio of the 5-year composite *X/Q* to specific year *X/Q* for the 40 years of data), a sample is drawn from a lognormal distribution with these parameters.
3. Values are stored from step (2) and the process is repeated.

This somewhat conservative procedure takes account of the year-to-year variability in the GM of the 5-year composite *X/Q* to specific year *X/Q* ratio, as well as the uncertainty associated with distance and direction from the source. Sampling from this distribution amounts to sampling from a lognormal distribution whose GM is itself an uncertain parameter [compare Hoffman and Hammonds (1994)]. For a sample size of 1000, a lognormal distribution was fitted with a GM = 1.0 and GSD = 1.7. Note that this factor is applied to all receptors and all years in the assessment period. The correction factor is also independent and identically distributed for each year in the assessment period.

Plume Depletion Uncertainty. One factor not considered in many of the field studies was plume depletion from dry deposition. Most field studies use inert tracers to avoid additional complications involving plume depletion and deposition. Miller et al. (1978) illustrates that plume depletion via dry deposition has little impact on inhalation dose for deposition velocities less than 1.0 cm s^{-1} and release heights greater than 50 m for receptors within 10 km of the release point. For ground-level releases, plume depletion has a greater effect. The ratio of the depleted to nondepleted plume was 0.02 for deposition velocities in the 1.0 cm s^{-1} range and 0.67 for deposition velocities in the 0.1 cm s^{-1} range. Beryllium was not released at ground level or at 50 m, and deposition velocities calculated in RATCHET ranged from 0.3 to 1.0 cm s^{-1}. Therefore, the actual amount of plume depletion would be somewhere between these values. Deposition velocities in the 1.0 cm s^{-1} range are associated with roughness lengths of around 2.0 m, which are limited to the foothills region of the model domain where few receptors are present. For these reasons, the uncertainty in the predicted concentration from plume depletion and deposition is expected to be small for most receptors in the model domain.

Deposition velocity is not an input parameter in RATCHET, but is calculated (using Equations [2–5]) for each hour of the simulation. Deposition velocity is a function of the frictional velocity, wind speed, and a user-defined transfer resistance (r_t). The frictional velocity (Equation 5) is a function of wind speed, roughness height, and a stability correction factor that is a function of the Monin-Obukhov length and wind speed measurement height. Our approach is to vary the Monin-Obukhov length and transfer resistance and calculate alternative values for deposition velocity for a given wind speed and stability classification. Airborne concentrations calculated with alternative values for deposition velocity are compared to the airborne concentrations of the base case. The base case concentrations represent model predictions made

using a transfer resistance of 100 s m^{-1} and a Monin-Obukhov length that represents the mid-range of possible values for a given stability class. (RATCHET uses the mid-range of the possible Monin-Obukhov lengths for a given stability class when run in a deterministic mode.)

The random sampling feature in RATCHET was used to vary the Monin-Obukhov length. When random sampling is selected, specific values of the inverse Monin-Obukhov length are obtained from the range of Monin-Obukhov lengths for a given stability class. A random value between 0 and 1 is obtained and used to calculate a value of the inverse Monin-Obukhov length assuming that the inverse Monin-Obukhov length is uniformly distributed within the range.

Distributions of the transfer resistance must be provided outside the RATCHET code. The rational for the distribution of r_t was based the distribution of deposition velocities reported in Harper et al. (1995). The 5th, 50th, and 95th percentile values for deposition velocity for 1 μm particles and 5 m s^{-1} wind speed were 0.01, 0.21 and 4.1 cm s^{-1}, respectively. Assuming a lognormal distribution and a 50th percentile r_t value of 100 s m^{-1}, we multiply the ratio of the 5th/50th percentile and 95th/50th percentile from the distribution of deposition velocities by the 50th percentile transfer resistance value. The 5th percentile for the distribution of r_t was 0.01/0.21 × 100 s m^{-1} = 4.8 s m^{-1}. The 95th percentile for the distribution of r_t was 4.1/0.21 × 100 s m^{-1} = 1952 s m^{-1}. A lognormal distribution containing 100 individual r_t values was generated in Crystal Ball and output to an ASCII file to be used in the uncertainty simulation. The corresponding 5th and 95th percentile deposition velocity calculated using a 5 m s^{-1} wind speed, roughness lengths from 0.001 to 2.0 m, and the mid-range value for the Monin-Obukhov length, was 0.05 and 1.5 cm s^{-1}, respectively. The range of deposition velocities used in plume depletion uncertainty simulations would be greater because the Monin-Obukhov length is also varied.

A shell program was written to facilitate the plume depletion uncertainty calculations. For each trial, a value of r_t was read from the distribution file created earlier, and written to the RATCHET input file. The RATCHET code was then called from the shell program and run using meteorological data spanning 1 year (1990) and a unit release rate. Concentrations were output for 156 receptors located 1 to 32 km from the source. Output concentrations were saved and the process was repeated until all 100 r_t values were run. A correction factor was calculated for each trial and each receptor. The correction factor is given by

$$CF_{i,j} = \frac{C_{i,j}}{Cb_j} \tag{11}$$

where $CF_{i,j}$ = the correction factor for i^{th} trial and j^{th} receptor, $C_{i,j}$ = the concentration calculated for the i^{th} trial and j^{th} receptor, and Cb_j = the base case concentration for the j^{th} receptor. Correction factors were segregated into bins according to receptor distance. The GM and GSD were then calculated for all CF values within a given bin (Table 10).

These data show a GM near 1.0 and a GSD that increases as a function of receptor distance. As expected, the uncertainty is small, especially near the source, but uncertainty increases at greater receptor distances. The plume depletion uncertainty correction factor was assigned a lognormal distribution with a GM of 1.0 and a GSD that varies with receptor distance as given in Table 10. The correction factor was assumed to independently and identically distributed for each year in the assessment period.

Table 10. Plume Depletion Uncertainty Correction Factors

Distance (km)	GM	GSD
4	0.99	1.05
8	1.00	1.09
12	1.01	1.12
16	1.00	1.14
20	1.00	1.16
24	1.00	1.17
28	1.01	1.18
32	1.01	1.18

Summary of Prediction Uncertainty. Three correction factors are applied to our model predictions. The first correction factor accounts for the uncertainty in an annual average concentration of a non-reactive, non-depleting tracer, assuming we have the meteorological data for the that year. The second correction factor accounts for the uncertainty associated with using a 5-year composite meteorological data set (1989–1993) to predict the annual average concentrations for years past (1953–1989). The third correction factor accounts for uncertainty in the dry deposition rate and resulting plume depletion for specific year. The three correction factors are independent of one another and are represented by lognormal distributions. The dispersion correction factor is assumed to be correlated from year to year (correlation coefficient = 1.0). The other correction factors are independent from year-to-year. Table 11 summarizes all three correction factors. Integration of these stochastic factors into the *TIC* estimates are discussed in the Risk Calculations section of this report.

Table 11. Summary of Uncertainty Correction Factors Applied to Annual Average Concentration Predictions

Receptor distance (km)	Dispersion uncertainty		Meteorology uncertainty		Depletion uncertainty	
	GM[a]	GSD	GM	GSD	GM	GSD
<4	1.1	2.2	1.0	1.7	1.0	1.05
8	1.1	2.0	1.0	1.7	1.0	1.09
12	1.1	2.0	1.0	1.7	1.0	1.12
16	1.1	2.0	1.0	1.7	1.0	1.14
20	1.0	2.2	1.0	1.7	1.0	1.16
24	1.0	2.2	1.0	1.7	1.0	1.17
28	1.0	2.2	1.0	1.7	1.0	1.18
>32	1.0	2.2	1.0	1.7	1.0	1.18

[a] Dispersion uncertainty GM is the inverse of the GM of predicted-to-observed ratios.

Annual Average *X*/*Q* Values

The procedure and models described in the previous sections were used to calculate an annual average *X*/*Q* for all concentration grid nodes in the model domain. Grid node spacing for the concentration grid was set at 1000 m. Annual average *X*/*Q* values were calculated separately for releases from Building 776 (Figure 8) and Building 444 (Figure 9). The annual average *X*/*Q* at each of the grid nodes for each year of meteorological data (1989–1993) were computed for a constant unit release (1 mg s^{-1}) from each building. The five *X*/*Q* values at each grid node were then averaged to yield a 5-year composite annual average X/Q. Isopleth maps were generated using *X*/*Q* data gridded using the minimum curvature routine found in the Surfer® software (Golden Software Inc. 1996).

The dispersion patterns shown in Figures 8 and 9 are characterized by a east northeast trending ellipsoid shaped plume. Wind roses constructed using RFP data from 1984–1993 (DOE 1995a) indicate the predominant wind direction to be from the west northwest. Higher concentration isopleths near the source trend mostly easterly; however, farther away from the source, concentration isopleths trend to the northeast. The northeast trend is believed to be due to the influence of the Platte River Valley and the diurnal pattern of upslope-downslope conditions that characterize the general air movement on the Colorado Front Range environs (Crow 1974). Downslope conditions typically occur during the evening hours and are characterized by drainage flow of cooler air from the foothills to the plains. Westerly winds predominate, but the direction may be altered by local topography. Upslope conditions are a result of daytime heating and typically result in easterly winds that prevail during the daylight hours with transition from upslope to downslope conditions occurring during the evening and transition from downslope to upslope occurring during the morning. During evening hours under stable conditions, cool air near the surface drains from the Denver metropolitan area down the Platte River Valley (which flows to the northeast) and out to the plains. During daylight hours and after surface heating has eliminated the cooler surface layer, the downslope conditions cease. This is followed by a brief period of relatively calm winds, which in turn is followed by return of air up the valley or upslope conditions. Meteorological data at Denver Stapleton International Airport captures these transitions in the Platte River Valley that are reflected in the *X*/*Q* isopleth maps.

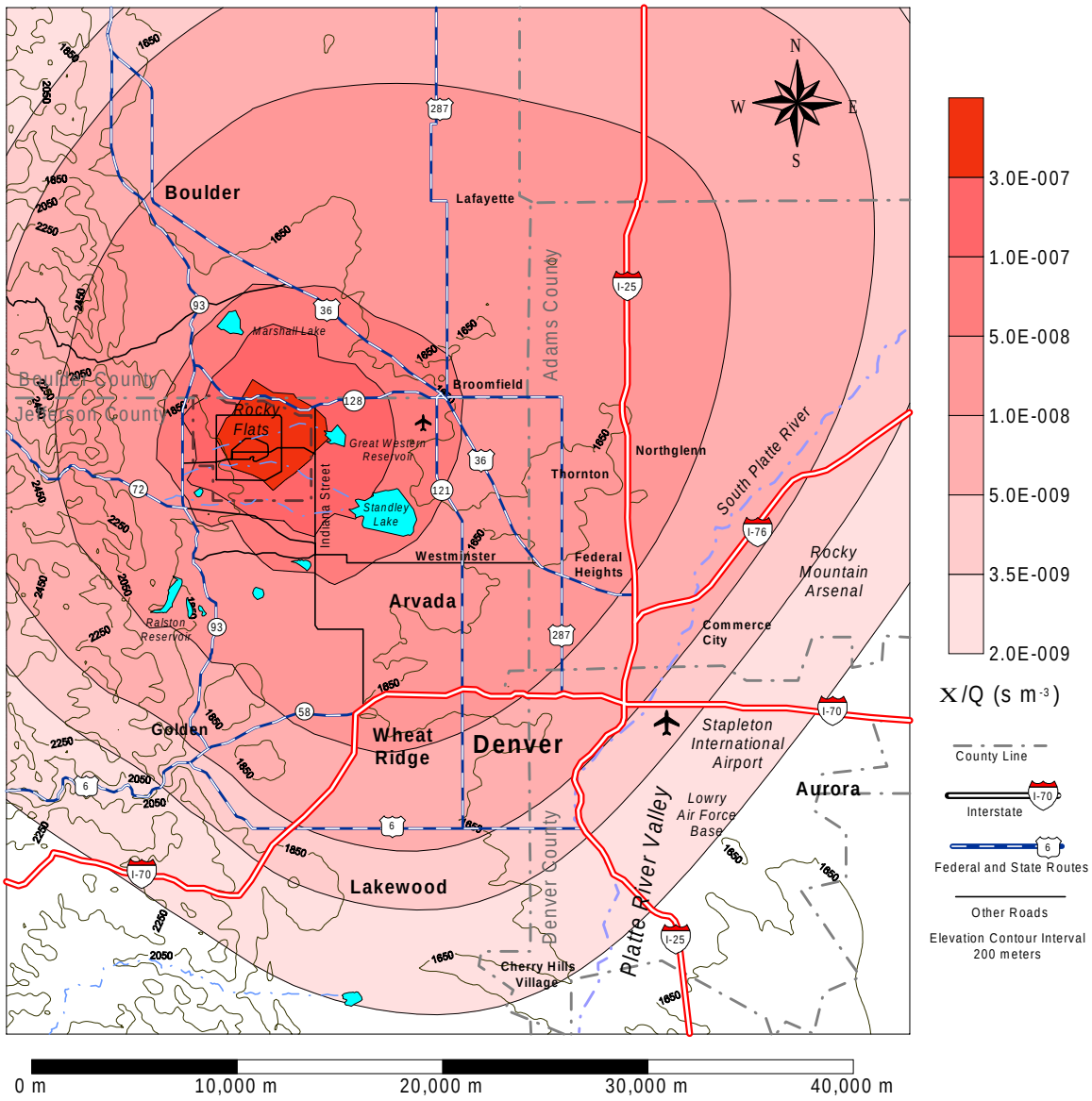

Figure 8. Isopleth map of the annual average X/Q for particulate releases from Building 776 using meteorological data from the RFP and Denver Stapleton Airport from 1989–1993.

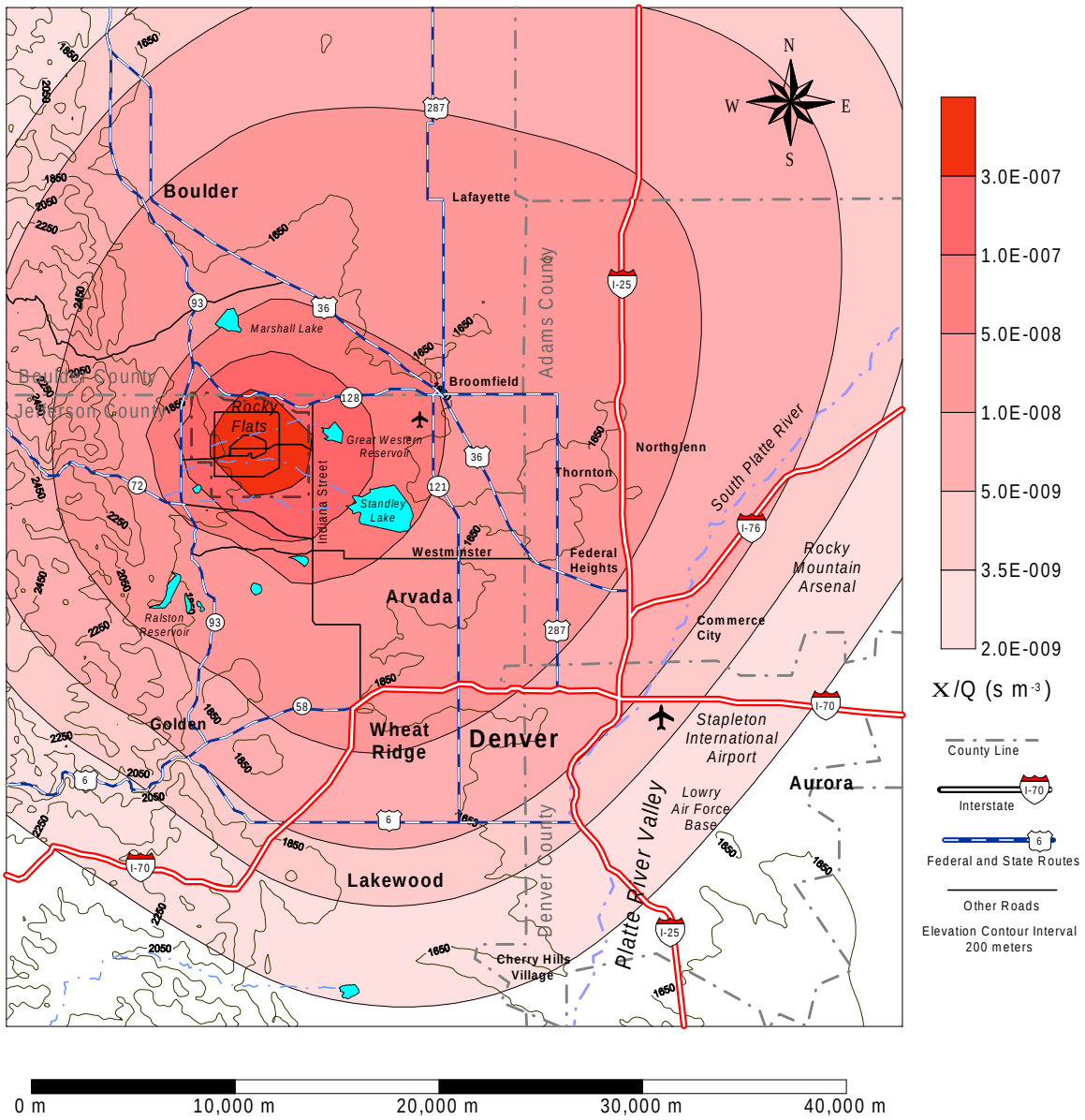

Figure 9. Isopleth map of the annual average X/Q for particulate releases from Building 444 using meteorological data from the RFP and Denver Stapleton Airport from 1989–1993.

Predicted Concentrations

Predicted concentrations of beryllium at specific receptors were calculated for each year in which source term information was available. Uncertainty in the predicted concentration included uncertainty in the dispersion estimate and source term. The concentration for the i^{th} year is given by

$$C_i = \sum_{j=1}^{2} X/Q_j \; Q_{i,j} \; CF_1 \; CF_2 \; CF_3 \qquad (12)$$

where

X/Q_j = dispersion factor for source j (concentration divided by source term, y m^{-3})
$Q_{i,j}$ = annual release of beryllium for the i^{th} year for j^{th} source (Building 776 or 444)
CF_1 = dispersion uncertainty correction factor
CF_2 = meteorology uncertainty correction factor,
CF_3 = plume depletion uncertainty correction factor.

The correction factors and source term are stochastic quantities. Therefore, the concentration is also stochastic quantity. The concentration a hypothetical receptor is exposed to is the sum of the prediction concentrations from Building 776 and Building 444 releases. Median value predicted concentrations at the location of highest concentration outside the buffer zone (east of the plant along Indiana Street) for all years in the assessment ranged from 1.3×10^{-6} ng m^{-3} in 1986 to 7.3×10^{-4} ng m^{-3} in 1968, the year of highest release (Figure 10). The maximum concentration in the model domain for the year of highest release (1968) was calculated within the plant buffer zone and ranged from 2.5×10^{-3} ng m^{-3} (5th percentile) to 6.8×10^{-2} ng m^{-3} (95th percentile). This can be compared with an annual average natural background range of 0.03 to 0.3 ng m^{-3}, (median of 1×10^{-1} ng m^{-3}) estimated in Rope et al. (1999). Note that the predicted offsite concentrations would be indistinguishable from background.

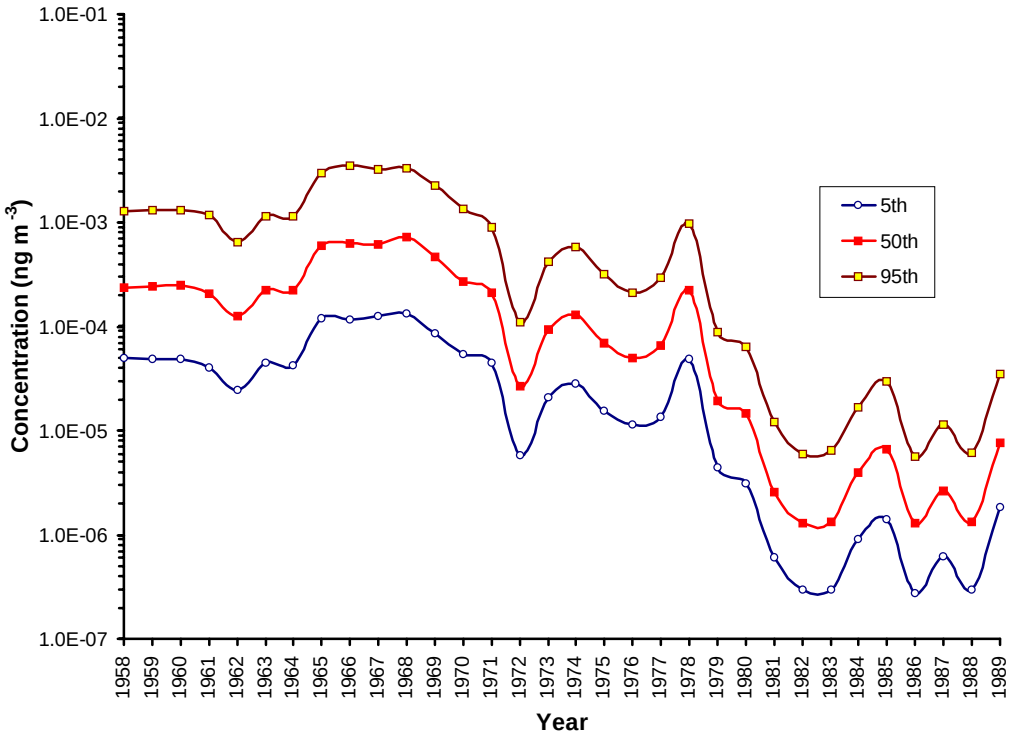

Figure 10. Predicted beryllium concentration as a function of year for a receptor located east of the plant on Indiana Street outside the current buffer zone.

Concentration of beryllium in soil from airborne deposition was calculated at the location of highest deposition outside the buffer zone and east of the plant along Indiana Street. Integrated

surface deposition from 1958 to 1989 was converted to soil concentration by conservatively assuming a sampling depth of 1 cm and a bulk density of 1.5 g cm^{-3}. Predicted soil concentrations ranged from 6.9×10^{-6} (5^{th} percentile) to 2.6×10^{-4} mg kg^{-1} (95^{th} percentile) with a median value (50^{th} percentile) of 4.2×10^{-5} mg kg^{-1}. Note that these values are well below the mean background soil concentration of 0.66 mg kg^{-1}. These calculations support the conclusions of Barrick (1982) and Allen and Litaor (1995) that soil concentrations in the vicinity of the plant were not above background and showed no spatial trends or recognizable plumes. Environmental monitoring of beryllium in soil is discussed in the Environmental Monitoring section of this report.

Time-integrated concentrations were calculated on a receptor-specific basis. Concentrations were integrated over the duration of time a receptor resided in a given location in the model domain and are reported in the Exposure Scenario and Risk Calculation section of the report.

EXPOSURE SCENARIOS AND RISK CALCULATIONS

One of the key parts of the Rocky Flats Dose Reconstruction work is calculating health impacts to people living in the surrounding area from materials released during RFP past operations. Dose reconstruction uses a pathways approach to study the potential radiation doses and health risks of these past releases on the surrounding communities. The pathways approach begins with learning what kinds of and how much materials were released from a facility and ends with estimating the health impacts these releases had on the residents in the area. Mathematical models described in the previous section were used to model the transport of materials released from the site to the surrounding communities. In this section, we calculate health impacts (lifetime cancer incidence risk) to people living offsite from exposure to these releases.

Clearly, at this point in the study, it is not realistic to calculate individual risks for every resident who may have lived or worked in the Rocky Flats area during its operational history. At the other extreme, it is not credible to calculate only a single risk that would apply to all residents. The risk that a person receives depends upon a number of factors, such as

- Lifestyle (that is, did the person spend a great deal of time outdoors or doing heavy work on a farm)
- When and how long that person lived near the RFP (for example, during the key release events in 1957 and late 1960s or in the 1970s when releases were less)
- Age and gender of the person
- Where the person lived and worked in relation to the RFP.

To consider these features of a person's life, we developed profiles, or exposure scenarios, of hypothetical, but typical residents of the RFP area for which representative risk estimates could be made. Each scenario represents one individual. These scenarios incorporate typical lifestyles, ages, genders, and lengths of time in the area. The scenarios can also specify and vary the home and work locations. These scenarios can help individuals determine risk ranges for themselves by finding a lifestyle profile that most closely matches their background. The scenarios are not designed to include all conceivable lifestyles of residents who lived in this region during the time of the RFP operations. Rather, they provide a range of potential profiles of people in the area.

They are used here to demonstrate the methodology used to calculate risk. Additional scenarios may be added at a later time.

We calculated risks from historical beryllium releases from the RFP for nine hypothetical exposure scenarios (Table 12). As discussed earlier, inhalation was the only pathway of exposure considered in the assessment. Ingestion of beryllium in water and food and inhalation of deposited beryllium and attached to soil are potential pathways that could have been considered in more detail. However, beryllium compounds are very insoluble and tend to adhere to soil making them relatively immobile and not readily taken up by plants or accumulating in the edible portions of animal products.

Table 12. Exposure Scenario Descriptions

Exposure scenario	Sex	Year of birth	Year beginning exposure	Year ending exposure	Location of occupational activities	Location of nonoccupational activities
Rancher	Male	1925	1953	1989	Indiana St.	Indiana St.
Office worker	Female	1951	1975	1989	Denver	Broomfield
Housewife	Female	1928	1953	1989	Broomfield	Broomfield
Retiree	Male	1923	1978	1989	Arvada	Arvada
Laborer #1	Male	1953	1974	1989	Thornton	Commerce City
Laborer #2	Male	1933	1953	1974	Commerce City	Westminster
Infant[a]	Female	1958	1958	1959	Broomfield	Broomfield
Child[a]	Female	1958	1960	1965	Broomfield	Broomfield
Student[a]	Female	1958	1966	1976	Westminster	Broomfield

[a] These receptors are the same individual. Total risk over their lifetime is also reported

Exposure scenarios for the nine hypothetical receptors described in Table 12 were organized according to occupational and nonoccupational activities. Occupational activities include work, school, and extracurricular activities away from the home. Nonoccupational activities include time spent at home doing chores, sleeping, and leisure activities such as watching television. For some scenarios, the receptor was assumed to perform occupational and nonoccupational activities at a different location. For example, the office worker lives in Broomfield but works in downtown Denver. The age of the receptor and years during which exposure occurred are also considered in the when calculating exposures. The last three exposure scenarios represent the same individual but at different periods in their life. Cumulative risks over this receptor's lifetime are also reported.

Breathing Rates and Time Budgets

Each exposure scenario was divided into three types of activities: sleeping, nonoccupational activity, and occupational activity. For the infant and child scenario, occupational and nonoccupational activities are irrelevant, so instead, activities were divided into sleeping and two other activities based on the child's age. For the infant, the other two activities were awake sedentary and awake active. For the child scenario, time spent at home (indoors and outdoors) and time spent at preschool and or day care were the other two activities.

For each activity, time spent at four different exercise levels were assigned. These exercise levels were resting, sitting (sedentary), light exercise, and heavy exercise. Some examples of light exercise are laboratory work, woodworking, housecleaning, and painting. Heavy exercise usually does not exceed 2 hours per day and corresponds to occupations such as mining, construction, farming, and ranching. For each exercise level, an age- and gender-specific breathing rate was assigned. Breathing rates (Table 13) for persons age 8 and higher were obtained from Roy and Courtay (1991) and from Layton (1993) for children age 0–7.

Table 13. Breathing Rates for Various Exercise Levels as Reported in Roy and Courtay (1991) and Layton (1993)

Gender	Age	Resting ($m^3\ h^{-1}$)	Sitting ($m^3\ h^{-1}$)	Light ($m^3\ h^{-1}$)	Heavy ($m^3\ h^{-1}$)
Male	30–60	0.45	0.54	1.50	3.00
Female	30–60	0.32	0.39	1.26	2.70
Male	18	0.50	0.60	1.58	3.06
Female	18	0.35	0.42	1.32	1.44
Male	16	0.43	0.52	1.52	3.02
Female	16	0.35	0.42	1.30	2.70
Male	15	0.42	0.48	1.38	2.92
Female	15	0.35	0.40	1.30	2.57
Male	14	0.41	0.49	1.40	2.71
Female	14	0.33	0.40	1.20	2.52
Male	12	0.38	0.47	1.23	2.42
Female	12	0.33	0.39	1.13	2.17
Male	10	0.31	0.38	1.12	2.22
Female	10	0.31	0.38	1.12	1.84
Male	8	0.29	0.39	1.02	1.68
Female	8	0.29	0.39	1.02	1.68
Male	3-7	0.24	0.29	0.72	1.68
Female	3-7	0.23	0.27	0.68	1.59
Male	0-3	0.19	0.23	0.58	1.35
Female	0-3	0.14	0.17	0.45	1.02
Average, male[a]	8–17	0.37	0.45	1.28	1.49
Average, female[a]	8–17	0.33	0.40	1.18	2.25

[a] The average female breathing rate from age 8–17 was used in Scenario 9

Time budgets for various receptor activities were also based on Roy and Courtay (Table 14), but they were modified to fit specific exposure scenarios. The fraction of time spent at a specific exercise level while engaged in a given activity was assigned based on the nature of the activity. For example, the fraction of time spent at the resting exercise level while the receptor slept would be 1.0 and the other exercise levels would be 0. A weighted-average breathing rate was then applied to each activity based on the number of hours spent at each exercise level. For some scenarios (housewife, retiree, and laborer), nonoccupational activities were separated into those performed indoors and those performed outdoors. Although no distinction was made between indoor and outdoor air concentrations, exercise levels for indoor and outdoor activities differed. A time-weighted average breathing rate that included indoor and outdoor activities was calculated and applied to nonoccupational time. Each receptor was assumed to spend 15 days per

year away from the Denver metropolitan area and outside the model domain. Contaminant concentrations were assumed to be the same for indoor and outdoor air.

Table 14. Time Budgets, Weighted Breathing Rates, and Body Weights (*BW*)[a] for the Exposure Scenarios

		Fraction of time spent at an exercise level				Hours per day	Hours per day	Hours per	Weighted breathing rate
Scenario	Activity	Resting	Sitting	Light	Heavy	(workweek)	(weekend)	year	(m^3 h^{-1})
Rancher	Occupational	0.00	0.00	0.25	0.75	8.0	8.0	2800	2.62
(*BW* = 78.7 kg)	Nonoccupational	0.00	0.50	0.38	0.13	8.0	8.0	2800	1.21
	Sleeping	1.00	0.00	0.00	0.00	8.0	8.0	2800	0.45
Office worker	Occupational	0.00	0.25	0.75	0.00	8.0	0.0	2000	1.04
(*BW* = 65.4 kg)	Nonoccupational	0.00	0.50	0.38	0.13	8.0	16.0	3600	1.00
	Sleeping	1.00	0.00	0.00	0.00	8.0	8.0	2800	0.32
Housewife	Occupational	0.00	0.13	0.75	0.13	8.0	8.0	2800	1.33
(*BW* = 65.4 kg)	Nonoccupational								
	Indoor	0.00	0.50	0.38	0.13	4.0	4.0	1400	1.00
	Outdoor	0.00	0.38	0.50	0.13	4.0	4.0	1400	1.11
	Total nonoccupational	0.00	0.44	0.44	0.13	8.0	8.0	2800	1.06
	Sleeping	1.00	0.00	0.00	0.00	8.0	8.0	2800	0.32
Retiree	Occupational	0.00	0.50	0.50	0.00	8.0	8.0	2800	1.02
(*BW* = 78.7 kg)	Nonoccupational								
	Indoor	0.00	0.50	0.38	0.13	6.0	6.0	2100	1.21
	Outdoor	0.00	0.50	0.38	0.13	2.0	2.0	700	1.21
	Total nonoccupational	0.00	0.50	0.38	0.13			2800	1.21
	Sleeping	1.00	0.00	0.00	0.00	8.0	8.0	2800	0.45
Laborer #1	Occupational	0.00	0.13	0.50	0.38	8.0	0.0	2000	1.94
(*BW* = 78.7 kg)	Nonoccupational								
	Indoor	0.00	0.50	0.38	0.13	6.0	8.0	2300	1.21
	Outdoor	0.00	0.50	0.25	0.25	2.0	8.0	1300	1.40
	Total nonoccupational	0.00	0.50	0.31	0.19			3600	1.28
	Sleeping	1.00	0.00	0.00	0.00	8.0	8.0	2800	0.45
Laborer #2	Occupational	0.00	0.13	0.50	0.38	8.0	0.0	2000	1.94
(*BW* = 78.7 kg)	Nonoccupational								
	Indoor	0.00	0.50	0.38	0.13	6.0	8.0	2300	1.21
	Outdoor	0.00	0.50	0.25	0.25	2.0	8.0	1300	1.40
	Total nonoccupational	0.00	0.50	0.31	0.19			3600	1.28
	Sleeping	1.00	0.00	0.00	0.00	8.0	8.0	2800	0.45
Infant	Awake–sedentary	0.00	0.71	0.14	0.14	7.0	7.0	2450	0.33
(*BW* = 9.4 kg)	Awake –active	0.00	0.00	1.00	0.00	1.0	1.0	350	0.45
	Sleeping	1.00	0.00	0.00	0.00	16.0	16.0	5600	0.14
Child	Home								
(*BW* = 15.8 kg)	Indoor	0.00	0.50	0.42	0.08	6.0	6.0	2100	0.55
	Outdoor	0.00	0.00	0.67	0.33	1.5	1.5	525	1.04
	Total home					7.5	7.5	2625	0.65
	School-indoor	0.00	0.80	0.20	0.00	2.5	2.5	875	0.35
	Sleeping	1.00	0.00	0.00	0.00	14.0	14.0	4900	0.23
Student	Home								
(*BW* = 44.4 kg)	Indoor	0.00	0.44	0.56	0.00	4.5	8.0	1925	0.83
	Outdoor	0.00	0.00	0.25	0.75	2.5	6.0	1225	1.98
	Total home	0.00	0.22	0.40	0.38	7.0	14.0	3150	1.28
	School								
	Indoor	0.00	0.75	0.25	0.00	6.0	0.0	1500	0.59
	Outdoor	0.00	0.00	0.25	0.75	1.0	0.0	250	1.98
	Total school	0.00	0.38	0.25	0.38	7.0	0.0	1750	0.79
	Sleeping	1.00	0.00	0.00	0.00	10.0	10.0	3500	0.33

[a] Body weights were obtained from Finley et al. (1994).

Time-weighted average breathing rates were calculated for the three activities for which each receptor was assumed to be engaged. The time-weighted average breathing rate is given by

$$WBR_j = \sum_{i=1}^{4} BR_i \; f_{i,j} \tag{13}$$

where

WBR_j = time-weighted average breathing rate for the j^{th} activity ($m^3\ h^{-1}$)
BR_i = breathing rate for the i^{th} exercise level ($m^3\ h^{-1}$)
$f_{i,j}$ = fraction of time spent at the i^{th} exercise level for the j^{th} activity.

To reiterate, three activities were defined for each exposure scenario. The location of exposure for occupational activities may be different from nonoccupational activities. The breathing rate during a given activity was the time-weighted average breathing rate of the four exercise levels. Exercise levels were grouped into resting, sitting, light exercise, and heavy exercise.

Risk Calculation and Uncertainty

Calculation of lifetime cancer incidence risk involved three steps:

1. Calculate the *TIC* at the point of exposure.
2. Calculate the amount of beryllium inhaled by the receptor.
3. Multiply the beryllium intake by a slope factor that relates the risk of cancer incidence to the amount of beryllium inhaled per day per unit body weight.

In each of these steps, Monte Carlo sampling techniques are used to propagate uncertainty through the calculation. A Monte Carlo calculation consists of multiple iterations or trials of a computational endpoint (risk). For each trial, parameter values are randomly chosen from distributions that quantitatively describe our knowledge of the parameter. After randomly selecting a set of parameter values, the endpoint is calculated and the procedure is repeated numerous times until an adequate distribution of the endpoint is obtained.

Uncertainty in risk estimates were based on uncertainty in the *TIC* and carcinogenic slope factors. Receptor behavior patterns (i.e., the time spent doing different activities at different exertion levels) and their physical attributes(body weight and breathing rate) were considered fixed quantities. The exposure scenarios were set up to evaluate risks for hypothetical individuals and did not consider variability within the population of potential receptors. Therefore, the parameters describing their physical attributes and behavior were considered fixed.

The procedure outlined above requires an estimate of the *TIC* at the point of exposure. A receptor can be exposed at two locations; place of work (occupational) and place of residence (nonoccupational and sleeping). Consider a Monte Carlo calculation consisting of m trials. The *TIC* of the k^{th} trial ($0 < k \leq m$) for source j and location i is

$$TIC_{i,j} = CF_1 \sum_{l=1}^{n} CF_2 \; CF_3 \; \mathrm{X} \; / \; Q_{i,j} \; Q_{j,l} \; \Delta t \tag{14}$$

where

$X/Q_{i,j}$ = dispersion factor for source j and location i (y m^{-3})
$Q_{j,l}$ = source term for year l and source j (mg y^{-1})
CF_1 = stochastic correction factor for dispersion (unitless)
CF_2 = stochastic correction factor meteorology (unitless)
CF_3 = stochastic correction factor for deposition and plume depletion (unitless)
n = number of years exposed
Δt = time increment (1 year).

Notice that the dispersion correction factor (CF_1) is outside the summation symbol. For each Monte Carlo trial, CF_1 is sampled once but the correction factors, CF_2, CF_3, and source term are sampled n times. This sampling scheme was used to allow for year-to-year correlation in annual dispersion estimates as discussed earlier. The amount of beryllium inhaled by a receptor for the k^{th} Monte Carlo trial is

$$I = \sum_{j=1}^{2} \left(TIC_{1,j}\, WBR_1\, T_1 + TIC_{2,j}\, WBR_2\, T_2 + TIC_{2,j}\, WBR_3\, T_3 \right) \quad (15)$$

where

I = intake of beryllium by the receptor for the exposure period (mg)
$TIC_{1,2}j$ = time-integrated concentration for occupational and nonoccupational (including sleeping) locations and j^{th} source (mg-y m^{-3})
$WBR_{1,2,3}$ = time-weighted average breathing rate for occupational, nonoccupational, and sleeping activity (m^3 h^{-1})
$T_{1,2,3}$ = hours per year for occupational, nonoccupational, and sleeping activity (h y^{-1}).

The subscripts 1, 2, 3 refer to occupational, nonoccupational, and sleeping activity respectively. Note that the *TIC* values (Table 15) are only calculated at 2 locations and that the same *TIC* value is applied to sleeping and nonoccupational awake activities. Distributions of *TIC* values in Table 15 are described in terms of their *GM* and *GSD*. Analysis of the data points that comprise these distributions show they are best represented by a lognormal distribution. However, in practice, calculations are performed using the actual distribution (made up of *m* number of trials) and not the lognormal representation. Magnitude of the *TIC* was dependent on the length of exposure, location of exposure, and magnitude of source during exposure. Differences in the GSD values between scenarios are mainly related to the length of exposure and magnitude of the dispersion correction factor. Longer integration time typically corresponds to lower *GSD*s (but not lower variance) because summation of the independent stochastic variables (CF_2 and CF_3) over the integration period results in a lower coefficient of variation (*CV*) of the sum compared to the *CV* of individual years. The *CV* is the standard deviation of the sum divided by the mean of the sum (σ/μ). Like the *CV*, the GSD is a relative measure of the spread of the data comprising the distribution. The decrease in the GSD for longer averaging times is because the *relative* variability in the *TIC* decreases with increasing integration time.

Table 15. Time-Integrated Concentrations for Each Receptor Scenario and Source for Occupational and Nonoccupational Activities

Scenario	Activity	Time-integrated concentration, Building 444[a] (mg-y m^{-3})	Time-integrated concentration, Building 776[a] (mg-y m^{-3})
Rancher	Occupational	3.6×10^{-09} (2.3)	3.2×10^{-09} (2.3)
	Nonoccupational	3.6×10^{-09} (2.3)	3.2×10^{-09} (2.3)
Office worker	Occupational	3.9×10^{-12} (2.3)	2.6×10^{-12} (2.3)
	Nonoccupational	5.6×10^{-11} (2.1)	4.3×10^{-11} (2.1)
Housewife	Occupational	7.7×10^{-10} (2.1)	5.9×10^{-10} (2.1)
	Nonoccupational	7.7×10^{-10} (2.1)	5.9×10^{-10} (2.1)
Retiree	Occupational	1.2×10^{-11} (2.5)	7.6×10^{-12} (2.5)
	Nonoccupational	1.2×10^{-11} (2.5)	7.6×10^{-12} (2.5)
Laborer #1	Occupational	2.1×10^{-11} (2.1)	1.5×10^{-11} (2.1)
	Nonoccupational	7.6×10^{-11} (2.3)	5.1×10^{-12} (2.3)
Laborer #2	Occupational	8.0×10^{-11} (2.3)	5.3×10^{-11} (2.3)
	Nonoccupational	6.3×10^{-10} (2.1)	4.2×10^{-10} (2.1)
Infant	Occupational	5.6×10^{-11} (2.5)	4.4×10^{-11} (2.5)
	Nonoccupational	5.6×10^{-11} (2.5)	4.4×10^{-11} (2.5)
Child	Occupational	2.1×10^{-10} (2.2)	1.6×10^{-10} (2.2)
	Nonoccupational	2.1×10^{-10} (2.2)	1.6×10^{-10} (2.2)
Student	Occupational	3.5×10^{-10} (2.2)	2.3×10^{-10} (2.2)
	Nonoccupational	4.2×10^{-10} (2.2)	3.2×10^{-10} (2.2)

[a] Geometric mean (geometric standard deviation)

Finally, calculating the incremental lifetime cancer incidence risk requires estimates of the slope factor (SF). Distributions of SFs were described previously in this report. Carcinogenic risk from beryllium inhalation was calculated using the standard risk equations described in EPA (1989) and given by Equation 16.

$$R = \frac{SF\ I}{BW\ AT} \tag{16}$$

where

R = cancer incidence risk
SF = carcinogenic slope factor (kg-d mg^{-1})
I = distribution of integrated contaminant intake (mg)
BW = body mass (kg)
AT = averaging time (70 years × 365 days per year).

Age-specific body weights used in Equation 16 are presented in Table 13. Monte Carlo sampling was performed using a FORTRAN program written specifically for this application. Each step of the Monte Carlo simulation is described below:

1. The distribution of *TIC* values (Equation 14) for each receptor activity and each source were calculated first. Nonoccupational and sleeping activities were assumed to be at the same location. Therefore, 2 *TIC* values were calculated for each receptor and each source. Each *TIC* distribution contained *m* number of individual trials. If occupational

and nonoccupational activities occurred at the same location, then a single *TIC* value was used for each source.

2. Each of the *TIC* trials are multiplied by the WBR_i and T_i, (corresponding to the i^{th} receptor activity), then summed over all sources and receptor activities to yield the total contaminant intake of the k^{th} trial (Equation 15). The procedure is repeated for all *m* trials.

3. Each estimate of total contaminant intake is multiplied by a randomly selected SF value and divided by body weight and averaging time to give an estimate of the lifetime cancer incidence risk. This calculation is repeated *m* times to yield a distribution of lifetime cancer incidence risks.

4. Percentiles, GM, and GSD values were then calculated from the distribution of *m* risk values.

The total risk over the lifetime of the individual that represents the infant, child, and student scenarios was calculated differently. For each trial, contaminant dose (intake divided by body weight, [mg kg^{-1}]) were calculated for each year the receptor was exposed. Note that body weight and breathing rate change as the individual matures. Meteorological, deposition, and source term uncertainty were applied to each years dose estimate. The dose was summed across all years of exposure then multiplied by the dispersion correction factor and slope factor and divided by the averaging time. This process was repeated *m* times resulting in a distribution of lifetime cancer risk estimates to the individual.

FORTRAN routines for generating random numbers and selecting values from normal, lognormal, triangular, and uniform distributions were adapted from Press et al. (1992). The output distributions provided in this report were generated from 2000 trials.

RISK ESTIMATES

Geometric mean incremental lifetime cancer incidence risk estimates for beryllium inhalation (Table 16) were greatest for the rancher scenario (3.9×10^{-10}) and least for the retiree scenario (7.5×10^{-13}). Appendix A contains detailed output from the computer code used to calculate time-integrated concentrations and risk values. The 5th and 95th percentile values of the risk estimates are illustrated in Figure 11. Using the rancher scenario as an example, these risks may be interpreted as follows:

- *There is a 90% probability that incremental lifetime cancer incidence risk to the rancher was between 7.5×10^{-11} (5% value) and 1.8×10^{-9} (95% value).*
- *There is a 5% probability that incremental lifetime cancer incidence risk for the rancher was less than 8.4×10^{-9}*
- *There is also a 5% probability the risk was greater than 7.5×10^{-11}.*

Estimated risks were a function of exposure time, exposure duration, and location of exposure. Also note that risk is inversely proportional to body weight which explains the relatively high risk for the infant scenario. Beryllium is only a suspected human carcinogen and SFs were based on extrapolations from animal data to adult humans. Age and sex dependencies

were not considered, and furthermore, little data exits to develop such values. Therefore, the risks presented here for the infant, child, and student scenarios must be interpreted with caution because SFs for adults were used to compute carcinogenic risk.

Despite these shortcomings, risk estimates are well below the EPA point of departure for acceptable risks (10^{-6} to 10^{-4}). As stated previously, the EPA SF values are not intended to represent the true carcinogenic risk to an individual, but were designed to be protective of human health. So the risk values reported here must be evaluated in light of the EPA point of departure for acceptable risk. The risks presented in this report are not comparable to risks calculated in Phase I and reported in the Task 8 report (ChemRisk 1994c). Differences are summarized as follows:

- Phase I reported the risk from 1 year of exposure. The risks presented in this report represent integrated lifetime exposure to airborne releases of beryllium from RFP while the receptor resided in the model domain.
- Risks from ingestion of contaminated food stuffs and inhalation of resuspended material that were computed for Phase I are not considered in this analysis.
- Absorption factors for inhalation (0.5) and ingestion (0.01) that were used in Phase I were considered inappropriate and not used in Phase II.
- Receptor scenarios differed between Phases I and II.

Table 16. Incremental Lifetime Carcinogenic Incidence Risk from Beryllium Inhalation Calculated for the Nine Exposure Scenarios

Scenario	GM	GSD
Rancher	3.9×10^{-10}	2.7
Office Worker	2.8×10^{-12}	2.5
Housewife	6.3×10^{-11}	2.5
Retiree	7.5×10^{-13}	2.8
Laborer #1	1.1×10^{-12}	2.6
Laborer #2	3.4×10^{-11}	2.5
Infant	7.6×10^{-12}	2.8
Child	2.9×10^{-11}	2.6
Student	4.1×10^{-11}	2.6
Total (Child)[a]	7.6×10^{-11}	2.6

[a] Total (Child) represents the integrated risk for the infant, child, and student scenarios

The scenario involving the rancher may be considered the maximum exposed individual in the model domain because he was placed at the point of highest concentration outside the RFP buffer zone and remained there for the entire operating period of the plant. However, it is recognized that ranchers could have been grazing cattle within the current buffer zone and up to the old cattle fence. There were also bunkhouses or some type of permanent overnight ranch camp to the northeast within the buffer zone. To increase the risk substantially from our estimates, the concentration within the buffer zone would have to be several orders of magnitude greater than outside it. This simply is not the case as is evidenced by the X/Q plots provided

previously in the report and differences between the predicted concentration at Indiana Street and the maximum concentration in the model domain. The resulting risk, accounting for occupancy time while exposed to concentrations within the buffer zone, would still be at or below the EPA point of departure for acceptable risk of 10^{-6} to 10^{-4}.

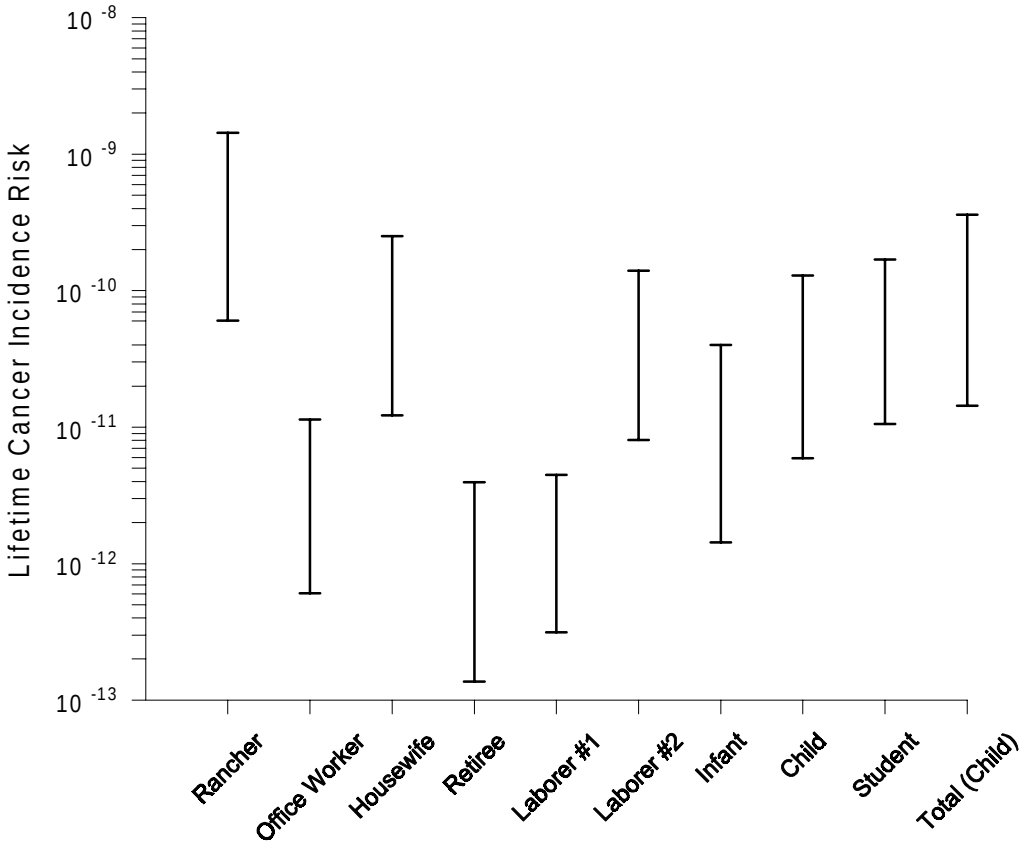

Figure 11. Incremental lifetime cancer incidence risk estimates for the nine exposure scenarios. The range of values shown represent the 5th and 95th percentiles on the cumulative density function. The Total (Child) represents the sum of the infant, child, and student scenarios.

Although beryllium exposures for workers at the RFP have been of great concern and this may have caused public concern about health effects due to beryllium exposure offsite, the results this study predicts that lung cancer risk from beryllium exposures offsite were negligible. The risk for chronic beryllium disease in the offsite public was not calculated at the time this assessment was done. However, the maximum concentration estimated in the entire model domain ranged from 2.5×10^{-6} μg m^{-3} to 6.8×10^{-5} μg m^{-3}, inside the plant complex, concentrations about 300 times less than the EPA's RfC of 2.0×10^{-2} μg m^{-3}. The maximum concentration predicted along Indiana Avenue ranged from 9.4×10^{-7} μg m^{-3} to 1.4×10^{-5} μg m^{-3}, concentrations more than 1400 times less than the RfC. A hazard index calculated using these values would be well below 1.

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