Health Risks of Toxic Emissions from a Coal-Fired Power Plant

J. G. Bolten, P. F. Morrison,
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RAND

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PREFACE

This report is the fifth in a series of RAND studies developing and applying a general methodology for assessing the chronic health risks of toxic pollutant emissions from coal-fired electric power plants. The first report, *Risk-Cost Assessment Methodology for Toxic Pollutants from Fossil Fuel Power Plants* (R-2993-EPRI), developed the methodological framework and applied it in a case study of arsenic emissions from a hypothetical coal-fired plant. The second, *Alternative Models for Risk Assessment of Toxic Emissions* (N-2261-EPRI), expanded the description of the methodology in the first report to discuss (1) alternative models for the framework, (2) simplified environmental transport models that may be used for initial analysis, and (3) the implications of certain assumptions used in the exposure analysis. It also described the application of the methodology, incorporated into the computer program WTRISK, to a second case study of selenium emissions from a coal-fired plant.

The third and fourth studies, *A Case Study of Selenium Emissions from a Coal-Fired Power Plant* (N-2387-EPRI) and *A Case Study of Beryllium Emissions from a Coal-Fired Power Plant* (N-2484-EPRI), described the application of the WTRISK program to a real case study of selenium and beryllium emissions from an existing coal-fired power plant in eastern Pennsylvania. These reports considered exposure to plant emissions arising from consumption of drinking water and aquatic organisms.

The present report summarizes the previous two Notes and broadens the case study to include population exposure associated with consumption of animal and vegetable products produced in the region. This effort, sponsored by the Energy Analysis and Environment Division of the Electric Power Research Institute, is directed toward federal, state, and local policymakers, as well as members of industry, concerned with risk assessment and pollution control regulation. These reports are intended to increase understanding of the toxic risk assessment process—an issue of increasing importance in recent years—and its inherent uncertainties.

SUMMARY

Toxic pollutants can be discharged by many types of industrial facilities, including fossil-fuel-fired steam electric power plants. These pollutants will be dispersed throughout the environment and may be inhaled, ingested, or absorbed by the population of the region around the source. To limit the risk associated with this long-term, low-level exposure, the discharge of such toxic materials is being regulated by state and federal governments.

In recent reports¹ we have described the development and application of a general approach and a computer program that can be used to compare the health risks associated with alternative levels of regulation. The general process of health risk assessment consists of five distinct stages: (1) source emissions, (2) environmental transport, (3) exposure analysis, (4) toxicology and pharmacokinetics, and (5) dose-response analysis. To assess the risk associated with discharge of a toxic pollutant from a particular source, one must use models, measured data, or a combination of both to (1) define emission rates, (2) determine consequent pollutant concentrations in the environment, (3) calculate exposure rates for the affected population, and (4) translate these exposure rates into a change in the incidence of specific chronic health effects in the affected population.

The computer program WTRISK includes (1) simplified environmental transport models for the atmosphere, overland, and surface water media, (2) procedures to calculate exposure rates through all potential primary pathways, and (3) six alternative dose-response functions for calculating chronic health risks. Using this program, an analyst can perform initial calculations of pollutant concentrations, exposure rates, and health risks for carcinogens. Parameters for the dose-response functions must be estimated outside the program or obtained from the literature. Moreover, for health effects that have thresholds or cannot, for other reasons, be determined using dose-response models, final risk calculations must be performed separately using the results of the exposure analysis.

Our previous work applied the general approach and WTRISK to illustrative case studies of arsenic and selenium discharges from hypothetical coal-fired power plants in Washington and Nebraska. In

¹Bolten et al. (1983); Bolten et al. (1985).

two recent Notes² we applied the WTRISK program to an actual case study of selenium and beryllium discharges from a coal-fired power plant in Pennsylvania (the Montour Steam Electric Station operated by Pennsylvania Power & Light Company, PP&L). In this report, we summarize that work and extend the analysis to include calculation of pollutant exposure associated with consumption of animal and vegetable products raised in the region surrounding the Montour plant. We recalculate the health risks incorporating these exposure pathways and those considered in the original analysis—consumption of drinking water and aquatic organisms.

Specifically, the study investigates the direct discharges from the plant waste treatment and storm water basins into the Chillisquaque Creek, which subsequently flows into the Susquehanna River. Using alternative beryllium and selenium concentrations in plant discharges, it calculates how much the incidence of cancer and selenosis might increase in the regional population that consumes drinking water or fish obtained from the Susquehanna River or any of several animal or vegetable products raised in the region. These products could accumulate pollutants through irrigation or animal use of contaminated surface water for drinking. The Susquehanna River serves as the source of drinking water for approximately 222,000 people distributed along its length below the mouth of the Chillisquaque. An additional 1,300,000 people may consume fish caught in the river or the various animal and vegetable products investigated in the analysis.

The investigation revealed that every stage of the analysis is characterized by high levels of uncertainty. In particular, plant discharge rates, fish production and consumption, animal and vegetable product consumption, and the chemical activity and behavior, background concentrations, and chronic health effects of beryllium and selenium are not well understood. As a consequence, the study incorporates extensive sensitivity analysis at each stage to investigate these uncertainties and how they affect the results. The analysis is separated into two parts, surface water transport calculations to determine how plant emissions affect pollutant concentrations in the waterways, and exposure analysis to examine how these pollutant levels affect the consumption of pollutants and the incidence of cancer and selenosis in the regional population. In each part, we analyze a base case and alternatives selected to emphasize the importance of various assumptions and parameters.

Selenium and beryllium concentrations in the Chillisquaque and Susquehanna are calculated with WTRISK for a variety of discharge

²Bolten and Resetar (1986); Bolten et al. (1986).

rates, river flows, and removal (decay) rates, including no removal. The results show that plant emissions cause only extremely small increases in pollutant concentrations in the Susquehanna River. These increases are below measurable levels using standard Environmental Protection Agency (EPA) techniques and remain so under virtually all conditions and parameter values. In the Chillisquaque Creek, calculated pollutant concentrations also remain low, although plant discharges do increase them under some conditions. In cases having long-term mean flow rates, even with high discharge rates and no removal, the maximum selenium concentrations do not exceed 0.010 mg/l, the EPA water quality criterion for selenium.

Under similar circumstances, the calculated maximum beryllium concentrations only slightly exceed 0.20 $\mu\text{g/l}$. These concentrations are above the EPA nominal water quality criterion level of 0.068 $\mu\text{g/l}$ (EPA risk level of 10^{-5}). Using the maximum beryllium levels insures that the analysis is reasonably conservative. Actual measurements over the course of a year have shown that beryllium is normally not detected in the Chillisquaque Creek, although it is observed in plant effluents. A median background concentration in the Chillisquaque of 0.10 $\mu\text{g/l}$ was used in the analysis, based on the detection limit for beryllium using standard EPA techniques. For cases having normal beryllium discharge rates, the calculated maximum concentrations in the Chillisquaque do not significantly exceed 0.15 $\mu\text{g/l}$, even assuming a conservative pollutant with no removal.

The exposure analysis results show that the incremental beryllium and selenium exposure associated with Montour discharges is essentially negligible in virtually all cases. Even in the worst case, the increase in beryllium exposure is less than 0.7 percent of the normal background rate and the increase in selenium exposure less than 0.3 percent of normal background. In cases having realistic assumptions about river flows, the incremental exposure is at most 0.15 percent for beryllium and 0.075 percent for selenium. The relative magnitude of these incremental exposures (related to plant emissions) remains relatively constant under varying assumptions about parameter values and other factors in the calculations.

The EPA approach can be applied to calculating the health risk of beryllium ingestion. Using this method with the results of the exposure analysis, we find that the number of cancer cases (in the exposed population) that might result from plant discharges is extremely small. More specifically, the incremental individual risk of cancer from the discharges is two orders of magnitude less than the nominal EPA risk criterion level of 10^{-5} . In the sensitivity analysis, even using worst-case assumptions, the individual lifetime health risk associated with

plant emissions still remains below 2.0×10^{-7} for long-term average river flow rates.

Because no experiments or epidemiological studies have shown selenium to be carcinogenic, the endpoint for the risk calculation must be selenosis. The EPA (1980d) has calculated a consumption rate of 7.0 mg/day as the probable human threshold for this health effect. Using literature data one can fit a log normal probability density function to the distribution of selenium consumption in the population. As plant selenium emissions shift the mean of this distribution, they cause a corresponding change in the expected number of people who might exceed the selenosis threshold.

Using this approach with the results of the selenium exposure analysis, we find that for mean river flows, the incremental individual risk for a member of the exposed population is less than 1.5×10^{-9} . When we assume reasonable worst-case conditions, this risk increases to about 6.3×10^{-6} . These conditions should be suspect, however, because even the calculated background exposure rates are far above the values measured or calculated in the literature. In all situations, including the worst case, exposure rates for the affected population remain at least a factor of nine below the EPA selenosis threshold intake value of 7 mg/day.

The results indicate that Montour selenium and beryllium discharges into the Chillisquaque should have a negligible effect on the incidence of selenosis and cancer in the regional population. Moreover, the incremental exposure rates associated with these discharges are below measurable levels and can only be seen when the results are presented to more significant digits than are warranted by the data. Although the values of certain parameters in the analysis are relatively uncertain, the general results are robust with respect to these parameters.

CONTENTS

PREFACE	iii
SUMMARY	v
FIGURES	xi
TABLES	xiii
Section	
I. INTRODUCTION	1
Background	1
Procedure	2
Results and Conclusions	3
II. RISK ASSESSMENT FRAMEWORK AND PROCEDURE	6
III. ANALYSIS OF THE CASE STUDY	12
Description of Case Study	13
Plant Emissions	15
Description of Analysis	23
Sensitivity Analysis	55
Appendix	
A. MASS BALANCE CALCULATIONS	65
B. HEALTH EFFECTS OF SELENIUM AND BERYLLIUM	82
BIBLIOGRAPHY	95

FIGURES

2.1.	Risk Assessment Approach	7
2.2.	Risk Assessment Framework—Part I	9
2.3.	Risk Assessment Framework—Part II	10
3.1.	Power Plant Location for Case Study	14
3.2.	Schematic Diagram of Power Plant Water System	17
3.3.	Historical Flow Rates for the Chillisquaque Creek	33

TABLES

3.1.	Description of Nominal Case Study Power Plant	15
3.2.	Selenium and Beryllium Input Rates	18
3.3.	Selenium Mass Balance and Discharge Concentrations	21
3.4.	Beryllium Mass Balance and Discharge Concentrations	22
3.5.	Approximate Exposure from Inhalation	24
3.6.	Total Population by County and Reach	28
3.7.	Potable Water Supplies on the Lower Susquehanna River	29
3.8.	Population Distribution for Consumption of Drinking Water	30
3.9.	Population Distribution by Reach and Subregion	32
3.10.	Definition of River Reaches	36
3.11.	Parameters for Surface Water Analysis	37
3.12.	Transport Analysis Cases	39
3.13.	Mean Concentrations in Surface Water Analysis	40
3.14.	Characteristics of Aquatic Organisms	45
3.15.	Consumption Rates for Aquatic Organisms	45
3.16.	Agricultural Production by Reach	46
3.17.	Miscellaneous Agricultural Analysis Parameters	48
3.18.	Consumption Rates for Exposure Analysis	49
3.19.	Miscellaneous Parameters for the Exposure Analysis	49
3.20.	Cases for Beryllium and Selenium Exposure Analysis	50
3.21.	Exposure Rates for Alternative Cases	51
3.22.	Total and Individual Incremental Health Risks	54
3.23.	Parameter Variations for Selenium Sensitivity Analysis	56
3.24.	Selenium Exposure Rates for Sensitivity Analysis Cases	58
3.25.	Total and Individual Incremental Health Risks for Selenium	60
3.26.	Exposure Rates and Health Risks for Beryllium	61
A.1.	Montour Coal Characteristics	66
A.2.	Pollutant Input Rates	68
A.3A.	Atmospheric Emission Calculations and Assumptions— Beryllium	71
A.3B.	Atmospheric Emission Calculations and Assumptions— Selenium	72

A.4.	Pollutant Discharges in Coal Pile Runoff	73
A.5.	Inputs for Ash Basin Loading Calculations	76
A.6A.	Beryllium Outflow and Accumulation in Ash Basin	76
A.6B.	Selenium Outflow and Accumulation in Ash Basin	77
A.7A.	Beryllium Inflow to the Detention Basin	79
A.7B.	Selenium Inflow to the Detention Basin	79
A.8A.	Beryllium Discharge Rate and Concentration	80
A.8B.	Selenium Discharge Rate and Concentration	80
B.1.	Threshold Selenium Concentrations for Selenosis	89

I. INTRODUCTION

The U.S. Environmental Protection Agency (EPA) and numerous state agencies have developed and implemented regulations to control the discharge of toxic pollutants from industrial plants. These regulations, primarily developed under the provisions of the Clean Water Act of 1977, strengthen government control over toxic emissions to surface waters. The electric power industry has been, and will be, subject to the guidelines and regulations developed under this and other federal and state legislation. In particular, the discharges to surface waters from coal- and oil-fired steam electric plants are currently being regulated.

These regulations are usually based on generalized risk assessments that do not consider the specific conditions in a particular case. Thus, depending on the circumstances, they may be too restrictive in some instances and insufficient to protect public health in others. By improving the accuracy and accessibility of risk analysis, we can help to insure protection of public health without arbitrarily requiring more control of pollutant emissions than is actually needed in a particular situation.

BACKGROUND

This is the fifth in a series of RAND reports dealing with the human health risks of toxic substances emitted by coal-fired power plants. In previous reports we have described the development and application of a general approach to compare the costs and risks of alternative levels of regulation. The initial report (Bolten et al., 1983) describes the formulation of a general methodology for assessing the relative costs and risks associated with the control of toxic substances emitted by coal-fired power plants. This method emphasizes the contamination of surface water bodies but includes and follows toxic emissions through all environmental media. Pollutants are tracked through the environment to determine population exposure rates and the consequent chronic health effects on the regional population. In the second report (Bolten et al., 1985) we expand on the general approach and provide additional information for the potential user. In particular, the report discusses the general types and features of surface water transport and exposure models and briefly reviews some of the specific models available in both areas. It also describes the development of

the relatively simple environmental transport models for atmospheric, overland, and surface water transport.

These first two reports also described the application of the WTRISK computer program to the analysis of arsenic and selenium emissions from hypothetical coal-fired power plants located on the Columbia River in eastern Washington (Bolten et al., 1983) and the XYZ River in central Nebraska (Bolten et al., 1985). Although both of these case studies were valuable in demonstrating how one would apply the approach to an actual problem, they were of limited interest. Because they dealt with hypothetical power plants and scenarios, the studies could produce results with only very general conclusions.

In contrast to these earlier studies, the third and fourth reports applied the models of the WTRISK program to an actual situation. They investigated the chronic health effects associated with the selenium and beryllium emissions to surface waters from the Montour power plant operated by Pennsylvania Power & Light Company (PP&L). These studies focused on population exposure through consumption of drinking water and fish from the Susquehanna River. To increase understanding of the exposure situation and potential overall risk, we expanded this analysis to include additional exposure routes. Specifically, the additional study investigated the incremental exposure rates and chronic health risks associated with consumption of several animal and vegetable products produced on land irrigated with the surface waters affected by Montour discharges. This report describes the entire analysis for the Montour steam electric station discharge of both selenium and beryllium, including not only the initial studies but also the expanded analysis of exposure through consumption of animal and vegetable products.

PROCEDURE

The overall analysis was separated into two parts, pollutant concentration analysis and exposure analysis. The concentration analysis used a surface water transport model in WTRISK to determine how plant emissions affect selenium and beryllium concentrations in the local rivers. As stated above, discharges to surface waters were the only emissions considered. The concentrations developed by the transport analysis were used with the exposure model in WTRISK to calculate the consequent exposure rates to the affected population. Exposure pathways from the surface water emissions included consumption of water and fish from the affected waterways, and consumption of milk, beef, pork, poultry, grains, and vegetables produced on land

irrigated with the affected water. These product categories were chosen because they include almost all of the agricultural production in the counties along the affected waterways.

In both the transport and exposure analyses, we used a base case and several alternative scenarios. These alternatives were selected to emphasize the importance of alternative assumptions and parameters. In the base case, we assumed nominal emission and removal rates, mean flows, and nominal values for all parameters in the exposure calculations. In the alternative cases, parameter values were varied in a number of ways to determine how pollutant concentrations and exposure rates changed under different, reasonable assumptions. The results of the exposure calculations for alternative cases were converted into estimates of the future chronic health risks for the regional population.

The health risks for beryllium ingestion could be calculated using the dose-response functions available in WTRISK. Beryllium has been found to be carcinogenic, and the EPA has developed a dose-response function for human ingestion (EPA, 1980a). In contrast, selenium has not been found to be carcinogenic, and the EPA has developed its water quality criterion using a threshold calculation. In the analysis, we used this threshold with a calculated distribution of selenium consumption in the exposed population to convert exposure levels to a measure of chronic health risk.

Section II of this report describes the general risk assessment framework, including the analysis procedure and potential exposure pathways. Section III discusses the case study and presents the results of the analysis. Appendix A contains the specific calculations of plant emission rates for both selenium and beryllium. Appendix B presents discussions of the health effects of both pollutants, including a derivation of our approaches to calculating chronic health risks.

RESULTS AND CONCLUSIONS

Plant emissions caused only extremely small increases in both selenium and beryllium concentrations in the Susquehanna River. In general, these increases were below measurable levels and remained so under virtually all circumstances and parameter values. In the Chillisquaque Creek, selenium concentrations remain below the EPA water quality criterion level of 0.010 mg/l in all cases except those involving *both* high emission rates and minimum flows. For cases having long-term mean flow rates, the concentration remains well below the water quality criterion. Similarly, for beryllium, calculated concentrations in

the Chillisquaque increased somewhat but remained low except for the high emission/low flow cases.¹

Assuming nominal values for parameters in the exposure calculations, for virtually all cases of the transport analysis, the incremental exposure rates associated with plant operation were minimal. For selenium, the plant-related exposure ranged from 0.02 percent of background in the nominal transport case to 0.2 percent of background exposure when river flow rates were at historical minimum values. These cases correspond to additional annual selenium intakes between 1.3×10^{-5} and 1.4×10^{-4} grams. In the same way, beryllium exposure rates are not significantly affected by plant emissions. In the nominal transport case the exposure increase associated with Montour discharges is only 0.01 percent of background levels. Under the minimum-flow assumptions, this incremental exposure increases to only 0.08 percent of background. These correspond to annual intake rate increases of 5.1×10^{-8} and 3.46×10^{-7} grams.

We performed sensitivity analysis on the results of the exposure assessment. For each pollutant, we calculated exposure rates for a number of alternative parameter values and assumptions. The results of these analyses for selenium indicate that although absolute exposure rates change dramatically from case to case, the relative contributions of plant emissions change little. Under the worst-case conditions in the sensitivity analysis, although the overall background exposure rate increases by a factor of five, the incremental exposure caused by Montour emissions is still only 0.04 percent of background. This type of behavior is true for all of the cases that we considered—the relative amount of selenium intake caused by Montour discharges varies only slightly with large changes in parameter values and assumptions in the analysis.

The results of the beryllium sensitivity analysis were similar. In the worst case, although background and plant-related exposure rates increase dramatically, the relative contribution of plant emissions remains small. Incremental exposure from plant operations is less than 0.08 percent of background under worst-case assumptions. Converted to annual intake, these assumptions thus lead to an intake rate of 1.1×10^{-6} grams.

The chronic health risk associated with selenium exposure is small. Under nominal conditions with average river flows, the lifetime risk of selenosis associated with plant emissions is less than 1.5×10^{-9} , or 1.5 in a billion. This compares to the often used EPA risk criterion levels

¹The 1985 monitoring program conducted by PP&L did not observe that actual beryllium levels in the Chillisquaque were affected significantly by plant discharges (Skinner, 1986).

of one in a million or one in one hundred thousand. Even in the sensitivity analysis cases, the risk associated with plant emissions remains below one in a million except for the worst-case analysis. In these cases, however, the calculated background exposure rates are far above the actual values measured and calculated in the literature.

The beryllium health risks change less between the various cases of the sensitivity analysis. In the nominal situation, the maximum incremental individual risk associated with any of the mean flow cases is 1.8×10^{-8} , or 1.8 in one hundred million. Although this risk increases with more pessimistic assumptions, it remains below one in a million even in the worst-case scenarios. Thus, in general, the health effects associated with both plant beryllium and selenium emissions under reasonable assumptions would be negligible.

II. RISK ASSESSMENT FRAMEWORK AND PROCEDURE

Fossil fuel power plants and other facilities can release toxic materials into the air, soil, surface water, and groundwater. Once emitted, these pollutants are dispersed and transported throughout the region, undergoing degradation and decay, adsorption, transformation, and biological uptake processes. Pollutants may also be transferred between media, such as when particulate material emitted from a power plant stack is deposited on the land. To determine the ultimate chronic health risk associated with such emissions, the method must be able to follow them through the environment and to calculate the long-term concentrations in all media of the region surrounding the emission source.

Pollutants in the environment reach the population through numerous exposure pathways that can be grouped into three general intake modes: inhalation, ingestion, and dermal absorption. Not all of these intake modes will be important for any given pollutant. Risk assessment procedures must be able to convert pollutant concentrations in the environment into exposure rates for each separate pathway. In addition to the direct pathways, such as inhalation or consumption of drinking water, exposure calculations may include pollutant transport through various stages of the food chain.

To assess whether the calculated exposure rates constitute a significant hazard to the population, the dose of each pollutant over time must be evaluated for each geographically and biologically distinct portion of the population. These doses are used with dose-response models to estimate the increased health risk over background of a particular response (or set of responses). Because we are concerned with chronic (long-term) health risks, we are particularly interested in dose-response models of chemical carcinogenesis.

The framework for this risk assessment process consists of five distinct stages: (1) source generation, (2) environmental transport, (3) exposure calculation, (4) toxicology and pharmacokinetics, and (5) dose-response modeling. To assess the risk associated with emission of a toxic pollutant from a specific source, we must use models, measured data, or a combination of both at each stage of the analysis.

The general risk assessment method is shown schematically in Fig. 2.1. A pollutant enters the environment through plant emissions. The emission rates serve as the inputs for environmental transport models

Fig. 2.1—Risk assessment approach

that calculate the distribution of pollutant concentrations in the different media (air, soil, groundwater, and surface water). As shown in the figure and explained below, the transport models may be either relatively simple and aggregated (Level I) or more detailed and complex (Level II). The pollutant concentrations determined by the transport models are used in the exposure analysis to calculate exposure rates as a function of location and type of exposure (e.g., inhalation or consumption of drinking water, animal products, or fish). These exposure rates can then be combined with local population data to determine a distribution of exposure rates for the region. Knowledge of the toxicology and pharmacokinetics of the pollutant are used with available data describing its health effects to develop appropriate dose-response models. Using the distribution of exposure rates as an input into one or more of these models, we calculate the expected chronic health risk associated with the pollutant emissions from the power plant.

Figures 2.2 and 2.3, taken from Bolten et al. (1983), present a schematic outline of the detailed risk assessment process.¹ Figure 2.2

¹In these figures, rectangles indicate processes or activities. Plant emissions or wastes are shown by slanted rectangles, and hexagons indicate pollutant concentrations either in the environmental media or parts of the exposure network. Circles are used to show pollution control points, where use of some treatment process or control technology could reduce the amount of pollutant reaching the general population. The arrows indicate the general flow of pollutants through the environment and the risk assessment process.

traces the toxic pollutant from the input coal (arrow entering at the left of the figure) through the source generation and environmental transport stages. The pollutant passes through the coal storage pile and the various processes within a power plant to be discharged into the environment. These discharges include emissions into the air, surface water, and soil or groundwater. Discharges into the different media are dispersed, transformed, and transferred between media. The figure indicates these general processes and the interactions between media that must be considered in determining the distribution of pollutant concentrations in the region surrounding the source. The models for atmospheric, overland, groundwater, and surface water transport are shown as darker rectangles in the figure.

Figure 2.3 begins with the environmental concentrations and shows the stages of exposure calculation, population description, and dose-response modeling. Pollutant concentrations in the environment are passed through food webs and other exposure pathways (both direct and indirect) to generate exposure rates by type of intake for the regional population. These exposure rates are combined with a regional population model or other information and with dose-response models for the pollutant to calculate the overall risk associated with the original pollutant emission levels.

The method has been designed to determine chronic health risks for the regional population. By limiting the assessment to chronic risks, we can restrict the study to long-term or average emission and exposure rates. This permits the use of steady-state models in the environmental transport and exposure calculations, considerably simplifying the calculation procedure.

Complex phenomena are involved in pollutant transport and transformation processes. These phenomena make it difficult to calculate the distribution of pollutant concentrations. Frequently, complex environmental transport models are used for these calculations. Such models attempt to incorporate all relevant phenomena and should be used in the detailed analysis of pollutant emissions problems. However, these models are expensive to use and require large amounts of time and data collection and preparation. Accordingly, before beginning the extensive data collection and analysis effort needed to apply these models it may be appropriate for the user to perform a first-order Level I (or screening) analysis.

This Level I analysis uses simplified calculations of pollutant concentrations to determine the relative importance of different media and exposure pathways and to calculate upper and lower bounds on concentrations and exposure rates. With this information, the analyst can develop a better understanding of the problem, direct study effort and

Fig. 2.2—Risk assessment framework—part I

Fig. 2.3—Risk assessment framework—part II

resources to the most important areas, improve selection of more complex models for subsequent Level II analysis, and reduce the need for subsequent iterations between complex models and other revisions of the study.

The Level I models used in this type of preliminary analysis, like the more detailed and complex Level II transport models, can be used to calculate pollutant concentrations and transfer rates between media. They are simplified models that use limited, aggregated data and can be run quickly and cheaply. Accordingly, they are appropriate for extensive sensitivity analysis and the development of approximate results and upper or lower bounds on concentrations, transfer rates, and subsequent exposure rates through the various pathways. Requiring limited amounts of data and other resources, these models can quickly produce results that permit the user to better understand and define the problem and solution approach and to isolate important aspects of a problem and determine where further detailed analysis is necessary.

The Level I models have been incorporated into a computer program that also contains routines for calculating exposure rates and chronic health effects using a variety of alternative dose-response models. This program, called WTRISK, is described in Bolten et al. (1985), which includes a discussion of the primary program variables, inputs and input formats, the types of output generated by the program, an example of the inputs and outputs associated with a typical analysis, and the listing of the computer code.

In the current analysis, we used the WTRISK program to investigate the chronic health risk associated with emissions of selenium and beryllium from an existing steam electric station. The source term (plant emissions) included direct discharges to the surface water and runoff from the coal storage pile. These emissions were used as inputs to the Level I surface water transport model, which calculated selenium and beryllium concentrations in the affected waters. Exposure routes that were considered included direct ingestion of treated drinking water, consumption of aquatic organisms from affected waterways, consumption of vegetable products grown on irrigated land, and consumption of animal products from animals that may have been exposed to contaminated vegetation or drinking water. The exposure rates calculated for these routes were combined with data describing the regional population and the potential health effects of selenium or beryllium ingestion to estimate the incremental health risk associated with plant emissions.

III. ANALYSIS OF THE CASE STUDY

In this case study we investigate the risks associated with alternative levels of selenium and beryllium emissions from a coal-fired power plant located in central Pennsylvania. More specifically, we consider the human health risk caused by exposure to selenium and beryllium from pathways associated with discharges to surface waters. The exposure routes identified in this analysis include consumption of drinking water and aquatic organisms from the affected waters, as well as animal and vegetable products produced on land irrigated with these waters.

Although the plant may also discharge the pollutants into the atmosphere and groundwater (via leaching from disposal ponds and landfills), the transfer of these emissions into surface waters should be much smaller than direct discharges.

In the analysis, potential plant selenium and beryllium emissions are estimated using mass balance equations and partitioning coefficients. The results of these calculations are compared with measured pollutant concentrations at the plant outfalls. This is done to verify our analysis of the pollutant's behavior within the power plant. If the particular pollutant emission was a problem, and controlling the discharge was insufficient to solve it, then knowing what happens to the pollutant within the plant itself may reveal the most efficient means of reducing emission levels when they are too high. However to minimize uncertainty in the analysis, measured concentrations are used as the source terms in the environmental transport calculations.

These transport calculations, made using the simplified procedures of the WTRISK program, determine the pollutant concentrations in the surface waters of the region surrounding the plant. The concentrations are the basis for calculating exposure rates and health effects in the exposed regional population.

The subsections that follow will briefly describe the power plant, site, and relevant risk analysis.

DESCRIPTION OF CASE STUDY

Plant Site

The power plant chosen for this study is the Montour steam electric station (SES), owned and operated by Pennsylvania Power & Light (PP&L). The plant is located on 4.71 km² in Derry Township, Montour County, Pennsylvania, approximately 1.5 km northeast of Washingtonville. As shown in Fig. 3.1, the plant site and treatment and disposal facilities are adjacent to the Chillisquaque Creek. The Montour plant draws its makeup water from the West Branch of the Susquehanna River. This water is pumped approximately 19 km to the plant. After the plant withdraws the amount needed for current operations, the remainder is pumped north to Lake Chillisquaque, which serves as an emergency makeup reservoir. The plant discharges into the Chillisquaque Creek. The area surrounding the power plant is predominantly rural, characterized by rolling hills and irrigated, dry land agriculture. Annual precipitation for the area is approximately 122 cm, and there are no major cities within 50 km of the power plant.

Detailed geological, meteorological, hydrological, and geographical data for this location were obtained from a variety of sources, including topographic maps. All sources are listed in the Bibliography at the end of this report and are referenced as appropriate in the discussion of the analysis.

Pollutants

This study examined the incremental change in health effects caused by selenium and beryllium emissions. Previous studies (Bolten and Resetar, 1986; Bolten et al., 1986) examined these pollutants using two exposure routes—consumption of drinking water and aquatic organisms. This study expands the analysis for both pollutants to consider two additional exposure routes. The additional routes are consumption of animal and vegetable products. Vegetable products grown in the region may be produced on land irrigated with the affected water. Animals raised in the area may either drink the affected water or consume feed that was contaminated through irrigation.

Two primary factors directed the analysis. First, both selenium and beryllium emissions have been of concern to the EPA, the Pennsylvania Department of Environmental Resources (DER), and PP&L. Second, although information about the sources and fate of selenium and beryllium within power plants is limited, it is generally better than the data related to organic priority pollutants and many other inorganic trace elements. Unfortunately, although the health effects

Fig. 3.1—Power plant location for case study

associated with some types of exposure to both pollutants have been studied, there is very little quantitative information about the human health effects of chronic exposure through ingestion of low doses. Specific health effects will be discussed later.

Power Plant

The Montour SES is a base load coal-fired power plant with a generating capacity of 1,641.20 MWe and a net continuous generating capability of 1,515.0 MWe. The characteristics of the plant are given in Table 3.1. A schematic diagram of plant water systems is shown in Fig. 3.2. Raw water for most uses is withdrawn from the Susquehanna River (or emergency supply in Lake Chillisquaque), passed through plant systems, and discharged to the Chillisquaque as overflow from the industrial waste treatment detention basin. Some additional water and collected leachate is discharged to the Chillisquaque from the plant storm water basin. These systems and other assumptions will be discussed in more detail as necessary below.

PLANT EMISSIONS

In this subsection we discuss and calculate pollutant emissions from the Montour SES. Plant pollutant discharges can be estimated using a general mass balance equation comparing inflows and outflows, including both potential sources and sinks within the plant. Stated simply, the balance is described by the basic equation

$$P_{in} = P_{out} + P_{rem} \quad (3.1)$$

Table 3.1

DESCRIPTION OF NOMINAL CASE STUDY POWER PLANT

Characteristic	Assumption
Generating capacity	
Number of units	2
Total capacity	1641.2 MWe (1,515.0 MWe net continuous)
Capacity factor	0.75
Thermal efficiency	0.36
Boiler type	Tangentially fired pulverized-coal burners; wet bottom furnace
Cooling system	Natural draft—crossflow—hyperbolic cooling towers (one per unit)
Air pollution control system	Cold-side electrostatic precipitators (one per unit)

where P_{in} = selenium/beryllium entering the plant (kg/day),
 P_{out} = selenium/beryllium exiting the plant (kg/day),
 P_{rem} = selenium/beryllium remaining in the plant (kg/day).

The following subsections will discuss the analysis using this approach, based on detailed calculations found in Appendix A.

Pollutant Inflows

The primary sources for incoming selenium and beryllium at the plant are the coal and plant makeup water (from the Susquehanna River). Plant processes add no other significant amounts of either selenium or beryllium to the system, although there may be small amounts of selenium or beryllium in the air used for combustion and some plant oils. Consequently, we assume that all discharges to the environment come from either or both of the primary sources (coal and plant makeup water). The coal input can be determined from the pollutant concentration in the raw coal used by the Montour plant and the average coal consumption rate. Similarly, the input from makeup water is the product of the background concentration in the makeup water (the Susquehanna River) and the average makeup water consumption rate.

The background concentration of selenium in the West Branch of the Susquehanna River has not been measured regularly. Old and limited measurements by PP&L found selenium concentrations in the plant makeup water that ranged between less than 0.002 mg/l and 0.012 mg/l. The validity of the 0.012 mg/l measurement has been cast into doubt by recent and more extensive measurements which found total selenium concentrations in the raw water ranging between less than 0.001 mg/l and 0.003 mg/l (Stine, 1985; Skinner, 1986). Although the recent data indicate a probable concentration of about 0.0014 mg/l, to be complete we assumed that the background selenium concentration in plant makeup water ranges between 0.002 mg/l and 0.010 mg/l. Using this information and plant data, we calculated the range of selenium inflows for Montour. These calculations are presented in Appendix A, and the results are summarized in Table 3.2.

The beryllium background concentration in the West Branch of the Susquehanna River is monitored quarterly by the U.S. Geological Survey (USGS). However, only one current measurement of beryllium for the North Branch, and not the West Branch, could be located (USGS, 1984). However, PP&L's Toxics Reduction Evaluation (TRE) measurement program (Skinner, 1986), required by the Montour plant NPDES Permit, measured beryllium and selenium levels in the plant

Fig. 3.2—Schematic diagram of power plant water system

raw water supply.¹ This water is drawn from the West Branch of the Susquehanna above the junction with the North Branch near Sunbury and should reflect the quality of part of the Susquehanna.

The TRE program found total beryllium concentrations in the raw water between less than 0.10 $\mu\text{g/l}$ and 1.80 $\mu\text{g/l}$, with a median of 0.10 and mean of 0.40 $\mu\text{g/l}$. Because the majority of measurements were below the detection limit of 0.10 $\mu\text{g/l}$, the median should be a more accurate representation of the long-term average concentration. The one measurement for the North Branch was also reported to be below the same detection limit (USGS, 1984). USGS data for the West Branch and other PP&L measurements of beryllium concentrations in the raw water intake at their Sunbury Steam Electric Station are consistent with these results, although detection limits in these studies were generally higher, and it is difficult to compare the results. To be conservative, we assumed that the background beryllium concentration in the plant makeup water ranged between 0.10 and 0.40 $\mu\text{g/l}$. Using this information and plant data, we calculated the range of beryllium inflows for Montour. These calculations are presented in Appendix A, and the results are summarized in Table 3.2.

Table 3.2

SELENIUM AND BERYLLIUM INPUT RATES

(kg/day)

Source	Selenium Concentration in Makeup Water ($\mu\text{g/l}$)		Beryllium Concentration in Makeup Water ($\mu\text{g/l}$)	
	2.0	10.0	0.10	0.40
Intake water	0.188	0.942	0.0094	0.0377
Coal	44.778	44.778	18.658	18.658
Total	44.967	45.720	18.667	18.695

NOTES: Input rates are based on a makeup water flow rate of 1.09 m^3/sec .

¹The NPDES permit for the Montour SES, issued in 1976 and reissued in September 1984, required that a toxics reduction evaluation be carried out for several trace elements and organic compounds. To comply with this requirement and subsequent determinations by the Pennsylvania Department of Environmental Resources, PP&L undertook a comprehensive chemical characterization of the in-plant waste streams, wastewater discharges, and receiving stream (Chillisquaque Creek). Samples were collected, processed, and analyzed following approved EPA analytical procedures by outside laboratories. The results of the TRE program were submitted to the DER as required in the permit.

Pollutant Sinks and Outflows

The selenium or beryllium that enters the power plant can either remain there in several different locations (sinks), or can be discharged in various ways. Although the distribution of selenium and beryllium (along with other trace elements) in power plants has been studied, the behavior of these pollutants is not yet well understood. At Montour, pollutant concentration in some plant effluents has been measured, but these measurements are limited and do not include all important waste streams. Consequently, although there are some data, it would be useful for the analysis to know how each pollutant entering Montour behaves within the plant and how plant operations affect the pollutant concentrations in the various types of discharge.

Selenium and beryllium entering the system in the coal may be leached out in runoff from the coal pile, discharged through the stack in gaseous or particulate form, or retained in either the collected fly ash or bottom ash. Coal pile runoff at the Montour SES is collected and discharged into the miscellaneous plant waste basin. Dry fly ash systems transport collected material to an ash storage area for disposal. Runoff and leachate from the disposal sites, which may contain selenium or beryllium, are collected and also returned to the miscellaneous plant waste basin. Bottom ash and pyrites are collected and transported in a slurry which is pumped into the ash basin, as shown in Fig. 3.2. The miscellaneous plant wastes, including those from water treatment processes, rinses following metal cleaning, and area drains are also treated in the miscellaneous plant waste basin.

The overflows from the ash basin and miscellaneous plant waste basin enter the detention basin to be mixed with the wastes there. The overflow from the detention basin is discharged into the Chillisquaque Creek. As a result, for selenium and beryllium entering the plant in the coal or makeup water, the primary outflows include only the stack gas or discharge in the overflow from the detention basin.² The potential sinks within the plant include the (1) fly ash disposal site, (2) ash basin, (3) miscellaneous plant waste basin, and (4) detention basin. All entering selenium or beryllium should either leave the plant or be retained in one of these sinks.

Appendix A presents calculations of plant emission rates and the behavior and distribution of selenium and beryllium within the system.

²As shown in Fig. 3.2, seepage from the ash basin is collected and returned to the storm water basin, from which it is discharged into the Chillisquaque. Although over the long term the flow from the storm water basin has averaged less than 0.002 m³/s, measurements in 1985 found an average flow of 0.0132 m³/s. Consequently, discharge from the storm water basin should be considered as a potential source of selenium or beryllium.

Because the distribution of these pollutants has not been extensively measured within Montour, some of these calculations have been performed using parameters with a range of values. The range should include the reasonable upper and lower bounds on the parameter values. Using the results of Appendix A, we can calculate the mass balance and corresponding detention basin discharge concentrations for selenium and beryllium under varying conditions. These results are presented in Tables 3.3 and 3.4.

The results depend strongly on the fraction of the pollutant that passes through the ash basin into the detention basin (parameter k). Unfortunately, there is large uncertainty associated with this value. For selenium this parameter was varied from 0.05 to 2.0. Normally this parameter would not have a value greater than 1.0, unless the ash basin had accumulated large amounts of selenium in its sediment. This could have happened while the plant fly ash was being deposited there, before installation of the dry fly ash transport and disposal system. After selenium inflow dropped significantly, some selenium from basin sediments could return to the water and subsequently be discharged into the detention basin. This is the meaning of the negative values in Table 3.3. The results in the table are consistent with the limited selenium measurements of the Montour detention basin discharges. The measured value of 0.017 mg/l (for a probable makeup water selenium concentration of 0.012 mg/l) corresponds to the range of parameter values between 0.50 and 1.00. More specifically, it is consistent with the case in which half of the selenium entering the ash basin is discharged to the detention basin and all of the detention basin selenium is discharged to the Chillisquaque in the overflow.

Beryllium measurements in the bottom ash transport water and the ash basin discharge give an approximate value of 0.125 for the fraction of beryllium that passes through the ash basin into the detention basin (parameter k). However, the concentration in the transport water may not reflect the total amount of beryllium entering the basin in the bottom ash. Calculations discussed in Appendix A, although highly uncertain, suggest that the value of k may be less than 0.01. Because of this inconsistency, we have chosen to rely exclusively on measured discharge concentrations in the analysis. As a result, the mass balance calculations shown in Table 3.4 only describe a possible partitioning of beryllium in the Montour power plant.

These results are consistent with the PP&L measurements of the beryllium concentration in Montour detention basin discharges. The median value of 0.20 $\mu\text{g/l}$ (for a makeup water concentration of 0.10 $\mu\text{g/l}$) corresponds roughly to values of k and m near 0.005 and 1.0, respectively. This represents a situation in which almost all of the

Table 3.3

SELENIUM MASS BALANCE AND DISCHARGE CONCENTRATIONS

Item	Parameter Values		Makeup Water Concentration ($\mu\text{g/l}$)	
	k	m	2.0	10.0
Inflow (kg/day)				
Total			44.967	45.720
Sinks (kg/day)				
Fly ash landfill			43.186	43.186
Ash basin	0.05		0.165	0.728
	0.50		0.087	0.383
	1.00		0.000	0.000
	2.00		-0.174	-0.776
Detention basin	0.05	0.05	0.039	0.166
	0.50	0.50	0.060	0.260
	0.50	1.00	0.000	0.000
	1.00	1.00	0.000	0.000
	2.00	1.00	0.000	0.000
Outflow (kg/day)				
Stack			1.566	1.566
Detention basin	0.05	0.05	0.002	0.009
	0.50	0.50	0.060	0.260
	0.50	1.00	0.119	0.520
	1.00	1.00	0.206	0.913
	2.00	1.00	0.380	1.669
Concentration ($\mu\text{g/l}$)				
Discharge	0.05	0.05	0.0001	0.0003
	0.50	0.50	0.0017	0.0076
	0.50	1.00	0.0035	0.0152
	1.00	1.00	0.0060	0.0268
	2.00	1.00	0.0111	0.0489

NOTE: k = fraction of selenium entering the ash basin that passes into the detention basin; m = fraction of selenium entering the detention basin that is discharged to the Chillisquaque.

beryllium settles out in the ash basin and the remainder passes through the detention basin to be discharged into the Chillisquaque. This is also consistent with the formation of beryllium oxide during coal combustion and its subsequent presence in the ash. Beryllium oxide should be highly resistant to chemical attack and would not be expected to dissolve significantly in the ash transport water.

Table 3.4

BERYLLIUM MASS BALANCE AND DISCHARGE CONCENTRATIONS

Item	Parameter Values		Makeup Water Concentration ($\mu\text{g/l}$)	
	k	m	2.0	10.0
Inflow (kg/day)				
Total			18.667	18.695
Sinks (kg/day)				
Fly ash landfill			17.554	17.554
Ash basin	0.001		0.706	0.728
	0.005		0.703	0.726
	0.01		0.700	0.722
	0.02		0.693	0.715
Detention basin	0.001	0.50	0.0021	0.0041
	0.005	0.50	0.0036	0.0056
	0.005	1.00	0.000	0.000
	0.010	1.00	0.000	0.000
	0.020	1.00	0.000	0.000
Outflow (kg/day)				
Stack			0.404	0.404
Detention basin	0.001	0.50	0.0021	0.0041
	0.005	0.50	0.0036	0.0056
	0.005	1.00	0.0071	0.0112
	0.010	1.00	0.0106	0.0148
	0.020	1.00	0.0177	0.0221
Concentration ($\mu\text{g/l}$)				
Discharge	0.001	0.50	0.0627	0.1207
	0.005	0.50	0.1043	0.1635
	0.005	1.00	0.2083	0.3270
	0.010	1.00	0.3121	0.4340
	0.020	1.00	0.5192	0.6476

NOTE: k = fraction of selenium entering the ash basin that passes into the detention basin; m = fraction of selenium entering the detention basin that is discharged to the Chillisquaque.

The results in Tables 3.3 and 3.4 have important implications. Although the coal is responsible for most of the selenium and beryllium entering the plant, the discharge concentration is very sensitive to the makeup water quality. A large portion of these pollutants entering Montour from the coal is captured by the fly ash collection system and remains in the fly ash landfill. On the other hand, most of the pollutant entering the plant in the makeup water ultimately ends up in the ash basin and detention basin. Thus, even though its inflow rate is

much smaller, the makeup water selenium and beryllium dominates the outflow calculations.

DESCRIPTION OF ANALYSIS

Under normal operations, the power plant emits selenium and beryllium to the air, surface water, and groundwater. Atmospheric emissions consist of stack gases; surface water emissions include discharges from the combined ash ponds and coal pile runoff; and groundwater emissions comprise leaching from the combined ash ponds and ash disposal landfill. This study investigates the risk associated with surface water pollution. For the Montour SES, the dominant source of surface water contamination is the direct discharge of detention basin overflow into the relevant water bodies (the Chillisquaque Creek and Susquehanna River).³

Atmospheric Emissions

Atmospheric discharges of either pollutant should not be a significant source of exposure for the regional population. The detailed analysis in Appendix A is summarized in Table 3.5. This analysis scaled previously calculated atmospheric concentrations to account for differences in stack emission rates.

These levels of exposure are not significant when compared to ingestion exposure rates. Moreover, only those people living close to the power plant would inhale even this much selenium or beryllium. As will be discussed below, the population exposed to Montour surface water discharges lives relatively far from the power plant. At such distances, the inhalation exposure would be totally insignificant. However, if the analysis had found that exposure resulting from Montour surface water discharges was significant, it might then have been necessary to return to this point and further consider the incremental exposure from air emissions.

³The plant also discharges overflow from the storm water basin. Because the estimated discharge rate from this basin has recently increased, it must be included as a potential source of beryllium, using measured concentrations from the TRE program (Skinner, 1986). The storm water basin discharge was not included in the selenium analysis because at that time the discharge rate was smaller. In addition selenium concentrations in that basin are significantly smaller than concentrations in the detention basin, which is not true for beryllium.

Table 3.5

APPROXIMATE EXPOSURE FROM INHALATION

Sample pollutant	
Discharge rate	0.260 kg/day
Atmospheric concentration	4.74×10^{-12} gm/m ³
Selenium—Montour	
Discharge rate	1.57 kg/day
Atmospheric concentration	2.86×10^{-11} gm/m ³
Exposure rate	0.62×10^{-3} µg/day
Beryllium—Montour	
Discharge rate	0.40 kg/day
Atmospheric concentration	7.29×10^{-12} gm/m ³
Exposure rate	0.16×10^{-3} µg/day
Adult male inhalation rate	21.9 m ³ /day

Risk Assessment Framework

We have earlier discussed the primary sources of selenium and beryllium discharged to surface waters from the power plant. Using the simplified surface water transport model in the WTRISK program, we calculated pollutant concentrations in the Chillisquaue Creek and Susquehanna River for a base case and several sensitivity cases. The base case was defined to have nominal emission rates from the plant, long-term average stream flows, and the normal or expected values for other parameters and input data.⁴ The calculated concentrations for this case and the sensitivity analysis cases were used to calculate the corresponding exposure rates for the affected regional population. The affected population are those who consume fish caught in the Susquehanna or Chillisquaue, drink water drawn from the Susquehanna, consume vegetable products irrigated from the Susquehanna, or consume animal products from animals that also may drink Susquehanna water or consume locally grown vegetable products.

In this study we also performed a sensitivity analysis that included a number of critical variables. The selection of these variables was guided by the results of previous sensitivity analyses performed for arsenic emissions from similar coal-fired power plants and selenium and beryllium emissions from Montour in the more limited studies (Bolten et al., 1983, 1985, and 1986; Bolten and Resetar, 1986). In the

⁴The *base case* is considered to be the starting point for the analysis. It generally corresponds to the existing situation and includes the best estimates for parameter values and inputs. Thus, the base case or nominal pollutant discharge corresponds to the median concentrations measured for the detention and storm water basin outputs.

analysis, the surface water transport calculations were separated from the exposure analysis. In other words, we calculated the pollutant concentrations in the Chillisquaque Creek and Susquehanna River using the simplified surface water transport model in WTRISK. The concentrations determined by these calculations were inputs to the exposure analysis which was then used to determine the final health risks.

Dividing the analysis in this manner made it possible to perform a sensitivity analysis on each part of the problem separately. In general, this simplifies the sensitivity analysis and makes it easier to understand how specific parameter variations affect the final results. It is a logical method of addressing the problem, because various organizations may be concerned with different stages of the analysis. For example, utility industry discharge permit limitations are based on plant emission rates and the corresponding pollutant concentrations in the receiving waters. Consequently this group would be particularly interested in the parameter variations that may significantly alter receiving water concentration levels. On the other hand, potential health risks of discharges depend on the exposure rates. Although these rates are also functions of the pollutant concentrations, they are frequently more sensitive to other parameters in the exposure analysis.

The three key parameters in the surface water transport calculations were (1) plant discharge rate, (2) pollutant removal rate, and (3) receiving water flow rates. In the sensitivity analysis, we used a range of values for these parameters and calculated the corresponding pollutant concentrations in the receiving waters. Exposure rates for these cases were then calculated using the nominal values of the exposure analysis parameters. As a consequence, the results should indicate how variations in the transport calculations affect the nominal health risks.

The corresponding procedure was used for the exposure calculations. Pollutant concentrations were calculated for several key surface water cases then exposure rates and health effects using different values for important exposure parameters were computed. The details of this analysis will be discussed below. The following subsections describe the various stages of the analysis.

Regional Description

As the first step in calculating pollution concentrations in the environment and population exposure rates, one must describe the region around the plant. For the preliminary analysis this process includes:

- Partitioning the region into subregions.
- Describing the distribution of population by subregion.
- Determining hydrological characteristics of the affected waterways.
- Describing the population characteristics necessary to calculate exposure rates.

In this subsection we discuss the definition of subregions and distribution of population. The remaining characteristics of the region will be considered in the description of the analysis itself.

Because this analysis did not consider atmospheric transport there was no need to partition the entire region into a regular pattern of subregions. Instead, the subregions could be defined in terms of the exposed population and waterway characteristics. The waterways for this analysis were limited to the Chillisquaque Creek and a portion of the Susquehanna River. The affected portion of the Susquehanna River begins at the entrance of the Chillisquaque Creek above the confluence of the West and North Branches, and ends at Chesapeake Bay. This region includes parts of Pennsylvania and Maryland, although water use and exposed population in Pennsylvania are much greater than in Maryland.

The total exposed population must include all people affected by one or more of the exposure pathways considered. In this analysis, the population that could consume animal and vegetable products affected by plant discharges is much larger than the population drinking water from the Susquehanna. Accordingly, we assumed that the exposed population consisted of all people in the region who could consume contaminated animal or vegetable products. This population also ate aquatic organisms caught in affected waterways. In the total population, we identified those people who also consumed drinking water from the Susquehanna. This subgroup can be partitioned into smaller groups specified by the intake location for their drinking water companies. All subgroups of the population were partitioned into age groups for the exposure analysis. Each cohort will differ in terms of how much drinking water, fish, animal products, and vegetable products they consume. The specific cohorts considered in the analysis were infants (<1 year of age), children (1-11), teenagers (11-17), and adults (>17).

Fourteen reaches were defined for the affected waterways. These reaches were determined by the hydrological conditions of the affected waterways. Conditions within each reach were kept as homogeneous as possible. For each reach, we defined from zero to two subregions, depending on the character of the exposed population. If no one living

along a reach would be exposed to either pollutant through any of the considered pathways, no subregion was specified. One subregion was used if the population could consume fish, animal, or vegetable products affected by plant discharges. A second subregion was identified for some fraction of this population, if they also consumed drinking water taken from the reach.

Population Consuming Agricultural Products

The total exposed population in the analysis was assumed to be the population living in counties bordering the affected sections of the Chillisquaque Creek and Susquehanna River. This total of 1,478,785 people was drawn from Dauphin, Cumberland, Lancaster, Montour, Northumberland, Perry, Snyder, and York Counties in Pennsylvania and Cecil and Hartford Counties in Maryland. Populations were determined from 1980 census data. Lancaster and York Counties, located south of Harrisburg, are the largest in population and agricultural production. The population in each county was then divided between stream reaches if the county bordered more than one reach. Table 3.6 presents the total population by age group, reach, and county.

Population Consuming Drinking Water

This study is concerned with the population that regularly consumes Susquehanna River water below the confluence of the West Branch and the Chillisquaque Creek. The first step in defining this population is to locate the water companies that regularly withdraw from the river.⁵ Such information on the population that could consume drinking water from the Susquehanna River was available from a computer data base (the Water Use Data System) of the Pennsylvania DER.

In general, much of the regional drinking water is taken from sources other than the Susquehanna. These sources include smaller rivers and streams, wells, and reservoirs. In some instances the Susquehanna is used for emergency supply only. Table 3.7 lists the 14 water suppliers that regularly use the Susquehanna River, and includes other relevant information about their operations and service areas. This additional information and data about daily water usage, other

⁵Several water companies use the Susquehanna River as a secondary or emergency source and withdraw little or no water from the river in most years. Because their use is intermittent, limited, and temporary, these companies were not considered to be important in an analysis of chronic health effects. These users might become important during a prolonged drought. During low flow years, dilution of either selenium or beryllium would be reduced, concentrations would be higher, and the intermittent intakes would more likely be in use as the smaller primary sources dried up.

Table 3.6
TOTAL POPULATION BY COUNTY AND REACH

County	Reach	Population Group				
		Total	Infant	Child	Teenager	Adult
Pennsylvania						
Montour	2	16,675	217	2,351	1,668	12,440
Northumberland	3	25,095	326	3,538	2,510	18,721
Northumberland	4	25,095	326	3,538	2,510	18,721
Northumberland	5	25,095	326	3,538	2,510	18,721
Snyder	5	16,792	218	2,368	1,679	12,527
Northumberland	6	25,095	326	3,538	2,510	18,721
Snyder	6	16,792	218	2,368	1,679	12,527
Dauphin	7	116,109	1,509	16,371	11,611	86,617
Perry	7	35,718	464	5,036	3,572	26,646
Cumberland	8	178,541	2,321	25,174	17,854	133,192
Dauphin	8	116,109	1,509	16,371	11,611	86,617
Lancaster	9	72,469	942	10,218	7,247	54,062
York	9	62,593	814	8,826	6,259	46,694
Lancaster	10	72,469	942	10,218	7,247	54,062
York	10	62,593	814	8,826	6,259	46,694
Lancaster	11	72,469	942	10,218	7,247	54,062
York	11	62,593	814	8,826	6,259	46,694
Lancaster	12	72,469	942	10,218	7,247	54,062
York	12	62,593	814	8,826	6,259	46,694
Lancaster	13	72,469	942	10,218	7,247	54,062
York	13	62,593	814	8,826	6,259	46,694
Maryland						
Cecil	14	60,430	846	8,762	6,526	44,295
Hartford	14	145,930	2,043	21,160	15,760	106,967
Total		1,478,785	19,431	209,334	149,529	1,100,491

primary sources, the number of domestic connections, population served, and water treatment processes were taken from the annual water supply reports sent to the DER Bureau of Water Management. A limited description of the Maryland water users was obtained from the Susquehanna River Basin Commission (Jackson, 1984).

The drinking water population for each water company was obtained by multiplying the number of domestic connections by 3.5 (Gast, 1985). These results were verified by comparison with 1980 census data for

Table 3.7

POTABLE WATER SUPPLIERS ON THE LOWER SUSQUEHANNA RIVER

Water Supplier	Water Use (mgd) ^a	% Susquehanna	Population Served	Water Treatment ^b
Pennsylvania				
Columbia Water Co.	1.207	100	13,733	D,FL,C,S,F,CC,F/M
Dalmatia Water Co.	0.022	100	585	
Dauphin Consolid. Water Supply	7.600	3.9	59,163	D,FL,C,S,F,CC,SF,TO
Herndon Municipal Water Auth.	0.040	100	704	
Lancaster Municipal Water Auth.	12.412	59	109,630	D,FL,C,S,F,FU,CC, F/M,TO
Northumberland Water Auth.	0.018	100	319	
PP&L—Holtwood	0.158	100	32	D,FL,C,S,F
Safe Harbor Water Power Co.	0.015	100	112	D,FL,C,S,F,CC
Shamokin Dam Water Auth.	0.148	100	1,590	D,FL,C,S,F,TO
Steelton Municipal Waterworks	1.247	100	6,712	D,FL,C,S,F,CC,TO
Sunbury Municipal Water Auth.	1.700	15	13,170	D,FL,C,S,F,FU,CC, F/M,TO
Wrightsville Water Supply Co.	0.290	100	3,623	D,FL,C,S,F
Maryland				
City of Havre de Grace	1.750		8,763	
Perryville	0.400		2,018	
Port Deposit	1.057		664	
Veterans Administration	0.230		1,150	
Intermittent Users				
Chester Water Auth.	15.640			
Red Lion Municipal Water Auth.				

SOURCES: DER (1984a and 1984b).

^aMillion gallons per day.^bTreatment Processes: C—coagulation; CC—corrosion control; D—disinfection; F—filtration; FL—flocculation; FU—fluoridation; F/M—iron and manganese removal; S—sedimentation; SF—softening; TO—taste and odor control

the municipalities served. When the water company served clearly identifiable locations and the census data agreed with the connection calculation, the census data were used for the population. If no specific census region could be found to correspond with the service area, the connection calculation was used for the population. Census information was then used to divide these total population results into the age groups discussed earlier (USDOC, 1982a, 1982b, 1982c, and 1983).

All water taken from surface sources must generally be thoroughly treated. This treatment normally includes disinfection, flocculation, coagulation, sedimentation, and filtration. In many cases, softening, corrosion control, iron and manganese removal, and taste and odor treatment may be required for some fraction of the water.

The final distribution for the drinking water population is presented in Table 3.8. In this table, the water companies and populations served are arranged by the downstream distance of their inlets from the confluence of the Chillisquaque and the West Branch of the Susquehanna. The populations are grouped into the reaches defined for the exposure analysis.

Fishing

Fishing along the Susquehanna River is for recreational purposes. Some estimates maintain that 85 percent of the fish caught are then returned (NRC, 1981). However, many are kept for consumption and these must be considered in the exposure calculations. The majority of anglers are from local areas, thus it may be reasonable to assume that the population that consumes Susquehanna drinking water and agricultural products also consumes fish caught from the river. It is possible that none of the people exposed through the other pathways consume fish obtained from the affected waterways. However, to insure that

Table 3.8

POPULATION DISTRIBUTION FOR CONSUMPTION OF DRINKING WATER

Water Company	Downstream Distance (km)	Reach	Population Group				Total
			Infant	Child	Teenager	Adult	
Northumberland	9.0	4	4	45	32	238	319
Sunbury	10.0	5	175	1,853	1,321	9,821	13,170
Shamokin	10.4	5	21	224	159	1,186	1,590
Herndon	26.6	6	9	99	71	525	704
Dalmatia	34.8	6	8	82	59	436	585
Dauphin	85.8	8	787	8,324	5,934	44,118	59,163
Steelton	99.4	8	89	945	673	5,005	6,712
Wrightsville	139.3	10	48	510	363	2,702	3,623
Columbia	140.2	10	183	1,938	1,381	10,271	13,773
Lancaster	143.8	10	1,458	15,425	10,996	81,751	109,630
Safe Harbor	157.0	12	1	16	11	84	112
Holtwood	169.4	13	0	5	3	24	32
Port Deposit	203.0	14	9	96	72	487	664
Perryville	208.9	14	28	292	219	1,479	2,018
Veterans Admin	210.0	14	16	166	125	843	1,150
Havre de Grace	210.2	14	122	1,269	950	6,422	8,763
Total			2,958	31,289	22,369	165,392	222,008

this work does not underestimate the potential health effects from all activities, we assume that the populations overlap completely. In other words, all people drinking water or eating affected animal and vegetable products are either fishermen or members of the families of fishermen and consume fish from the river.

Population Distribution

The total population considered in this analysis is 1,478,785 people, all of whom consume fish as well as animal and vegetable products affected by plant discharges, and 222,008 of whom consume drinking water withdrawn from the Susquehanna. Table 3.9 presents the final population distribution by reach, separated into groups that are or are not exposed through their drinking water.

Surface Water Transport

In this analysis, we used the simplified transport model in WTRISK to simulate the migration and fate of selenium and beryllium entering the Chillisquaue Creek through direct discharge from the power plant. The analysis did not incorporate the pollutant deposition from runoff or the pollutant interchanges between the surface water and groundwater components of the region. Preliminary investigation and calculations in recent work (see Bolten et al., 1985) indicated that both runoff and the net interchanges would be small.

The Chillisquaue Creek, located in Montour County, is a stream with a low average flow. Portions of the creek are slow moving and deep and others are swift moving and extremely shallow. The flow of the Chillisquaue is closely tied to precipitation levels. As a consequence, its flow varies widely, as shown in Fig. 3.3, which includes the contributions from the Montour detention basin. During heavy rainfall, the creek can swell its banks and may scour accumulated sediment from the bottom in many locations.

Water for the Montour plant is pumped approximately 19.2 km from the Susquehanna River at Watsontown. By means of a "T" junction at the plant, sufficient makeup water can be withdrawn to meet plant requirements, whereas the remaining water is pumped to Lake Chillisquaue. Lake Chillisquaue was formed to provide an emergency water supply to the power plant and feeds into the Middle Branch of the Chillisquaue Creek. Susquehanna water is pumped as needed, with no records of how much is pumped to either location. However, Pennsylvania Power & Light is required to maintain a minimum discharge from the lake of $0.023 \text{ m}^3/\text{s}$ (WPRB, 1969). The Middle

Table 3.9
POPULATION DISTRIBUTION BY REACH AND SUBREGION

Region	Reach	Population Group					Exposure Source
		Total	Infant	Child	Teenager	Adult	
1	2	16,675	217	2,351	1,668	12,440	Food
2	3	25,095	326	3,538	2,510	18,721	Food
3	4	319	4	45	32	238	All
4	4	24,776	322	3,493	2,478	18,483	Food
5	5	14,760	196	2,077	1,480	11,007	All
6	5	27,127	349	3,829	2,709	20,241	Food
7	6	1,289	17	182	130	962	All
8	6	40,598	528	5,724	4,059	30,286	Food
9	7	151,827	1,974	21,408	15,183	113,263	Food
10	8	65,875	876	9,269	6,607	49,123	All
11	8	228,775	2,954	32,277	22,858	170,686	Food
12	9	135,062	1,756	19,044	13,506	100,756	Food
13	10	127,026	1,689	17,873	12,740	94,724	All
14	10	8,036	67	1,171	766	6,032	Food
15	11	135,062	1,756	19,044	13,506	100,756	Food
16	12	112	1	16	11	84	All
17	12	134,950	1,755	19,028	13,495	100,673	Food
18	13	32	0	5	3	24	All
19	13	135,030	1,756	19,039	13,503	100,732	Food
20	14	12,595	176	1,823	1,366	9,231	All
21	14	193,765	2,713	28,099	20,921	142,031	Food
Total All		222,008	2,958	31,290	22,369	165,392	
Total Food		1,256,777	16,472	178,045	127,161	935,099	
Total		1,478,785	19,431	209,334	149,529	1,100,491	

Branch of the creek and the East Branch meet above the plant discharge, forming the main stem of the creek. The West Branch of Chillisquaque Creek joins above the town of Washingtonville and below the plant discharge. The Chillisquaque meanders approximately 34 km from the plant to its mouth, which is 9.5 km above the confluence of the West and North Branches of the Susquehanna River.

Above the plant the minimum stream flows can be as low as 0.023 m³/s from both the Middle and East Branches. Similarly, the minimum flow on the West Branch can typically be as low as 0.028 m³/s. In contrast to this, the maximum flows have been approximately

Fig. 3.3—Historical flow rates for the Chillisquaque Creek

16.00 m³/s above the Montour plant and 24.00 m³/s downstream (Skinner et al., 1985). Stream velocities can vary greatly throughout (Snyder, 1977; GCEC, 1978/1979). These conditions make modeling the stream extremely difficult, particularly in view of the relatively limited information on discharges and pollutant concentrations.

There are no surface water withdrawals from the Chillisquaue Creek. From aerial photographs and visual inspection, it is clear that most of the land around the creek is primarily used for farming. Main crops harvested are corn and soybeans. Land owned by the power company and leased to farmers is not subject to any more stringent regulations than state laws for such an area. Consequently, crops can be, and are, planted up to the banks of the creek and unspecified fertilizers and pesticides may be employed.

The Chillisquaue Creek was divided into three reaches, corresponding to the sections (1) above the plant, (2) between the plant and the West Branch, and (3) between the West Branch and the Susquehanna River. Plant discharges affect the flow and quality of the second and third reaches. There are no withdrawals from any reach, and the area and velocity of the creek vary frequently in each of the reaches. Selenium or beryllium can enter the Chillisquaue from the Montour plant or, conceivably, in runoff from the farming area around the plant. These emissions pass downstream and enter the West Branch of the Susquehanna River south of Lewisburg (see Fig. 3.1).

The West Branch joins the North Branch of the Susquehanna at Northumberland, to flow south through eastern Pennsylvania into Chesapeake Bay. The surface water analysis considered the Chillisquaue and those parts of the West Branch and Susquehanna River downstream of the Chillisquaue junction. It did not include Chesapeake Bay for several reasons. First, the Bay is not used as a drinking water source because its salinity is too high. Second, it is an estuarine system which receives fresh water input from many different rivers. Although the Susquehanna provides about 50 percent of this input, any pollutant reaching the Bay from the Montour SES would be significantly diluted. Third, because the Bay is a moderately stratified estuary having complex hydrology, it represents a difficult analysis problem beyond the scope and resources of the study.

The Susquehanna River was divided into 11 reaches of varying length, based on the location of river junctions, dams, and drinking water withdrawal points. The river is free flowing from the mouth of the Chillisquaue Creek to just below Harrisburg. Within this region are reaches 4, 5, 6, 7, and 8. Reach 4 extends from the mouth of the Chillisquaue to the confluence of the West and North Branches of the Susquehanna. The Northumberland Water Authority withdraws from

this reach, thus constituting the first drinking water use below the plant. Reach 5 extends from Sunbury to Selinsgrove and includes the surface water withdrawal of Sunbury Municipal Water Authority. Reach 6 terminates at the border of Juniata and Perry Counties. Included in this reach are the surface water withdrawals of Dalmatia Water Company and Herndon Municipal Water Authority and the incremental flow from Penns Creek. No withdrawals or additions of flow are included in reach 7. Reach 8 begins at the mouth of the Juniata River and includes withdrawals from Dauphin Consolidated Water Supply and Steelton Municipal Waterworks.

From Harrisburg south, the Susquehanna is controlled by a series of four dams: York Haven, Safe Harbor, Holtwood, and Conowingo. The flow through the reservoirs is thermally mixed and no stratification exists. Therefore, analytically, these sections may be treated as slow moving rivers rather than lakes (Roy F. Weston, Inc., 1982). Sediment is trapped behind these dams and in periods of unusually high flows it becomes dislodged and travels downstream to Chesapeake Bay. The withdrawals from Wrightsville Water Supply Company, Columbia Water Company, and Lancaster Municipal Water Authority all occur within reach 10 which begins at Columbia and extends to the beginning of the Safe Harbor Reservoir. Each dam is included within its own reach. Reach 12 begins at Safe Harbor, 13 at Holtwood, and 14 at Conowingo.

Information on the flows, velocities, and cross-sectional areas along the Susquehanna was provided by the USGS district office at Harrisburg. Long-term average flows are computed by the USGS for three locations along the river: Sunbury, Harrisburg, and Marietta (USGS, 1984). Although the long-term mean flow data could be used in the analysis, the historical minimum flows were inconsistent for the three points. Minimum flows have occurred at one or more of these locations typically when an upstream dam has been shut down. River velocity and area data for the above locations were taken from discharge measurement worksheets used by USGS field officers. Pollutant information was taken from USGS STORET data and recent measurements made by PP&L (Skinner, 1986). Finally, procedures for operating the dams downstream from Harrisburg were obtained from water quality studies done for the power companies operating the dams (Roy F. Weston, Inc., 1982 and 1983; Potera et al., 1982). These reports contain time of travel studies, information on daily discharges, and general reservoir conditions.

The lengths, locations, and flow data for the reaches on the Chillisquaque and Susquehanna are summarized in Table 3.10. For each reach pollutant concentrations were calculated for two or more

downstream points, although all point source discharges and major tributaries were located at the beginning of a reach. Small tributaries were treated as distributed sources along the length of the reach.

The behavior of both pollutants in natural waters is not well understood. Moreover, the chemical form of the selenium and beryllium discharged from power plants or reaching waterways through runoff may vary and has not been studied adequately. As a consequence, there are no accepted values for the surface water decay or removal coefficients. Accordingly, we used an intermediate value for the removal coefficient for each pollutant, to reflect removal through sedimentation processes and conversion to relatively nontoxic states. This parameter was varied in the sensitivity analysis to determine how

Table 3.10

DEFINITION OF RIVER REACHES

River/Reach	Downstream Distance ^a (km)	Length (km)	Incremental Flow ^{b,c} (m ³ /s)		Velocity (m/s)	
			Minimum	Mean	Minimum	Mean
Chillisquaue						
1	0.0	4.5	0.023	0.4	0.04	0.2
2	4.5	1.5	0.0	0.0	0.04	0.2
3	6.0	28.0	0.028	0.43	0.04	0.2
Susquehanna						
4	34.0	9.5	11.0	306.1	0.16	0.40
5	43.5	10.5	16.3	444.9	0.16	0.40
6	54.0	20.0	0.2	12.4	0.16	0.45
7	74.0	32.0	0.0	0.0	0.16	0.45
8	106.0	33.0	5.5	121.6	0.17	0.50
9	139.0	30.0	0.4	16.6	0.15	0.45
10	169.0	9.0	0.0	0.0	0.10	0.43
11	178.0	13.0	0.0	0.0	0.014	0.10
12	191.0	12.0	0.0	0.0	0.018	0.14
13	203.0	15.0	0.0	0.0	0.015	0.08
14	218.0	27.0	0.0	0.0	0.25	0.60

SOURCES: DER (1980a); Gilbert (1978/1979); Roy F. Weston, Inc. (1982 and 1983); Potera et al. (1982); USGS (1982/1983, 1984).

^aDistance of beginning downstream from Chillisquaue Lake.

^bPlant discharges at beginning of reach 2.

^cDistributed flow rates are: (1) 0.001 m³/s/m for reaches 6 through 8, (2) 0.0017 m³/s/m for reach 9, and (3) 0.0002 m³/s/m for reaches 10 through 14. Minimum distributed flow rates are: (1) 0.00004 m³/s/m for reaches 6 through 8, (2) 0.000068 m³/s/m for reach 9, and (3) 0.000008 m³/s/m for reaches 10 through 14.

much the pollutant concentrations in the river would be affected by a substantial error in the original estimate. The worst case, of course, would be to treat the pollutant as conservative by using a very low removal rate. The nominal values of the removal coefficient and other parameters used in the analysis are given in Table 3.11. As discussed above, the background pollutant concentrations in the table represent our best estimates of the likely range of values in the Susquehanna, based on the relatively limited data in Stine (1985) and Skinner (1986).

The transport analysis assumed steady-state conditions and used annual mean discharge rates, pollutant concentrations, and flow rates. The most significant attenuation mechanism in rivers is dilution. Therefore, pollutant concentrations will be affected more by the river flow rate than by most other parameters. Accordingly, although the analysis used the mean flow rates of the Chillisquaque and Susquehanna as the nominal values, in the sensitivity analysis we also considered the low flow rates shown in Table 3.10 among the parameter variations.

The first reach of the Chillisquaque is not affected by the plant discharge. All discharges enter the system at the beginning of reach 2. These discharges, and the additional contributions from tributaries determine the pollutant concentrations in subsequent reaches.

In the initial analysis of pollutant concentrations we defined a number of alternative cases to be considered. The variables that changed between cases included the (1) pollutant concentration in plant discharges, (2) discharge sources within the plant, (3) pollutant removal rate, and (4) river flow rates. Nominal and high values of each variable were determined. The nominal values of each variable were used in the base case. Alternative cases were then defined using combinations of the nominal and high values for the variables.

Table 3.11

PARAMETERS FOR SURFACE WATER ANALYSIS

Parameter/Variable	Value
Removal coefficient (s^{-1})	
Nominal	8.02×10^{-7}
No removal	1.00×10^{-15}
Slow	8.02×10^{-9}
Rapid	8.02×10^{-4}
Background selenium concentration ($\mu g/l$)	2.0/10.00
Background beryllium concentration ($\mu g/l$)	0.10/0.40

The nominal selenium plant discharge concentration was chosen to be 0.010 mg/l, which corresponds to both the current EPA water quality criterion for selenium and the Montour permit level. The high emission rate was set equal to 0.017 mg/l, the maximum value measured by PP&L and the emission level reported in the permit application. Only emissions from the detention basin were considered for selenium. At the time of the study, the storm water basin discharge rate was not large enough to make it a significant source.

The nominal beryllium concentrations in the plant discharge were chosen to be 0.20 $\mu\text{g/l}$ (200 ng/l) for the detention basin and 0.30 $\mu\text{g/l}$ (300 ng/l) for the storm water basin. These levels correspond to the median concentrations measured during the TRE program. By comparison, the current EPA water quality criterion for beryllium is 68 ng/l and the normal detection limit is 200 ng/l (0.20 $\mu\text{g/l}$). Beryllium is not normally measured in the Chillisquaque, so we assumed a background concentration in the Susquehanna and Chillisquaque of 100 ng/l, or half the normal detection limit. The high discharge rates in the analysis were set at 0.30 $\mu\text{g/l}$ for the detention basin and 0.71 $\mu\text{g/l}$ for the storm water basin, the mean concentrations calculated for the TRE program.⁶

The high emission rate for selenium was selected for several reasons. First, in general it is better to use measured values instead of calculated ones. The mass balance analysis (Table 3.3) indicates that selenium concentrations in the detention basin overflow could be higher than 0.017 mg/l only if the background concentration in the Susquehanna is near 0.010 mg/l (the maximum value recorded in 1985 was 0.003 mg/l) and neither the ash basin nor the detention basin removes selenium from plant wastes. Second, ten months of recent measurements at PP&L support this choice. The results of this sampling program show that selenium concentrations in the Montour discharge range between 0.005 mg/l and 0.016 mg/l with a mean value of 0.010 mg/l (Stine, 1985). Third, using the selected nominal and high emission rates allows the analysis to compare directly the health effects of the Montour permit application discharge level (0.017 mg/l) and the permit requirement (0.010 mg/l).⁷

The discharge sources for beryllium included the detention basin (DB) and the storm water basin (SWB). The storm water basin

⁶The mean values were calculated assuming that all measurements below the detection limit represented concentrations at one-half this limit.

⁷The results of our analysis, presented later in this section, indicate that the Montour selenium discharges cause a negligible increase in total selenium intake for the exposed population. This outcome would not have been changed by assuming the somewhat higher discharge rate shown for the worst-case situation in Table 3.3.

discharge was set to the 1985 mean flow rate of 0.0132 m³/s (Skinner, 1986). The pollutant removal rate was varied between very rapid removal and no removal, a conservative case. Finally, river flows were set either at mean values or historical minimum rates. The primary cases incorporated into the analysis are defined in Table 3.12. The complete set of cases used in the two analyses can be found in Bolten and Resetar (1986) and Bolten et al. (1986).

Table 3.13 presents the mean concentrations for the surface water analysis. Notice that the units for selenium differ from the beryllium units. The concentrations shown in the table represent average values calculated over the length of each reach. Because the surface water transport model calculates incremental concentrations, the results in Table 3.13 can be quickly converted if other background concentrations are desired. In all selenium cases the background concentration was 0.002 mg/l for each reach. If one wanted the results for a background selenium concentration of 0.010, 0.008 needs to be added to each value.

Table 3.12

TRANSPORT ANALYSIS CASES

Case Number	Emission Rate ^a	Emission Source	Removal Rate ^b	Flow Rate ^b
Selenium Cases				
S1	Nominal	DB	Nominal	Mean
S4	Nominal	DB	None	Mean
S5	Nominal	DB	None	Minimum
S6	High	DB	None	Minimum
S7	High	DB	None	Mean
Beryllium Cases				
B2	Nominal	DB+SWBH	Nominal	Mean
B2C	Nominal	DB+SWBH	Nominal	Minimum
B5	High	DB+SWBH	Nominal	Mean
B5C	High	DB+SWBH	Nominal	Minimum
B7	Nominal	DB+SWBH	None	Mean
B7A	Nominal	DB+SWBH	None	Minimum
B7B	High	DB+SWBH	None	Mean
B7C	High	DB+SWBH	None	Minimum

^aSelenium concentrations in plant discharge: nominal—detention basin, 0.01 mg/l; high—detention basin, 0.017 mg/l. Beryllium concentrations in plant discharge: Nominal—detention basin, 0.2 µg/l, storm water basin, 0.3 µg/l; high—detention basin, 0.3 µg/l, storm water basin, 0.6 µg/l.

^bRemoval and flow rates are shown in Tables 3.8 and 3.7.

A similar situation occurs for beryllium. The background concentration level used for Table 3.13 is 100 ng/l in each reach. This corresponds to the lower measured value discussed above. To get the concentration for the higher background concentration of 400 ng/l, add 300 to each value in the table.

From the table, background selenium concentrations in the Susquehanna are affected only slightly by Montour discharges. Under worst-case assumptions (high emissions/no removal/minimum flows—case S6), selenium concentrations in all but reach 4 (the West Branch of the Susquehanna from the mouth of the Chillisquaque to Sunbury) increase by 10 percent or less over the background level. In other, less stressing cases, the increase is generally less than 1 percent of the background level. The recommended EPA water quality criterion for selenium is currently 10 $\mu\text{g/l}$. In no case does the plant ever contribute more than a negligible fraction of this level. Even in the Chillisquaque, selenium concentrations remain below the criterion level, except for cases having both high emission rates and minimum flows.

Similarly, background beryllium concentrations in the Susquehanna are affected only slightly by the Montour discharges. Under the

Table 3.13

MEAN CONCENTRATIONS IN SURFACE WATER ANALYSIS

Case	Reach													
	1	2	3	4	5	6	7	8	9	10	11	12	13	14
Selenium ($\mu\text{g/l}$)														
S1	2.00	5.96	4.42	2.01	2.00	2.00	2.00	2.00	2.00	2.00	2.00	2.00	2.00	2.00
S4	2.00	5.97	4.58	2.01	2.00	2.00	2.00	2.00	2.00	2.00	2.00	2.00	2.00	2.00
S5	2.00	9.56	9.09	2.28	2.11	2.11	2.11	2.09	2.08	2.08	2.08	2.08	2.08	2.08
S6	2.00	16.2	15.3	2.52	2.21	2.21	2.20	2.16	2.15	2.15	2.15	2.15	2.15	2.14
S7	2.00	9.46	6.84	2.02	2.01	2.01	2.01	2.01	2.01	2.01	2.01	2.01	2.01	2.01
Beryllium (ng/l)														
B2	100	152	132	100	100	100	100	100	100	100	100	100	100	100
B2C	100	197	185	103	101	101	101	101	101	100	100	100	100	100
B5	100	206	165	100	100	100	100	100	100	100	100	100	100	100
B5C	100	298	272	106	102	102	102	101	101	101	101	100	100	100
B7	100	152	134	100	100	100	100	100	100	100	100	100	100	100
B7A	100	198	192	104	102	101	101	101	101	101	101	101	101	101
B7B	100	206	169	100	100	100	100	100	100	100	100	100	100	100
B7C	100	299	286	107	103	103	103	102	102	102	102	102	102	102

NOTE: Results are based on background concentrations of 2 $\mu\text{g/l}$ for selenium and 100 ng/l for beryllium.

worst-case conditions (high emissions/minimum flows/no removal—case B7C), beryllium concentrations in all reaches increase by 7 percent or less over the background level of $0.1 \mu\text{g/l}$. In other, less stressing cases, the increase is usually less than 1 percent of the background level. The recommended EPA water quality criterion for beryllium is currently $0.068 \mu\text{g/l}$ (68 ng/l). Although the calculated beryllium concentration is above this level in all cases, the natural background levels in the river (and probably the creek) are also above the criterion. Consequently, it is more important to consider the incremental effects of plant discharges, which are negligible for the Susquehanna.

For the Chillisquaque, the total beryllium concentrations remain below $0.30 \mu\text{g/l}$ in all cases. For those cases having long-term mean flow rates, even with high discharge rates and no removal, the maximum concentrations only slightly exceed $0.20 \mu\text{g/l}$. Under normal discharge rates, these maximum concentrations in the Chillisquaque do not significantly exceed $0.15 \mu\text{g/l}$, even assuming a conservative pollutant with no removal.

When calculating chronic health effects, the analysis should consider the results for cases using mean, long-term river flow rates. Although the minimum flow conditions may be important for acute or short-term toxicity problems, these flows do not occur frequently or last very long. The water quality criteria are calculated on the basis of chronic health effects. Because of their limited occurrence and duration, the minimum flow conditions should not significantly affect the incidence of these chronic health conditions. For completeness and reference, however, some minimum flow cases will be considered in the exposure analysis.

The results for the mean flow cases show that both selenium and beryllium concentrations in the Susquehanna River increase by less than 1 percent, even with high emission rates and no pollutant removal. These concentrations are low because the flow rates on the Susquehanna are normally high, and plant discharges are quickly diluted when they leave the Chillisquaque.

Exposure Calculations

General Information. In the discussion of the regional description and population specification, we explained that the exposure calculations for this preliminary study included four pathways for surface water pollutants: (1) ingestion of drinking water, (2) consumption of aquatic organisms (3) consumption of vegetation produced on land irrigated with water from the Susquehanna or Chillisquaque, and (4) consumption of animal products from animals that either drink

contaminated water or consume affected vegetable products. The populations exposed through these pathways have been described above. In this subsection we discuss the additional information required by the exposure analysis.

The amount of pollutant consumed by an individual through drinking water will depend on the subregion in which the individual lives. Each subregion collects its drinking water from a different section of the Susquehanna and thus uses water that may have different pollutant concentrations.

For the consumption of fish, however, the situation is different. Here we assume that the total catch is distributed uniformly throughout the entire exposed population. Thus each member of a population age group will consume the same amount of contaminated fish, although the amount consumed by the different population age groups will vary. If the supply is less than normal consumption, the difference is made up from sources outside the area. As a consequence of these assumptions, the exposure rate for ingestion of fish will be constant across subregions for each population group.

A similar procedure is used for consumption of animal and vegetable products. Pollutant concentrations in these products are calculated as a function of where they are grown. Vegetables and grains are irrigated from the closest reach of the Susquehanna or Chillisquaque. Animals consume water from the nearest reach and may eat grains that have been irrigated with local water. After determining the pollutant concentrations in each product by subregion, we assume that all local production is mixed and distributed equally to the exposed population. Thus, for each product, we calculate total production levels and a mean pollutant concentration. The exposed population consumes this local production first and, as with fish, supplements this where necessary with products from outside the region. These external products will have background concentrations of both beryllium and selenium.

This procedure means that the entire exposed population will consume fish, animal, and vegetable products with the mean pollutant concentrations calculated for each product. Exposure rates will vary across subregions, therefore, only because of the variation in drinking water characteristics. These assumptions should not dramatically affect exposure rates or the calculated health risks. For that part of the population consuming drinking water but not actually eating fish or contaminated animal and vegetable products, total exposure will be overestimated. On the other hand, there will be a corresponding underestimation of exposure rates for the population whose consumption rates for affected products are higher than the mean values. We

are still being conservative in the analysis, however, because of the assumption that all locally grown products are consumed locally. In our results, as described below, we found the incremental exposure rates to be an extremely small fraction of the total exposure rate. Under these circumstances changing the distribution of the increment should have virtually no effect on the results or conclusions.

The current calculations assume that plant operation levels, emission rates, and regional population are constant over a plant life of 35 years. If emissions are not constant, one can calculate the environmental concentrations associated with varying emission levels and average these over the life of the plant. Similarly, one can use population projections to estimate an average population over the plant lifetime. Although we have investigated the possibility of incorporating a variable population model in this study, such a complication is generally not warranted for this level of analysis.

Consumption of Aquatic Organisms. The production of aquatic organisms for this study is the total catch of fishermen using the Susquehanna. Very little is known about fishing activities in Pennsylvania and, in particular, on the Susquehanna. In the absence of useful data, this information must be estimated from limited statistics and surveys using a number of assumptions and simplifications. Pennsylvania has approximately 1,900,000 licensed and unlicensed fishermen (FWS, 1982). Of these, we assumed that 14 percent spent half their time fishing the Susquehanna River (Johnson, 1984). The average fishing trip lasts four hours with a mean harvest of 0.34 fish per hour (Nardacci, 1984). Each fisherman takes 22 trips per year (FWS, 1982), yielding a total of 3.97 million fish harvested each year from the Susquehanna.⁸ This is undoubtedly a very high estimate which should produce an upper bound on the exposure to beryllium and selenium through consumption of fish.

Bioconcentration factors for selenium are known to differ by fish species. Accordingly, for the selenium analysis, we divided the total fish harvest into five general types: (1) sunfishes—smallmouth bass, largemouth bass, rock bass, pumpkinseed, and bluegill; (2) pike such as the muskellunge; (3) carp; (4) freshwater catfishes; and (5) other (Bolten and Resetar, 1986). The first four groups constitute approximately 76 percent of the fish harvested (NRC, 1981) or 94 percent of the total weight. The fifth group includes species harvested in relatively small

⁸Another study (Denoncourt, 1984) of limited areas along the Susquehanna found that fishermen spent between 2.3 and 2.8 hours per trip, catching between 0.63 and 0.73 fish per hour. These numbers give between 1.45 and 2.07 fish per trip, somewhat higher than our estimate of 1.36. However, the Denoncourt study investigated areas (near power plants) that were believed to have good fishing conditions.

proportions, which are grouped together and given common mean characteristics to simplify the calculations. Because bioconcentration factors for beryllium are not known by species, total fish production was used and given mean parameter values. Therefore, for beryllium, total fish production was used and given mean parameter values. Each species catch was multiplied by assumed mean weights taken from the literature to get the species production. Total fish production was calculated by summing each species production rate.

In addition to the production rates, bioconcentration factors had to be determined for each species. These factors, which depend to some extent on the pollutant background concentration level in the surface water, were found in the literature. Specific selenium bioconcentration factors were available for the sunfish and carp (SRBC, 1979). Selenium factors for other species generally range between 8 and 78 (NAS, 1976). Because of their behavior and other characteristics, catfish were assigned a bioconcentration factor near the higher end of the range, and pike were assumed to be more similar to the sunfish at the lower end of the range. The other fish were given a mean bioconcentration factor based on literature values.⁹ For all species, we assumed that the background selenium concentration in fish from outside the region was 0.5 mg/kg (NAS, 1976).

Bioconcentration factors for beryllium are not known by species. Accordingly, we assumed a value of 19 for all five fish types described above (EPA, 1980a). In the sensitivity analysis, we also calculated exposure rates and health effects assuming a bioconcentration factor of 100, although our previous results for selenium had found that exposure rates were relatively insensitive to this parameter. In addition, the analysis assumed that the background beryllium concentration in fish from outside the region was 0.56 $\mu\text{g/kg}$ (EPA, 1980a). Table 3.14 summarizes the input information for the aquatic organisms used in the analysis.

Consumption rates for fish were estimated from national averages by population group (NRC, 1977).¹⁰ We assumed that (1) each group would divide its fish consumption equally between fresh and saltwater species, and (2) amounts consumed over and above those harvested from the Susquehanna have the mean pollutant concentrations of fish from outside the region. Table 3.15 presents consumption rates by population group for fresh fish and all aquatic organisms.

⁹The results were very insensitive to this parameter. When exposure rates were calculated with bioconcentration factors equal to twice the nominal values, total exposure rates changed by less than 0.5 percent. Incremental exposure caused by plant operations increased negligibly.

¹⁰These 1977 consumption rates may underestimate current and future fish consumption somewhat because of the recent trend away from red meat consumption.

Table 3.14

CHARACTERISTICS OF AQUATIC ORGANISMS

Species	Harvest Rate (kg/day)	Bioconcentration Factor	
		Be	Se
Sunfish	19,000	19	30
Carp	4,400	19	80
Pike	2,200	19	25
Catfish	14,800	19	50
Other	2,400	19	18

Table 3.15

CONSUMPTION RATES FOR AQUATIC ORGANISMS

(kg/day)

Item	Population Group			
	Infant	Child	Teenager	Adult
Total	0.0	0.006027	0.014247	0.018904
Fresh fish	0.0	0.003014	0.007124	0.009452
Sunfish	0.0	0.001338	0.003163	0.004197
Carp	0.0	0.000313	0.000741	0.000983
Pike	0.0	0.000157	0.000370	0.000492
Catfish	0.0	0.001040	0.002458	0.003261
Other	0.0	0.000166	0.000392	0.000520

Drinking Water Consumption. Pollutant concentrations in the Susquehanna River will be reduced by drinking water treatment processes used in the various communities. Table 3.7 describes the type of treatment used by most of the water companies. Although some companies did not specify the type of treatment processes used, they would have to treat the water from the Susquehanna. We thus assumed that they would apply conventional coagulation and filtration processes to their Susquehanna supplies.

Using information from Sorg and Logsdon (1978) we estimated that the general processes listed in Table 3.7 would remove about 20 percent of the selenium from Susquehanna River water. We do not assume a higher efficiency because the removal process is sensitive to relatively small changes in some of the assumptions. In the sensitivity analysis we examine this assumption and consider the case where no removal

takes place. Unfortunately, very little is known about the effectiveness of such processes for removing beryllium. As a consequence, we made the conservative assumption that no beryllium would be removed by drinking water treatment.

Consumption of Animal and Vegetable Products. This analysis also considered two additional types of exposure, consumption of animal and vegetable products produced in the region. The analysis of these two pathways was based on the approach described in Bolten et al. (1983). Using USDOC (1984a and 1984b) with area maps, we determined the primary agricultural goods produced in the counties bordering the Susquehanna River and Chillisquaque Creek downstream from the Montour SES. These goods include milk, beef (cattle and calves), pork, poultry, vegetables, and grains (several types). For these aggregated categories, we determined the (1) production rates by county, (2) irrigation rates, (3) the quantities of each crop irrigated, and (4) the values of other parameters necessary for the calculations. Assuming that production of each crop was uniformly distributed within counties, we partitioned total production between river reaches on the basis of geographical considerations. Table 3.16 contains the production rates by river reach. Table 3.17 presents the values of other parameters used in the agricultural analysis, including background pollutant concentrations in products grown outside the region.

Table 3.16

AGRICULTURAL PRODUCTION BY REACH

Reach	Vegetables (kg/day)	Grain (kg/day)	Milk (l/day)	Pork (kg/day)	Poultry (kg/day)	Beef (kg/day)
1	0	0	0	0	0	0
2	7,930	83,926	37,576	3,558	3,250	2,027
3	6,294	83,172	20,206	2,557	6,456	1,482
4	6,294	83,172	20,206	2,557	6,456	1,482
5	11,091	167,659	83,001	6,275	30,625	3,570
6	11,091	167,659	83,001	6,275	30,625	3,570
7	5,738	313,481	194,147	4,980	49,708	9,120
8	11,215	488,898	385,213	8,500	21,841	14,525
9	19,233	409,239	349,293	31,093	95,947	24,110
10	19,233	409,239	349,293	31,093	95,947	24,110
11	19,233	409,239	349,293	31,093	95,947	24,110
12	19,233	409,239	349,293	31,093	95,947	24,110
13	19,233	409,239	349,293	31,093	95,947	24,110
14	45,890	621,332	260,855	2,049	15	10,371
Total	201,709	4,055,491	2,830,666	192,214	628,709	166,695

The surface water transport analysis calculated selenium and beryllium concentrations by reach for each case of the analysis. These concentrations were used in the irrigation water for crops and drinking water for animals produced along each reach. Moreover, in some cases animals were assumed to consume appropriate irrigated crops as feed.

Using information from Bolten et al. (1983), we estimated mean consumption rates by population group for each animal and vegetable product. These rates are presented in Table 3.18. Miscellaneous data required for the exposure analysis are shown in Table 3.19.

These tables present only single values of the parameters shown. Previous studies have shown that the exposure rates would be relatively insensitive in this case to almost all of these parameters, over the appropriate range of values. Modifying one or more parameters would cause a corresponding change in the overall exposure rates but would leave the relative incremental exposure from plant operations unchanged. The most sensitive parameter in Table 3.19 is the background pollutant concentration in outside food. The values shown are within the range expected for a reasonably balanced diet and give total pollutant ingestion rates that are consistent with the literature (EPA, 1980a and 1980d).

Analysis Cases. The exposure analysis was based on the results of the surface water transport calculations discussed above. The pollutant concentrations shown in Table 3.13 were used with the subregion population data presented in Table 3.9 and the input parameter values shown in Tables 3.14 through 3.19. In the initial analysis, we defined a number of alternative cases to be considered for each pollutant. These cases were selected to cover a wide range of conditions and to demonstrate the potential range of population exposure rates that could be encountered. Variables that changed between these cases included the (1) pollutant emission rate or concentration in plant discharges, (2) emission sources, (3) pollutant removal rate, and (4) river flow rates. The cases incorporated into the analysis for the two pollutants are defined in Table 3.20.

Several of these cases are particularly significant. The nominal cases (SE1 and BE2) use the nominal emission rate (with or without the storm water basin discharge) with long-term mean flow rates and nominal selenium and beryllium removal from the water. These cases show how the Montour discharges affect exposure rates in the population under expected conditions. Cases SE4 and BE7 are similar, except that they assume conservative pollutant transport with no removal from the water. Finally, cases SE7 and BE7B incorporate the high emission rate with mean flow rates and no pollutant removal. The results of these cases should show how controlling plant emissions will

Table 3.17

MISCELLANEOUS AGRICULTURAL ANALYSIS PARAMETERS

Parameter	Selenium	Beryllium
Animal feed consumption rates		
Beef (grass)	50 kg/day	
Pork (grains)	4.2 kg/day	
Milk (grass)	50 kg/day	
Poultry (grains)	0.15 kg/day	
Animal water consumption rates		
Beef	50 l/day	
Pork	10 l/day	
Milk	60 l/day	
Poultry	0.4 l/day	
Irrigation rate	1.37 l/m ² /day	
Fraction of crop irrigated		
Vegetables	0.120	
Grains	0.05	
Bioconcentration factor		
Grains	1.0	0.01
Vegetables	0.025	0.01
Translocation factor for pollutant movement to edible portions		
Grains	0.1	
Vegetables	0.3	
Vegetative surface density		
Grains	0.59 kg/m ²	
Vegetables	1.15 kg/m ²	
Vegetation parameters		
Weathering loss rate	0.05 /day	
Mean capture fraction	0.25	
Surface soil density	240 kg/m ²	
Pollutant transfer loss rate through soil	0.00001 /day	
Background pollutant concentration		
Beef and pork	0.2 mg/kg	0.2 µg/kg
Milk	0.02 mg/kg	0.2 µg/kg
Poultry	0.1 mg/kg	0.2 µg/kg
Vegetables and grains	0.3 mg/kg	0.2 µg/kg
Fraction of daily selenium intake in animal product		
Beef	0.015	0.001
Pork	0.20	0.01
Milk	0.004	0.0000009
Poultry	4.0	0.05

SOURCES: Baes et al. (1984), Bolten et al. (1983 and 1985), EPA (1980a and 1980d), FASEB (1971), Lane (1983), Meehan and Smythe (1967), Morris and Levander (1970), NRC (1977), and Reeves (1979).

Table 3.18

CONSUMPTION RATES FOR EXPOSURE ANALYSIS

Product	Population Group			
	Infant	Child	Teenager	Adult
Total food (kg/day)	0.55	1.19	1.50	1.28
Drinking water (l/day)	0.55	0.71	0.71	1.01
Animal products				
Beef (kg/day)	0.00	0.03	0.04	0.07
Pork (kg/day)	0.00	0.03	0.04	0.07
Milk (l/day)	0.55	0.47	0.55	0.30
Poultry (kg/day)	0.00	0.01	0.02	0.03
Vegetable products				
Grains (kg/day)	0.00	0.12	0.15	0.12
Vegetables (kg/day)	0.00	0.18	0.21	0.19

Table 3.19

MISCELLANEOUS PARAMETERS FOR THE EXPOSURE ANALYSIS

Variable	Assumed Value
Specific ingestion rate coefficient	0.1154
Ingestion exposure absorption fraction	1.00
Background concentration in outside food	
Selenium	0.10 mg/kg
Beryllium	0.2 µg/kg

affect exposure rates and health risks even when no pollutant is removed from the water through natural processes.

The majority of the regional population affected by plant discharges are exposed to the pollutants through direct or indirect consumption of water from the Susquehanna River. No one consumes drinking water from the Chillisquaque Creek, and only a small fraction of the population eats animal or vegetable products raised on Chillisquaque water. As a result, because beryllium and selenium concentrations in the Susquehanna are low in the transport analysis cases, the net chronic health risks should also be small. Although the pollutant concentrations in the Chillisquaque can be affected by plant discharges, there are no drinking water withdrawals from that creek and virtually no fishing.

Table 3.21 presents the results of the exposure analysis cases defined above. These results are shown as overall mean background and total

Table 3.20

CASES FOR BERYLLIUM AND SELENIUM EXPOSURE ANALYSIS

Case Number	Emission Rate	Emission Source	Removal Rate	Flow Rate
Selenium				
SE1	Nominal	DB	Nominal	Mean
SE4	Nominal	DB	None	Mean
SE5	Nominal	DB	None	Minimum
SE6	High	DB	None	Minimum
SE7	High	DB	None	Mean
Beryllium				
BE2	Nominal	DB+SWBH	Nominal	Mean
BE2C	Nominal	DB+SWBH	Nominal	Minimum
BE5	High	DB+SWBH	Nominal	Mean
BE5C	High	DB+SWBH	Nominal	Minimum
BE7	Nominal	DB+SWBH	None	Mean
BE7A	Nominal	DB+SWBH	None	Minimum
BE7B	High	DB+SWBH	None	Mean
BE7C	High	DB+SWBH	None	Minimum

NOTE: DB—detention basin; DB+SWBH—detention basin plus storm water basin at high discharge level.

exposure rates (for the affected population) in nanograms of beryllium (ng/day) and micrograms of selenium ($\mu\text{g/day}$) ingested per day. As will be discussed below, one can calculate health risks for selenium and beryllium exposure from these rates using information developed by the EPA (1980a and 1980d).

The results in Table 3.21 are presented to six significant digits. This has been done to show the size of the predicted variation in exposure rates, not because these results are significant to that level of precision. Actual exposure rates for the region are highly uncertain for both pollutants and may differ from the background levels shown in the table by a factor of two or more. For selenium, the EPA has calculated average consumption rates near these levels (EPA, 1980a), but beryllium consumption rates are less well known.

For both pollutants, however, the variations between cases and between background levels and plant-related exposure rates are significant. Because all of these values have been calculated with the same set of inputs and parameters, the differences between exposure rates are far less uncertain than the rates themselves. These differences should reflect the real variations between cases and background levels.

Table 3.21

EXPOSURE RATES FOR ALTERNATIVE CASES

Case Number	Mean Population Exposure Rates	
	Background	With Plant
Selenium ($\mu\text{g/day}$)		
SE1	184.055	184.090
SE4	184.055	184.092
SE5	184.055	184.251
SE6	184.055	184.423
SE7	184.055	184.125
Beryllium (ng/day)		
BE2	1238.13	1238.27
BE2C	1238.13	1238.90
BE5	1238.13	1238.41
BE5C	1238.13	1239.70
BE7	1238.13	1238.28
BE7A	1238.13	1239.08
BE7B	1238.13	1238.42
BE7C	1238.13	1240.06

For selenium, the incremental exposure associated with the nominal case (SE1) is only 0.02 percent of the background level. Similarly, for the two cases based on long-term mean flow rates in the affected waterways (SE4 and SE7), the exposure increments are only 0.02 and 0.04 percent. These increases correspond to annual intakes of between 1.3 and 2.5×10^{-5} grams. Even for the minimum flow cases, the incremental exposure associated with plant emissions is at most 0.2 percent of background levels, corresponding to an additional annual intake of less than 1.4×10^{-4} grams of selenium. As discussed below and in Appendix B, these compare to an EPA selenosis threshold intake rate of 7.0×10^{-3} grams per day. Including background, the worst case in Table 3.21 produces a daily intake of only 1.844×10^{-4} grams, a factor of 37 below the threshold.

Beryllium exposure rates are also not significantly affected by plant emissions, as shown in the table. The exposure increase associated with Montour discharges in both the nominal case (BE2) and the nominal case with no pollutant removal in the surface waters (BE7) is only 0.01 percent of background exposure. Under historical low-flow conditions, which clearly do not persist, exposure rates increase only by 0.08 percent (BE7A). Note that this case also assumes no beryllium decay or removal either in the surface waters or in drinking water treatment

processes. On an annual basis, these exposure rates correspond to intakes of 5.1×10^{-8} grams (BE2) and 3.46×10^{-7} grams (BE7A).

For both pollutants, the mean flow cases should be given primary attention. They represent the long-term exposure rates that the affected population would expect to receive over the life of the Montour plant. Although the minimum flow cases cause higher exposure rates, these minimum flows are historical lows which will occur only rarely and will not persist. Consequently, the exposure rates associated with such flows will not significantly affect total exposure or dose over the lifetime of the power plant.

Health Effects for the Exposure Analysis

In this section we use the results of the discussion in Appendix B to calculate the chronic health risks associated with the exposure cases defined earlier. Because the health data and health effects of selenium and beryllium differ, we must use a different approach for each pollutant.

Because selenium has not been shown to be carcinogenic, the EPA has chosen to use a selenosis threshold approach to deriving its water quality criterion (EPA, 1980a). This approach does not provide very much information about health effects, however, because the calculated total ingestion rates shown in Table 3.21 are more than a factor of 35 lower than the EPA threshold level of 7.0 mg/day. Accordingly, we have decided to use an additional method of estimating health effects.

Although the average selenium intake level of the population may be low, compared to the selenosis threshold, certain subgroups of people, particularly those who eat large quantities of vegetables, may be at a somewhat higher risk than the average person. Some of those may be sufficiently near the toxic limit to conceivably be pushed over the limit by additional exposure to power plant selenium. The selenium dietary distribution can be crudely fit to a log normal distribution. As described in Appendix B, this distribution is characterized by a standard deviation of $\sigma_{\log I} = 0.3238$. Accordingly, the probability, P , that an individual will have a dietary intake exceeding the toxic threshold, $I_t = 7.0$ mg/day, is given by

$$P = 1 - \frac{1}{\sqrt{2\pi}} \int_{-\infty}^{(\log I_t - \langle \log I \rangle) / \sigma} e^{-z^2/2} dz \quad (3.2)$$

where $z = (\log I - \langle \log I \rangle) / \sigma_{\log I}$,

I = selenium ingestion rate ($\mu\text{g/day}$),

I_t = toxicity threshold ($\mu\text{g/day}$),

σ = standard deviation of log normal distribution.

For each case WTRISK calculates the mean background exposure rate and the total exposure rate (including plant emissions) by population group and subregion. From these results, weighted mean exposure rates are calculated for the total exposed population, using the subregion populations as the weights. These mean exposure rates (presented in Table 3.21) can be used to estimate the probability that members of the exposed population will exceed the selenosis threshold.

The probability function describes the distribution of selenium exposure among the general population. If this distribution is log normal (as assumed), a certain fraction of the population at the upper end of the distribution may exceed the selenosis threshold for a given mean exposure rate. If this mean exposure rate increases above the background level, because of plant selenium discharges, the probability density curve shifts to slightly higher values. As a consequence, a few more individuals may exceed the toxic threshold. The increased probability of an individual exceeding this threshold is the difference between the P calculated with a $\langle \log I \rangle = \log (184.090 \text{ } \mu\text{g/day})$ and the P with a $\langle \log I \rangle = \log (184.055 \text{ } \mu\text{g/day})$, for example. These mean probabilities can be combined with the population information to calculate an expected value for the number of people exceeding the selenosis threshold for each case.

As discussed in Appendix B, beryllium health risks can be calculated more conventionally. Although the data are limited and controversial, the EPA used them to derive a water quality criterion based on a one-hit dose-response function for carcinogenesis. We can use this one-hit model to calculate the chronic health risk associated with the exposure rates presented in Table 3.21. This can be done automatically in the WTRISK program. The results, calculated for each analysis case, include the expected incidence of cancer in the exposed population for both the normal background exposure rates and the exposure associated with plant emissions.

Using these approaches, we calculated the chronic health risks associated with each of the cases presented in Tables 3.20 and 3.21. These results—the incidence of adverse health effects in the exposed population (about 1.5 million) and the individual incremental risk for a member of the population—are shown separately for selenium and beryllium in Table 3.22. As before, the cases are separated by river flow, to highlight the differences between the expected long-term, average conditions and the extreme situations.

Again, the number of digits in these results was chosen to show how the various cases differ from the background situation. They do not imply that the results are accurate or significant to that level, only that the relative exposures rates and health risks increase by extremely

Table 3.22
TOTAL AND INDIVIDUAL INCREMENTAL HEALTH RISKS

Case	Mean Exposure Rate ^a	Incidence of Disease ^b		Incremental Individual Risk
		Total	Incremental	
Selenium				
Background	184.055	0.8133	—	—
Mean flows				
E001	184.090	0.8143	0.0010	6.8×10^{-10}
E004	184.092	0.8144	0.0011	7.4×10^{-10}
E007	184.125	0.8154	0.0021	1.4×10^{-9}
Minimum flows				
E005	184.251	0.8191	0.0058	3.9×10^{-9}
E006	184.423	0.8242	0.0109	7.4×10^{-9}
Beryllium				
Background	1238.13	111.88	—	—
Mean flows				
E002	1238.27	111.89	0.0121	8.2×10^{-9}
E005	1238.41	111.90	0.0248	1.7×10^{-8}
E007	1238.28	111.89	0.0127	8.5×10^{-9}
E007B	1238.42	111.90	0.0259	1.8×10^{-8}
Minimum flows				
E002C	1238.90	111.95	0.0686	4.6×10^{-8}
E005C	1239.70	112.02	0.140	9.5×10^{-8}
E007A	1239.08	111.96	0.0847	5.7×10^{-8}
E007C	1240.06	112.05	0.172	1.1×10^{-7}

^aExposure rate units: selenium ($\mu\text{g}/\text{day}$), beryllium (ng/day).

^bDisease of concern: selenium—selenosis, beryllium—cancer.

small amounts. If all results were presented with the precision that could actually be measured or identified, none of the cases would appear to be different from the background levels.

The selenium results in Table 3.22 show that under varying conditions with mean river flows (cases SE1, SE4, and SE7), the lifetime risk of selenosis associated with plant emissions is less than 1.5×10^{-9} , or 1.5 in a billion. This compares to the often used EPA risk criterion levels of one in a million or one in one hundred thousand (EPA, 1980c). Even with historical low flows, the risk associated with plant emissions remains below 7.5×10^{-9} , or 7.5 in a billion. In fact, the difference between the nominal surface water transport case (SE1) and the worst case (SE6) is about a factor of ten in individual risk.

The beryllium health risks are also relatively small and change little between the various surface water cases of the analysis. In the mean flow rate cases, the maximum health risk occurs when we assume high

emission rates and no pollutant removal during transport (case BE7B). This incremental individual risk is 1.8×10^{-8} , or 1.8 in one hundred million. With historical minimum flows, the maximum risk is 1.1×10^{-7} , or 1.1 in ten million, also assuming high emission rates and no pollutant removal (case BE7C). Again the difference between the nominal surface water transport case (BE2) and the worst case (BE7C) is about a factor of 13 in incremental individual risk.

The exposure and health risks calculated for the alternative cases do not differ dramatically from the nominal cases. This lack of variation is caused by the flow of the Susquehanna River, which is large enough to dilute the pollutant emissions from the Montour plant to below measurable levels. As a result, exposure rates for the downstream population do not increase significantly and the chronic health risk is correspondingly low. Although the pollutant concentrations in the Chillisquaque Creek are increased somewhat by plant discharges¹¹ these increases do not significantly affect a large part of the regional population.

SENSITIVITY ANALYSIS

We simplified the sensitivity analysis in this risk assessment by separating the transport analysis from the exposure calculations. As discussed above, we used alternative cases to investigate the sensitivity of transport results to variations in emission rates, river flows, discharge sources, and pollutant removal coefficients. This work is described in more detail in Bolten and Resetar (1986) and Bolten et al. (1986). The exposure sensitivity analysis was performed by incorporating exposure parameter variations and assumptions into the analysis done with alternative surface water transport cases.

Sensitive Parameters and Assumptions

The sensitivity analysis was performed separately for each pollutant. Parameter variations were based on previous experience, literature ranges, and reasonableness. The specific parameter values used in the various sensitivity cases are shown in Table 3.23. In general, for each of the pollutants, we calculated exposure rates under various combinations of assumptions about parameter values and population behavior.

¹¹Measurements made during the TRE program (Skinner, 1986) indicate that beryllium concentrations in Chillisquaque Creek downstream of the Montour discharge do not increase significantly over the background concentrations measured above the plant in most cases.

In this subsection we discuss some of these parameters and assumptions.

The food consumption rates used in the analysis are national averages by age group. Regional data would be much more appropriate, but these are generally not available. To compensate for this lack of information and uncertainty about local preferences, we doubled normal consumption rates for all age groups and all products in the sensitivity analysis.

Bioconcentration factors for animals, plants, and aquatic organisms have been found to be important parameters in risk analysis. The aquatic bioconcentration factors used for beryllium and selenium were derived from limited studies of a few fish species (EPA, 1980a and 1980d). Because this factor may be both important and highly uncertain, we used WTRISK to calculate the health risks assuming alternative values. As with previous analysis of water and fish consumption only (Bolten et al., 1986), we found that fish consumption is not a primary dietary source of beryllium or selenium. Accordingly, exposure rates and health risks are not particularly sensitive to changes in bioconcentration factors alone. Less is known about bioaccumulation in vegetable species and the various types of animals. Our nominal

Table 3.23

PARAMETER VARIATIONS FOR SELENIUM SENSITIVITY ANALYSIS

Parameter	Nominal	Pessimistic
Consumption rates		
Total food	Table 3.18	2 × Nominal
Drinking water	Table 3.18	2 × Nominal
Fish	Table 3.15	2 × Nominal
Animal products	Table 3.18	2 × Nominal
Grains	Table 3.18	2 × Nominal
Vegetables	Table 3.18	2 × Nominal
Fraction of selenium remaining after water treatment	0.80	1.00
Bioconcentration factors		
Selenium	Table 3.14	2 × Nominal
Beryllium	19	100
Background pollutant concentration in products from outside region	Table 3.17	2 × Nominal
Fraction of daily selenium intake in animal product	Table 3.17	2 × Nominal
Irrigation rate	Table 3.17	2 × Nominal
Fraction of crop irrigated	Table 3.17	1.00
Animal feed	Table 3.17	All eat grains

values were taken from Baes et al. (1984), whose work has been predominantly in nuclear risk assessment. We varied these parameters by a factor of two in the sensitivity work. Although they could be increased by much larger factors, there is no evidence for such higher values, and overall pollutant exposure rates and health effects would be unreasonably high.

Background concentration levels in food and water affect selenium health risks far more than they affect beryllium health risks. This is a consequence of the methods used to calculate health risks. The adverse effect for selenium is selenosis, associated with exceeding a threshold consumption rate. When background levels increase, members of the population move closer to the threshold, making plant-related selenium contributions more significant. For beryllium, the adverse effect is cancer. Because the health risk is calculated using a linear model, changes in the overall background level of consumption will not affect the incremental risk associated with plant discharges.

The treatment effectiveness for drinking water processes may also be important in the selenium analysis. The literature does not specify the chemical form of selenium discharged from power plants. As a result, the removal efficiency of drinking water treatment is also uncertain. In the sensitivity analysis, we reduced the nominal 20 percent removal to zero, effectively making selenium a conservative pollutant from the power plant through water treatment.

We also varied assumptions about irrigation practices and animal feeds, to represent potential changes in these factors over time and to account for worst-case conditions. More specifically, we assumed in the analysis that all animals ate only locally produced grains, and that all grains and vegetable products were irrigated, and only with water from the Chillisquaque or Susquehanna.

The exposure rates for the sensitivity analysis cases for selenium are presented in Table 3.24. The worst-case scenarios use the pessimistic values of all parameters and pessimistic assumptions, except as noted. We chose to keep the assumed background concentration of selenium constant in one worst-case scenario because the health risk depends so strongly on overall consumption rates and because the limited data indicate that consumption rates are generally much lower than those calculated in the regular worst-case scenario.

From these results it is clear that, although absolute exposure rates change dramatically from case to case, the relative contributions of plant emissions change little. For example, in the base case with no pollutant decay or removal (BE4), incremental exposure in the nominal situation is 0.02 percent of background levels. Under worst case conditions in the sensitivity analysis, although the overall background

Table 3.24
SELENIUM EXPOSURE RATES FOR SENSITIVITY ANALYSIS CASES

Case	$(\mu\text{g/day})$				
	Mean Flow Rates			Minimum Flow Rates	
	BE1	BE4	BE7	BE5	BE6
Nominal case					
Plant	184.090	184.092	184.125	184.251	184.423
Background	184.055	184.055	184.055	184.055	184.055
Irrigate all crops					
Plant	185.008	185.012	185.112	185.297	185.647
Background	184.898	184.898	184.898	184.898	184.898
Animals eat crops					
Plant	219.060	219.062	219.095	219.222	219.396
Background	219.024	219.024	219.024	219.024	219.024
Animals eat irrigated crops					
Plant	220.243	220.247	220.356	220.554	220.931
Background	220.123	220.123	220.123	220.123	220.123
Double consumption rates					
Plant	407.336	407.339	407.398	407.646	407.974
Background	407.272	407.272	407.272	407.272	407.272
Worst case with constant background					
Plant	779.058	779.070	779.396	780.092	781.314
Background	778.698	778.698	778.698	778.698	778.698
Worst case					
Plant	934.089	934.102	934.429	935.123	936.345
Background	933.729	933.729	933.729	933.729	933.729

exposure rate increases by a factor of five, incremental exposure caused by Montour emissions is still only 0.04 percent of background. This type of behavior persists for all cases considered—the relative amount of selenium intake caused by Montour emissions varies only slightly with large changes in parameter values and assumptions in the analysis.

The one published study of selenium consumption in the affected region (EPA, 1980d) suggests that normal exposure rates are probably around 125 or 135 $\mu\text{g/day}$, below even those calculated for the nominal situation, much less the values found for the various sensitivity studies. Accordingly, we should consider those cases in Table 3.24 having much higher background exposure rates to be less likely than the basic situation.

As before, we calculated the total and individual incremental health risks for the selenium sensitivity analysis cases. The results for the most important cases are shown in Table 3.25. As seen above, under nominal conditions with mean river flows, the lifetime risk of selenosis associated with plant emissions is less than 1.5×10^{-9} , or 1.5 in a billion. This compares to the often cited EPA risk criterion levels of one in a million or one in one hundred thousand (EPA, 1980c). In these sensitivity analysis cases, the incremental individual risk clearly depends strongly on the mean exposure rate (including background). However, except in the worst case, the risk associated with plant emissions remains below one in a million. For these worst cases, however, the background exposure rates rise significantly above the levels measured or calculated in the literature. Moreover, these levels are sufficiently high to cause more than 2300 people to exceed the selenosis threshold. Clearly this situation does not exist, as it would represent a serious health problem that has never been observed.

The exposure rates of the sensitivity analysis cases for beryllium are presented in Table 3.26. This table also includes the total incidence of cancer and incremental individual health risk for members of the exposed population. These health risks were calculated from the exposure rates as described above.

Comparing the worst case with the nominal exposure rates in Table 3.26, we see that although the assumptions significantly change overall exposure rates, the relative contribution of plant emissions remains small. For case BE7 in the nominal situation, plant discharges increase beryllium consumption by 0.01 percent. Under worst-case assumptions, Montour discharges increase beryllium consumption by 0.08 percent in this same case. Without more information about actual beryllium consumption levels in the population, however, we cannot say which assumptions are the most likely.

The health risks associated with plant emissions are not as sensitive to actual beryllium exposure rates as is the case for selenium. This is again a result of the approach used to calculate health risks—a linear dose-response function. In the nominal situation, for example, the maximum incremental individual risk in any of the mean flow cases is 1.8×10^{-8} , or 1.8 in one hundred million. Although this risk increases as the assumptions become more pessimistic, it remains below one in a million even in the worst-case scenarios. This lack of variation arises because the background exposure rate does not affect the incremental individual risk, which is a function only of the added exposure caused by plant discharges.

Table 3.25

TOTAL AND INDIVIDUAL INCREMENTAL HEALTH RISKS FOR SELENIUM

Case	Mean Exposure Rate ($\mu\text{g}/\text{day}$)	Incidence of Cancer		Incremental Individual Risk
		Total	Incremental	
Nominal				
Background	184.055	0.8133	—	—
Mean flows				
E001	184.090	0.8143	0.0010	6.8×10^{-10}
E004	184.092	0.8144	0.0011	7.4×10^{-10}
E007	184.125	0.8154	0.0021	1.4×10^{-9}
Minimum flows				
E005	184.251	0.8191	0.0058	3.9×10^{-9}
E006	184.423	0.8242	0.0109	7.4×10^{-9}
Animals Eat Irrigated Crops				
Background	220.123	2.6322	—	—
Mean flows				
E001	220.243	2.6415	0.0093	6.3×10^{-9}
E004	220.247	2.6418	0.0096	6.5×10^{-9}
E007	220.356	2.6502	0.0180	1.2×10^{-8}
Minimum flows				
E005	220.554	2.6655	0.0333	2.3×10^{-8}
E006	220.931	2.6949	0.0627	4.2×10^{-8}
Double Consumption Rates				
Background	407.272	100.99	—	—
Mean flows				
E001	407.336	101.07	0.0861	5.8×10^{-8}
E004	407.339	101.08	0.0901	6.1×10^{-8}
E007	407.398	101.16	0.1695	1.1×10^{-7}
Minimum flows				
E005	407.646	101.49	0.5037	3.4×10^{-7}
E006	407.974	101.94	0.9470	6.4×10^{-7}
Worst Case with Constant Background				
Background	778.698	2384.5	—	—
Mean flows				
E001	779.058	2389.2	4.78	3.2×10^{-6}
E004	779.070	2389.4	4.94	3.3×10^{-6}
E007	779.396	2393.7	9.28	6.3×10^{-6}
Minimum flows				
E005	780.092	2403.0	18.56	1.3×10^{-5}
E006	781.314	2419.4	34.90	2.4×10^{-5}

NOTE: Exposed population is 1,478,785.

Table 3.26

EXPOSURE RATES AND HEALTH RISKS FOR BERYLLIUM

Case	Mean Exposure Rate (ng/day)	Incidence of Cancer		Incremental Individual Risk
		Total	Incremental	
Nominal				
Background	1238.13	111.88	—	—
Mean flows				
E002	1238.27	111.89	0.0121	8.2×10^{-9}
E005	1238.41	111.90	0.0248	1.7×10^{-8}
E007	1238.28	111.89	0.0127	8.5×10^{-9}
E007B	1238.42	111.90	0.0259	1.8×10^{-8}
Minimum flows				
E002C	1238.90	111.95	0.0686	4.6×10^{-8}
E005C	1239.70	112.02	0.140	9.5×10^{-8}
E007A	1239.08	111.96	0.0847	5.7×10^{-8}
E007C	1240.06	112.05	0.172	1.1×10^{-7}
Animals Eat Irrigated Crops				
Background	1239.45	112.04	—	—
Mean flows				
E002	1240.48	112.13	0.094	6.3×10^{-8}
E005	1241.56	112.23	0.190	1.3×10^{-7}
E007	1240.52	112.14	0.097	6.5×10^{-8}
E007B	1241.62	112.24	0.196	1.3×10^{-7}
Minimum flows				
E002C	1242.28	112.30	0.255	1.7×10^{-7}
E005C	1245.21	112.56	0.519	3.5×10^{-7}
E007A	1242.78	112.34	0.300	2.0×10^{-7}
E007C	1246.22	112.65	0.610	4.1×10^{-7}
Double Consumption Rates				
Background	1983.79	89.614	—	—
Mean flows				
E002	1983.97	89.622	0.0081	5.5×10^{-9}
E005	1984.16	89.630	0.0166	1.1×10^{-8}
E007	1983.98	89.622	0.0086	5.8×10^{-9}
E007B	1984.18	89.631	0.0175	1.2×10^{-8}
Minimum flows				
E002C	1985.12	89.673	0.0594	4.0×10^{-8}
E005C	1986.50	89.735	0.1210	8.2×10^{-8}
E007A	1985.46	89.688	0.0743	5.0×10^{-8}
E007C	1987.18	89.765	0.1510	1.0×10^{-7}

Table 3.26 (Continued)

Worst Case with Constant Background				
Background	3687.69	166.56	—	—
Mean flows				
E002	3690.48	166.68	0.126	8.5×10^{-8}
E005	3693.38	166.81	0.257	1.7×10^{-7}
E007	3690.59	166.69	0.131	8.9×10^{-8}
E007B	3693.58	166.82	0.266	1.8×10^{-7}
Minimum flows				
E002C	3698.75	167.05	0.496	3.4×10^{-7}
E005C	3710.20	167.57	1.010	6.8×10^{-7}
E007A	3700.84	167.15	0.590	4.0×10^{-7}
E007C	3714.45	167.76	1.200	8.1×10^{-7}
Worst Case				
Background	3922.46	177.12	—	—
Mean flows				
E002	3925.25	177.24	0.126	8.5×10^{-8}
E005	3928.15	177.38	0.257	1.7×10^{-7}
E007	3925.36	177.25	0.131	8.9×10^{-8}
E007B	3928.36	177.38	0.266	1.8×10^{-7}
Minimum flows				
E002C	3933.52	177.61	0.496	3.4×10^{-7}
E005C	3944.97	178.13	1.010	6.8×10^{-7}
E007A	3935.61	177.71	0.590	4.0×10^{-7}
E007C	3949.22	178.32	1.200	8.1×10^{-7}

NOTE: Exposed population is 1,478,785.

Conclusions

From these results one can conclude that plant selenium and beryllium discharges into the Chillisquaque should have a negligible effect on the potential for cancer and selenosis in the regional population. Moreover, the incremental exposure rates associated with these discharges are below measurable levels and can only be seen when the results are presented to more significant digits than are warranted by the data. Although certain parameters in the analysis may be relatively uncertain, the general results are robust with respect to these parameters. Under varying assumptions about bioconcentration factors, general background levels, and consumption rates, both the incremental pollutant exposure rates and individual health risks remain relatively constant. Based on the EPA selenosis threshold level of 7 mg/day, even the worst-case assumptions do not cause population exposure rates to increase significantly.

The general conclusions apply to both beryllium and selenium discharges from the Montour plant. Plant discharges are diluted significantly in the Susquehanna River. The consequent pollutant concentrations are far below detectable levels or levels that would lead to any measurable health risk. For the Chillisquaque Creek, pollutant concentrations associated with Montour discharges may be measurable. However, the calculated increases in concentrations during normal flow conditions are not large compared to background levels, and the pollutants in the Chillisquaque could affect only a small fraction of the exposed population. Even in this small fraction of the population, any increased exposure from plant operations would be so small compared to background exposure levels that no measurable health effects would be expected.

Appendix A

MASS BALANCE CALCULATIONS

The general subject of plant emissions has been discussed extensively in the previous reports. Although that discussion will not be repeated here, we will present the relevant equations, calculations, and measured data needed to describe both the selenium and beryllium emissions from the Montour SES. These calculations are necessary to understand the pollutant mass balance within the power plant as described in Sec. III. Moreover, they also (1) increase our understanding of transport and distribution of the pollutant within the power plant, (2) provide a comparison for the measured emission rates, and (3) supplement the measurements in the sensitivity analysis.

The equations themselves are simple linear equations with partitioning or conversion factors as coefficients. Where possible, actual measurements of plant emissions are used in the risk analysis. Ideally, the calculations described below should give discharge rates comparable to the measured values.

POLLUTANT INFLOWS

The pollutant, be it selenium or beryllium, can enter power plant systems through the coal supply, makeup water, treatment chemicals, combustion air, miscellaneous oils, or removal from plant equipment. At the Montour plant, no water treatment chemicals contain either selenium or beryllium and no appreciable amounts are removed from plant equipment during operation or maintenance. Similarly, contributions from combustion air and oils are minor. As a consequence, the primary sources for both pollutants are the input coal and plant makeup water. We will discuss each of these sources in turn before considering the fate of the input pollutant.

Coal Consumption

The amount of pollutant entering the plant in the raw coal can be calculated from the daily coal consumption rate using

$$P_c = 1000.0M_cp_c \quad (\text{A.1})$$

where P_c = pollutant content of coal consumed (kg/day),
 p_c = pollutant concentration in coal (kg/kg),¹
 M_c = daily coal consumption (metric tons/day),
 1000.0 = conversion factor (kg/metric ton).

To estimate the quantity of pollutant entering the power plant in the coal, one must first characterize this coal. The Montour plant uses coal from four sources in Western Pennsylvania (Plangemann, 1983). The characteristics of the coals from each source, the relative fractions used, and the consequent mean values are presented in Table A.1.

No beryllium or selenium concentration measurements have been made for these coals. Beryllium concentrations in coal can vary between 0.16 ppm and 4.0 ppm, and selenium concentrations can vary between 0.4 ppm and 8.0 ppm (Bahor et al., 1981; Braunstein et al., 1977; Kuhn et al., 1980). Smaller variations will occur within and between seams at a given location. In the above literature, as well as in Dvorak et al. (1977) and IARC (1980), the average beryllium concentration in eastern or Appalachian coal is found to be between 0.8 ppm and 2.5 ppm. Accordingly, for this analysis we assume that the coal used at the Montour has a mean beryllium content of 2.0 ppm. According to Spigarelli (1976), the coal used at the Montour plant is assumed to have a mean selenium content of 4.8 ppm, with minimum and maximum concentrations of 3.9 and 5.3 ppm.

As stated above, we can calculate the amount of pollutant entering the power plant in the coal using Eq. (A.1). If the mean coal consumption rate M_c is 9,328.8 t/day and the mean pollutant concentrations in coal are as shown in Table A.1, the average input rates for beryllium and selenium in coal are 18.66 kg/day and 44.78 kg/day.

Table A.1
 MONTOUR COAL CHARACTERISTICS

Item	Source				Mean Value
	Greenwich	Chers Pride	Benjamin	Hoffman	
Fraction (1983)	0.70	0.03	0.13	0.14	—
Moisture (%)	6.2	6.0	5.9	7.5	6.3
Ash (%)	15.0	13.7	13.5	11.1	14.2
Heat content (Btu/lb)	12,358	12,334	12,445	12,746	12,423
Sulfur (%)	1.4	1.8	1.7	1.8	1.5
Beryllium (ppm)	—	—	—	—	2.0
Selenium (ppm)	—	—	—	—	4.8

SOURCES: Spigarelli (1976), and Plangemann (1983).

¹Gravimetric concentration or fraction by weight.

Makeup Water

Montour SES officials do not keep detailed records of makeup water use for the plant. However, on the basis of calculated water use information from PP&L (1982a and 1982b), we can estimate the mean demand for makeup water to be about $1.09 \text{ m}^3/\text{sec}$. The mean pollutant inflow rates from plant makeup water can then be found from

$$P_{mw} = 0.001 p_{mw} Q_{mw} \quad (\text{A.2})$$

where P_{mw} = pollutant inflow from makeup water (kg/day),

p_{mw} = pollutant content of makeup water (mg/l),

Q_{mw} = mean flow rate for makeup water (m^3/day),

0.001 = conversion factor from mg/l to kg/m^3 .

The measurements of beryllium concentration in the makeup water vary significantly. The water monitoring performed by PP&L under its Toxics Reduction Evaluation program (Skinner, 1986) found total concentrations (dissolved plus particulate) that ranged between less than $0.100 \text{ } \mu\text{g}/\text{l}$ (the detection limit) and $1.800 \text{ } \mu\text{g}/\text{l}$. This range undoubtedly reflects a combination of real variation in time and space and possible errors in the sampling and analytical processes.

Aside from this difficulty, however, the results create another problem. The preponderance of measurements below detection limits artificially raise the mean value of the distribution by truncating it above the true concentrations. In this situation, using a mean value as the measure of discharge concentration is not valid. Instead, it is more appropriate to use the median, a nonparametric estimator for nonnormal distributions. Accordingly, we will assume a value of $0.10 \text{ } \mu\text{g}/\text{l}$ for the beryllium concentration in the makeup water. For the purposes of sensitivity analysis, we will also perform the calculations of beryllium emissions using a mean value of $0.40 \text{ } \mu\text{g}/\text{l}$, calculated assuming all measurements below the detection limit were at one-half of this limit.

The selenium concentration in the makeup water also varies significantly over time or between measurements. Two sets of measurements reported by Analytical Laboratories (1983 and 1984) give concentrations of $12 \text{ } \mu\text{g}/\text{l}$ and less than $2 \text{ } \mu\text{g}/\text{l}$. Subsequent measurements by PP&L have found maximum concentrations of $3 \text{ } \mu\text{g}/\text{l}$ or less (Stine, 1985; Skinner, 1986) with a median value of 1.00 and a mean of $1.38 \text{ } \mu\text{g}/\text{l}$. Accordingly, because the concentration is most likely below $3.0 \text{ } \mu\text{g}/\text{l}$, we assume a nominal selenium concentration in makeup water of $2.0 \text{ } \mu\text{g}/\text{l}$ and a value of $10.0 \text{ } \mu\text{g}/\text{l}$ for sensitivity analysis. Using these concentrations and the assumed values for beryllium, we can calculate the pollutant input rates for the power plant. These results are presented in Table A.2.

Table A.2
 POLLUTANT INPUT RATES
 (kg/day)

Source	Selenium Concentration in Makeup Water ($\mu\text{g/l}$)		Beryllium Concentration in Makeup Water ($\mu\text{g/l}$)	
	2.0	10.0	0.10	0.40
Intake water	0.188	0.942	0.0094	0.0377
Coal	44.778	44.778	18.658	18.658
Total	44.967	45.720	18.667	18.695

POLLUTANT EMISSIONS AND SINKS

The beryllium and selenium that enter the plant may either remain there or be released to the environment in several ways. The most important of these release mechanisms are:

- Waterborne release via direct discharge from the plant or as runoff from coal piles and ash disposal areas.
- Atmospheric release in flue gas and fly ash or as dust from coal piles and ash disposal sites.
- Discharge to soil and groundwater by leaching from coal piles, ash disposal areas, and FGD sludge disposal areas.

At Montour, leachate from the coal pile and ash disposal areas is collected and either recirculated or sent to the detention basin. Seepage from the ash basin is also collected and returned to the storm water basin, from which it is discharged into the Chillisquaque. The flow rate for this waste stream, however, has typically averaged less than $0.002 \text{ m}^3/\text{s}$, making it a negligible source of beryllium compared to the detention basin overflow under normal conditions. However, measurements made in 1985 indicate that average storm water basin flow increased to $0.0132 \text{ m}^3/\text{s}$. Under these conditions, the basin should be considered as a pollutant source for the surface water analysis.

Unfortunately, it is not possible to calculate the amount of beryllium and selenium entering or leaving the storm water basin. Consequently, in this mass balance analysis, we will consider only the potential atmospheric discharges of pollutants and the direct discharges to surface water from the detention basin.

Atmospheric Emissions

For the wet bottom, tangentially fired, pulverized-coal furnaces used at Montour, the ash production is partitioned as 85 percent fly ash (FA) and 15 percent bottom ash (BA) (EEI, 1983). The partitioning factor method of Dvorak et al. (1977) and Klein et al. (1975) can be used to calculate the distribution of pollutants in plant emissions.² This approach will allow us to calculate the distribution of pollutant between fly ash and bottom ash. Pollutant concentrations in the two types of ash can then be used to determine the amount entering the bottom ash transport system and subsequently the detention basin.

To apply the method, we first determine with Eq. (A.1) how much of the pollutant enters the plant through its daily coal consumption. With this information, the amount of pollutant leaving the furnace in the collected bottom ash can be found using the partitioning factors from

$$P_{ba} = P_c F_{ba} f_{ba} \quad (\text{A.3})$$

where P_{ba} = pollutant content of bottom ash (kg/day),
 P_c = pollutant content of coal consumed (kg/day),
 F_{ba} = bottom ash fraction of total ash (dimensionless),
 f_{ba} = partitioning factor for bottom ash (dimensionless).

The partitioning factor represents the ratio of pollutant concentration in the bottom ash to concentration in the coal. Klein et al. (1975) assign a value of 0.0039 to this fraction for selenium.³ Unfortunately Klein et al. (1975) and other investigators have not followed beryllium through the combustion process to determine the value of the fraction. However, Coles et al. (1979) found that the behavior of beryllium during combustion put it in the class of intermediate elements, including barium, cobalt, chromium, copper, nickel, strontium, uranium, and vanadium. A comparison of the relative concentrations in coal and ash fractions indicates that beryllium seems to behave very similarly to copper during combustion. Accordingly, we used to copper bottom ash partitioning fraction (0.25) for beryllium in the analysis. As described above and shown in Fig. 3.2, bottom ash and pyrites are collected and sluiced to the ash basin for treatment. Overflow from this basin flows

²This method measures the weight fraction of pollutant in the various ash fractions, input coal, and flue gas to characterize the behavior of different elements during the combustion process.

³Limited data from PP&L (1984a) and Spigarelli (1976) indicate that the selenium concentration in Montour fly ash can vary between 12 ppm and 70 ppm, with a mean of about 41 ppm. As shown in Table A.3, the calculated selenium concentration in fly ash is 39.7 ppm using the assumed partitioning factor.

by gravity to the detention basin for combination with other plant wastewaters.

The amount of pollutant in the fly ash can be found by subtracting the amount in bottom ash from the total in the coal, as shown below.

$$P_{fa} = P_c - P_{ba} \quad (\text{A.4})$$

where P_{fa} = pollutant content of fly ash (kg/day)

Trace element pollutants in the fly ash will be captured to some extent by the electrostatic precipitators (ESP) used to collect the fly ash from each unit. For this analysis, it is assumed that each ESP has a particulate removal efficiency of 99.0 percent.⁴ The total fly ash and bottom ash generated by the plant can be found from

$$G_{fa} = 1000.0 M_c A_c F_{fa} \quad (\text{A.5})$$

$$G_{ba} = 1000.0 M_c A_c F_{ba} \quad (\text{A.6})$$

where G_{fa} = fly ash generation rate (kg/day),

G_{ba} = bottom ash generation rate (kg/day),

A_c = ash content of coal (dimensionless),

F_{fa} = fly ash fraction of total ash (dimensionless).

Given an assumed control efficiency, fly ash collection and emission rates can easily be determined using

$$D_{fa} = G_{fa} e_{pt} \quad (\text{A.7})$$

$$E_{fa} = G_{fa} - D_{fa} \quad (\text{A.8})$$

where D_{fa} = fly ash collection rate (kg/day),

E_{fa} = fly ash emission rate (kg/day),

e_{pt} = particulate control efficiency (dimensionless).

With this information, one can calculate the amount of pollutant emitted to the air by the plant and the amount retained in the collected fly ash. Emitted pollutant is given by

$$P_{efa} = P_{fa} (1.0 - e_{pt}) f_{fa} \quad (\text{A.9})$$

where P_{efa} = pollutant emitted in stack gas (kg/day),

f_{fa} = partitioning factor for emitted fly ash (dimensionless).

⁴Beryllium and selenium will not be distributed uniformly with respect to particle size, and the ESP does not have a 99.0 percent collection efficiency for all sizes. These factors are considered below in the calculations.

Retained (collected) pollutant is determined by subtracting emitted pollutant from total pollutant as shown below.

$$P_{rfa} = P_c - P_{ba} - P_{efa} \quad (\text{A.10})$$

where P_{rfa} = pollutant retained in collected fly ash (kg/day).

This last result represents the total amount of pollutant retained in the fly ash. All of this collected beryllium and selenium will be transported to the dry fly ash storage area. Runoff and leachate from this site are collected and pumped to the detention basin for treatment with other plant wastes.

Using Eqs. (A.1) through (A.10) in sequential order with data describing the Montour plant, we have calculated the expected amounts of beryllium and selenium emitted to the air and collected in bottom ash and fly ash. Assumed values, data, results and appropriate equation numbers are shown in Tables A.3A and A.3B. To save space, Table A.3B only presents values that differ from Table A.3A.

Table A.3A
ATMOSPHERIC EMISSION CALCULATIONS AND
ASSUMPTIONS—BERYLLIUM

Variable	Equations	Calculated or Assumed Value
M_c	A.1	9,328.8 t/day
P_c	A.1	2.0×10^{-6} kg/kg
P_c	A.1/A.3/A.4	18.66 kg/day
F_{ba}	A.3/A.6	0.15
f_{ba}	A.3	0.25
P_{ba}	A.3/A.4/A.10	0.6997 kg/day
P_{fa}	A.4/A.9	17.96 kg/day
A_c	A.5	0.142
F_{fa}	A.5	0.85
G_{fa}	A.5/A.7/A.8	1.126×10^6 kg/day
G_{ba}	A.6	1.987×10^5 kg/day
e_{pt}	A.7/A.9	0.990
D_{fa}	A.8	1.115×10^6 kg/day
E_{fa}	A.8	1.126×10^4 kg/day
f_{fa}	A.9	2.25
P_{efa}	A.9/A.10	0.404 kg/day
P_{rfa}	A.10	17.554 kg/day

Table A.3B
ATMOSPHERIC EMISSION CALCULATIONS AND
ASSUMPTIONS—SELENIUM

Variable	Equations	Calculated or Assumed Value
P_c	A.1	4.8×10^{-6} kg/kg
P_c	A.1/A.3/A.4	44.78 kg/day
f_{ba}	A.3	0.0039
P_{ba}	A.3/A.4/A.10	0.02620 kg/day
P_{fa}	A.4/A.9	44.75 kg/day
f_{fa}	A.9	3.5
P_{efa}	A.9/A.10	1.566 kg/day
P_{rfa}	A.10	43.186 kg/day

Surface Water Sinks and Emissions

Power plants may discharge to surface waters a large number of different waste streams. As shown in Fig. 3.2, discharges from many of the minor streams (totaling roughly $0.156 \text{ m}^3/\text{s}$) are collected and transported to the miscellaneous plant waste basin and from there to the detention basin for final treatment. Normally, most of these waste streams will not contain significant amounts of beryllium or selenium not already present in plant intake water. Consequently, we will not consider them in detail now, but will instead concentrate on potentially more important emission sources.

Coal Pile Runoff. For the mean annual precipitation rate of about 43 inches, PP&L engineers have calculated that runoff from the coal storage pile averages $0.00262 \text{ m}^3/\text{s}$ or 0.06 mgd (million gallons per day) (PP&L, 1982b). As shown in Fig. 3.2, this runoff is collected and discharged into the detention basin with other plant wastes. The amount of beryllium or selenium leached from the stored coal by precipitation depends on a number of factors, including the pH of the infiltrating water and the characteristics of the rainfall and coal pile. Because the problem is so complex, and because no measurements have been made at the Montour plant, for this calculation we will assume that the pollutant concentrations in coal pile runoff are equivalent to one-half percent of that in the raw coal. Given the runoff flow rate and the beryllium and selenium concentrations in the raw coal, one can calculate the content in the runoff water from

$$P_{cp} = 86.4k_rRp_c \quad (\text{A.11})$$

where P_{cp} = pollutant content of runoff water (kg/day),
 k_r = empirical constant (0.005 mg/l per mg/kg),
 R = runoff flow rate (m^3/s),
 p_c = pollutant concentration in coal (mg/kg),
 86.4 = conversion factor ($\text{kg-l-s/mg-m}^3\text{-day}$).

The results of this calculation are presented in Table A.4.⁵

Cooling Water. The Montour power plant uses natural draft, crossflow cooling towers. The only significant discharge from this system is cooling tower blowdown, at a nominal rate for the system of $0.218 \text{ m}^3/\text{s}$. As shown in Fig. 3.2, the combined blowdown for both units is used for bottom ash and pyrite sluicing and is discharged into the ash basin. No beryllium or selenium are present in any of the chemicals added to the cooling water for control of biological growth, scale, or corrosion. Consequently, the only beryllium or selenium in the cooling water discharge will be that present in the makeup water, although the concentration may be greater because the cooling water is recycled through the system (EPA, 1980e). For this study, we have assumed that the cooling system operates at four cycles of concentration (PP&L, 1983). If the beryllium concentration in the makeup water is $0.1 \text{ } \mu\text{g/l}$ and the selenium concentration 0.002 mg/l , and no pollutants remain within the cooling system and its sludge, the

Table A.4

POLLUTANT DISCHARGES IN COAL PILE RUNOFF

Variable	Calculated or Assumed Value
Storage pile area	21.0 acres
p_c —beryllium	2.0 mg/kg
selenium	4.8 mg/kg
Nominal precipitation	43.0 in./yr
R—nominal	$0.00262 \text{ m}^3/\text{s}$
P_{cp} —beryllium	0.0023 kg/day
selenium	0.0054 kg/day

⁵Limited data found in EPA (1980e) indicate that the concentration of beryllium in coal pile runoff can vary between less than 0.01 and 0.07 mg/l. Similarly, selenium concentrations in coal pile runoff can vary between 0.001 and 0.03 mg/l, with a mean value around 0.005 mg/l. Our values of 0.01 mg/l and 0.024 mg/l thus fall within the ranges, although near the extremes.

concentrations in the blowdown should average $0.4 \mu\text{g/l}$ for beryllium and 0.008 mg/l for selenium. These pollutants will enter the ash basin with the pollutants contained in the bottom ash and pyrites.

Ash Transport Water. The Montour plant uses wet handling for bottom ash and pyrites but has installed dry fly ash handling and transport systems on both units. Other wastes entering the ash basin include economizer water wash, sludge from the miscellaneous plant waste basin, detention basin sludge, precipitator washes, and cooling tower sludge. Overflow from the ash basin (approximately $0.214 \text{ m}^3/\text{s}$) is discharged into the detention basin.

A concentration of pollutant in the ash basin water can be calculated based on a number of simplifying assumptions. First, we assume that no pollutant is returned to the ash basin in sludges from the miscellaneous plant waste or detention basins. Because our objective is to determine the amount of pollutant in the detention basin, this assumption implies that once the pollutant has entered the detention basin, it will either remain there or be discharged in the overflow. This assumption will make the final emission rates more pessimistic, because it leaves more pollutant in the detention basin. Second, no beryllium or selenium enters the basin in either the economizer water wash or precipitator washes. The amount of pollutant from either source should be small because these processes involve low volumes of waste. Third, because we previously assumed that all beryllium and selenium in the cooling system blowdown remained in the water, none enters the ash basin in the sludge from the cooling towers.

Given these assumptions, beryllium and selenium will enter the ash basin only in the ash transport water. This transport water (cooling tower blowdown) will contain the pollutant from the collected ash as well as that concentrated from the makeup water. The total mass of either pollutant entering the basin can thus be found from

$$P_s = 0.001cp_{mw}Q_{ba} + P_{ba} \quad (\text{A.12})$$

where P_s = pollutant inflow from ash transport water (kg/day),

0.001 = conversion factor (kg-l/mg-m^3),

c = cycles of concentration in cooling system (dimensionless),

p_{mw} = pollutant concentration in makeup water (m^3/day),

Q_{ba} = bottom ash transport water flow (m^3/day),

P_{ba} = pollutant content in bottom ash (kg/day).

The concentration of pollutant in the ash basin overflow will depend on how much of it is trapped in the collected ash and how much

remains in the overflow water. Much of the entering pollutants should settle out in the basin with the ash and pyrites, but many different factors could affect this process and the resulting concentration in basin overflow in both the short and long run. In addition to chemical conditions in the basin, which might lead to a long-run balance between inflow and outflow of the material, the previous fly ash disposal and basin dredging could affect pollutant accumulation and discharges. Before the dry fly ash system began operation, collected fly ash was deposited in the basin for treatment and disposal. As a result, a significant accumulation of pollutant could have occurred during the ten years of operation before conversion. With the large decrease in pollutant loading and concentration during the four years since conversion, previously accumulated beryllium or selenium could be coming out of basin sediments. Conversely, if the ash basin were dredged regularly, this operation would remove pollutant in the sediments and reduce the basin content and corresponding discharge rate.

As a consequence of these complex and unknown influences, it is difficult to estimate the balance between the inflow and outflow of pollutant from the ash basin. Accordingly, we can calculate the outflow from the basin parametrically as a function of the ratio of inflow to outflow of pollutant. Given the inflow to the basin, the outflow and accumulation rates for beryllium or selenium can be calculated from

$$P_b = kP_s \quad (\text{A.13})$$

$$P_{aa} = P_s - P_b \quad (\text{A.14})$$

where P_b = pollutant content of ash basin overflow (kg/day),
 k = ratio of mean outflow to inflow rate (dimensionless),
 P_s = pollutant inflow from ash transport water (kg/day),
 P_{aa} = pollutant accumulation rate in ash basin (kg/day).

Using Eqs. (A.12) through (A.14) with the previous results and data from plant operations, we can calculate the beryllium and selenium content of ash basin overflow and accumulation rate in the ash basin for various values of the parameter k . The inputs and assumed values for these calculations are shown in Table A.5. The results of the calculations for alternative pollutant concentrations in plant makeup water and alternative k ratios are presented in Tables A.6A and A.6B. Beryllium concentrations in the ash basin overflow were measured during the TRE monitoring program. Over the twelve month period, the total concentration ranged between less than 0.1 and 0.69 $\mu\text{g/l}$, with a median value of 0.16 and a mean of 0.23 $\mu\text{g/l}$ (Skinner, 1986). For a background concentration of 0.10 $\mu\text{g/l}$ in the makeup water, these

Table A.5
INPUTS FOR ASH BASIN LOADING CALCULATIONS

Variable	Equation	Calculated or Assumed Value
Q_{ba}	A.12	18,500 m ³ /day
P_{ba} —beryllium	A.12	0.6997 kg/day
selenium		0.0262 kg/day
c	A.12	4.0 cycles
p_{mw} —beryllium	A.12	0.1/0.4 µg/l
selenium		0.002/0.010 mg/l
k —beryllium	A.13	0.001/0.005/0.01/0.02
selenium		0.05/0.50/1.00/2.00

Table A.6A
BERYLLIUM OUTFLOW AND ACCUMULATION
IN ASH BASIN
(kg/day)

Variable	Beryllium Content of Makeup Water (µg/l)	
	0.10	0.40
P_s	0.7071	0.7293
P_b		
$k = 0.001$	0.00071	0.00073
$k = 0.005$	0.00354	0.00365
$k = 0.01$	0.00707	0.00729
$k = 0.02$	0.01414	0.01459
P_{aa}		
$k = 0.001$	0.7064	0.7286
$k = 0.005$	0.7036	0.7257
$k = 0.01$	0.7000	0.7220
$k = 0.02$	0.6930	0.7147

Table A.6B

SELENIUM OUTFLOW AND ACCUMULATION
IN ASH BASIN

(kg/day)		
Variable	Selenium Content of Makeup Water ($\mu\text{g/l}$)	
	0.10	0.40
P_s	0.174	0.766
P_b		
$k = 0.05$	0.009	0.038
$k = 0.50$	0.087	0.383
$k = 1.00$	0.174	0.776
$k = 2.00$	0.348	1.532
P_{aa}		
$k = 0.05$	0.165	0.728
$k = 0.50$	0.087	0.383
$k = 1.00$	0.000	0.000
$k = 2.00$	-0.174	-0.776

concentrations correspond to k values of 0.00419 and 0.00583, respectively.⁶

Detention Basin. The overflow from the ash basin enters the detention basin where it mixes with other plant wastes. The overflow from the detention basin is then discharged to the Chillisquaque Creek. In addition to the ash basin discharge, beryllium and selenium can enter the detention basin in coal pile runoff (calculated above) or in the other wastewater streams in the plant waste line. From PP&L (1982a and 1982b) the total flow from these streams is $0.152 \text{ m}^3/\text{s}$. Although pollutants may be concentrated by the processes generating some of these streams (e.g., demineralizer regeneration, filter backwashes, or ash hopper overflows), the amount of pollutant in this stream should be the difference between the total entering the plant in

⁶The TRE program also measured the beryllium concentration in the ash transport water. The mean concentration was found to be $0.80 \mu\text{g/l}$. This would indicate that the amount of beryllium entering the ash basin is only about 1 percent of what our calculations show. It is hard to believe that so little beryllium ends up in the bottom ash during combustion. One explanation might be that the beryllium concentration in the ash transport water does not accurately reflect the amount of particulate beryllium being carried by the water.

the makeup water and that passing through the cooling system. Under these circumstances, the contribution to the detention basin can be calculated from

$$P_m = 0.001Q_m p_{mw} \quad (\text{A.15})$$

where P_m = pollutant content of miscellaneous wastewater (kg/day),
 p_{mw} = pollutant concentration in makeup water (mg/l),
 Q_m = daily flow rate of miscellaneous wastewater (m^3/day),
 0.001 = conversion factor (1-kg/mg- m^3).

The miscellaneous wastewater streams are collected and passed through the miscellaneous plant waste basin before entering the detention basin (see Fig. 3.2). Some beryllium and selenium may be removed from the water and retained in this basin, but to simplify the analysis we assume that all of the pollutant entering this basin passes through into the detention basin. Pollutants retained in the detention basin will have the same effect as those retained in this basin for the type of calculations we are performing.

Using Eq. (A.15) with previous results from Table A.6A and A.6B and the two pollutant concentrations in plant makeup water, we can calculate the beryllium and selenium inflow rates to the detention basin for different values of k . Tables A.7A and A.7B present these results.

As with the ash basin, some pollutant may settle out and be retained in the sediments of the detention basin. This should be small for beryllium, however, because so much is retained in the ash basin (according to the value of k). Accordingly, the outflow and accumulation rates for pollutants in the detention basin, given the inflow, can be found from

$$P_d = mP_w \quad (\text{A.16})$$

$$p_d = 1000.0P_d / Q_d \quad (\text{A.17})$$

where P_d = pollutant content of detention basin overflow (kg/day),
 m = ratio of mean outflow to inflow rate (dimensionless),
 P_w = pollutant content of inflow to detention basin (kg/day),
 p_d = pollutant concentration in detention basin discharge (mg/l),
 Q_d = discharge rate of detention basin (m^3/day).

Using Eqs. (A.16) and (A.17) with the previous results in Tables A.7A and A.7B and the detention basin overflow rate of $0.395 \text{ m}^3/\text{s}$ ($34,132 \text{ m}^3/\text{day}$), we can calculate the pollutant content of detention basin overflow and the pollutant concentration in the discharge for various

Table A.7A

BERYLLIUM INFLOW TO THE DETENTION BASIN

(kg/day)

Beryllium Source	Beryllium Content of Makeup Water ($\mu\text{g/l}$)	
	0.10	0.40
Ash basin		
k = 0.001	0.00071	0.00073
k = 0.005	0.00354	0.00365
k = 0.010	0.00707	0.00729
k = 0.020	0.01414	0.01459
Miscellaneous	0.00131	0.00525
Coal pile runoff	0.00226	0.00226
Total		
k = 0.001	0.00428	0.00824
k = 0.005	0.00711	0.01116
k = 0.010	0.01064	0.01481
k = 0.020	0.01771	0.02211

Table A.7B

SELENIUM INFLOW TO THE DETENTION BASIN

(kg/day)

Selenium Source	Selenium Content of Makeup Water (mg/l)	
	0.10	0.40
Ash basin		
k = 0.05	0.009	0.038
k = 0.50	0.087	0.383
k = 1.00	0.174	0.776
k = 2.00	0.348	1.532
Miscellaneous	0.026	0.131
Coal pile runoff	0.005	0.005
Total		
k = 0.05	0.041	0.175
k = 0.50	0.119	0.520
k = 1.00	0.206	0.913
k = 2.00	0.380	1.669

values of the parameter m . The results of the calculations for the two makeup water concentrations and alternative m ratios are presented in Tables A.8A and A.8B.

During 1985, Pennsylvania Power and Light made a number of measurements of the beryllium and selenium concentrations in the

Table A.8A

BERYLLIUM DISCHARGE RATE AND CONCENTRATION

Inflow (kg/day)	k	Discharge Rate (kg/day)				Concentration (mg/l)			
		m = 0.10	0.25	0.50	1.00	0.10	0.25	0.50	1.00
Background—0.10 $\mu\text{g/l}$									
0.0043	0.001	0.0004	0.0011	0.0021	0.0043	0.00001	0.00003	0.00006	0.00013
0.0071	0.005	0.0007	0.0018	0.0036	0.0071	0.00002	0.00005	0.00010	0.00021
0.0106	0.010	0.0011	0.0027	0.0053	0.0107	0.00003	0.00008	0.00016	0.00031
0.0177	0.020	0.0018	0.0044	0.0089	0.0177	0.00005	0.00013	0.00026	0.00052
Background—0.40 $\mu\text{g/l}$									
0.0082	0.001	0.0008	0.0021	0.0041	0.0083	0.00002	0.00006	0.00012	0.00024
0.0112	0.005	0.0011	0.0028	0.0056	0.0112	0.00003	0.00008	0.00016	0.00033
0.0148	0.010	0.0015	0.0037	0.0074	0.0148	0.00004	0.00011	0.00022	0.00043
0.0221	0.020	0.0022	0.0055	0.0111	0.0221	0.00006	0.00016	0.00032	0.00065

Table A.8B

SELENIUM DISCHARGE RATE AND CONCENTRATION

Inflow (kg/day)	k	Discharge Rate (kg/day)				Concentration (mg/l)			
		m = 0.05	0.50	1.00	2.00	0.05	0.50	1.00	2.00
		Background—0.002 mg/l							
0.041	0.05	0.0021	0.0205	0.041	0.082	0.0001	0.0006	0.0012	0.0024
0.119	0.50	0.0060	0.0595	0.119	0.238	0.0002	0.0017	0.0035	0.0070
0.206	1.00	0.0103	0.103	0.206	0.412	0.0003	0.0030	0.0060	0.0121
0.380	2.00	0.0190	0.190	0.380	0.760	0.0006	0.0056	0.0111	0.0223
Background—0.010 mg/l									
0.175	0.05	0.0088	0.0875	0.175	0.350	0.0003	0.0026	0.0051	0.0103
0.520	0.50	0.0260	0.269	0.520	1.040	0.0008	0.0076	0.0152	0.0305
0.913	1.00	0.0457	0.457	0.913	1.826	0.0013	0.0134	0.0268	0.0535
1.669	2.00	0.0835	0.835	1.669	3.338	0.0024	0.0245	0.0489	0.0978

detention basin overflow (Skinner, 1986). As discussed in Sec. II, these measurements found a median beryllium concentration in the overflow of $0.22 \mu\text{g/l}$ and median selenium concentration of 0.001 mg/l . This information, when combined with the measurements of ash basin discharge described above, indicates that the value of m for beryllium in the detention basin probably lies somewhere around 1.0. For selenium, the value of m should be between 0.5 and 1.0, based on a k value of about 0.1. Consequently, most of the beryllium and selenium settle out in the ash basin and the remainder that passes into the detention basin remains in solution or suspension and passes out of the basin in the overflow.

Appendix B

HEALTH EFFECTS OF SELENIUM AND BERYLLIUM

HEALTH EFFECTS OF BERYLLIUM

The primary routes of beryllium intake are inhalation and ingestion. Acute health effects have most commonly been associated with inhalation of concentrations greater than $100 \mu\text{g}/\text{m}^3$ of beryllium, which leads to chemical pneumonitis and possibly death. Chronic health effects can probably be caused by long-term inhalation of concentrations above $0.1 \mu\text{g}/\text{m}^3$, although this level has not been verified. Dermal contact with soluble beryllium salts can also lead to contact dermatitis and other dermatological effects in occupational exposures, but there is no evidence of such reactions in the general population (Tepper, 1980; EPA, 1980a).

Once the exposure rate of an individual to beryllium has been estimated, the corresponding risk to that person may be determined from an appropriate dose-response relationship. Although both inhalation and ingestion routes of exposure are possible for power plant sources, we will focus our attention on ingestion only. The principal risk that will be considered is carcinogenicity. This end effect leads to the lowest allowable estimate of chronic exposure and thus underlies the water quality criterion set by EPA (1980a).

Estimating carcinogenic risk generally involves the analysis of two phenomena—the distribution and metabolism of the toxic agent, i.e., pharmacokinetics, and the frequency with which various exposures of an agent induce tumor formation, i.e., dose-response relationships. In the case of beryllium, pharmacokinetics provides information about the amount of beryllium absorbed from the gastrointestinal tract following ingestion and its final tissue destination in the body, this latter observation providing clues about likely target tissues. Dose-response data from individuals exposed to beryllium via ingestion provide bounds on tumor occurrence.

Both the pharmacokinetic and dose-response data that we examine are from animal studies. Epidemiologic data, although available for inhalation, are not extensively available for ingestion. This lack of information is probably caused by the low tumor risk. The risk is so low that epidemiologic sample sizes have difficulty in both uncovering

the toxicity and separating the relative contributions of inhalation and ingestion exposures.

A major finding that we will discuss is the incomplete and often poor quality of the beryllium data base, both pharmacokinetic and dose-response. As a consequence, the present beryllium water quality standard does not rest on a sound footing, and our ability to make estimates of risk for a given exposure is poor.

Basis and Foundation of Beryllium Water Quality Criterion

Dose-response data relating the tumor response of animals to ingested quantities of beryllium are quite limited. An early study (Barnes, 1948) reported no detectable tumors in mice following administration of 1 percent beryllium sulfate in their drinking water for one year. A later data set appears in Morgareidge et al. (1975), an abstract presenting only summary results on rats and no pathology or reports of numbers of animals used. Two other data sets involving mice and rats appear in Schroeder and Mitchener (1975a and 1975b).

The study by Morgareidge et al. presented the results of exposing Wistar-derived albino rats to beryllium sulfate in the diet at concentrations of 5, 50, and 500 mg/kg over a two-year period. As determined by this group, no significant effects on tumorigenesis were observed. The Cancer Assessment Group at EPA obtained and reviewed the unpublished data from this study and found statistically significant increases in lung reticular cell sarcomas over controls (using Fischer exact values) (EPA, 1980a). However, the trend of response with dose was opposite to that generally observed—the most significant increase in response occurred at the lowest dose. Hence, EPA concluded that the uncertainty introduced by this trend overwhelmed the marginal statistical significances of the lower-dose responses and thus rejected the Morgareidge et al. data from use in standard setting.

In the studies by Schroeder and Mitchener (1975a and 1975b), animals were exposed for life to beryllium sulfate at 5 ppm in their drinking water, 100 (48 males and 52 females) white Swiss CD mice in the former study and 50 (33 males and 17 females) Long-Evans rats in the latter study. Excesses in tumor prevalence over controls were reported in lymphomas/leukemias in female mice (but not in males), in gross tumor occurrence in mice and rats of both sexes, and in malignant multiple-site tumor occurrence in rats of both sexes, but none of these excesses proved to be statistically significant. In spite of this insignificance, EPA (1980a) was able to make use of these observations to derive the beryllium water quality standard. They reasoned that if 5 ppm beryllium were near the observable tumor response level, a real difference between exposed and control animal tumor rates, $P(d) - P(0)$, could exist but only to the degree that statistical tests for

significance just fail. This difference would establish an upper bound on beryllium toxicity at this dose and provide a basis for estimating the maximum possible effect at lower environmental dose levels.

By an analysis of proportions, this difference can be shown to be about 0.21 at the 95 percent confidence level. If one then assumes that the one-hit model,

$$P(d) - P(0) = [1 - P(0)][1 - \exp(-q_1 d)], \quad (\text{B.1})$$

where q_1 = one-hit model dose-parameter (kg-day/mg),

d = dose rate (mg/kg/day),

applies (as did EPA via the multistage model), substitution of the 0.21 maximum possible difference for $P(d) - P(0)$ along with the experimental dose rate d (0.25 mg/kg/day) allows one to estimate an upper-bound value of q_1 for rats, $q_{1r}^* = 1.4$ kg-day/mg. ($P(0)$ can be estimated from the reported control response of 4/26.)

This value can be scaled for use in human risk prediction. The basis for such scaling is the assumption that all species will have the same cancer induction probability when their target tissues have the same concentration of carcinogen. Thus we assume that one-compartment pharmacokinetics applies to animals ingesting water containing beryllium, and, from the balance of ingested dose and clearance rates (Bolten et al., 1983), obtain steady-state estimates of the target tissue concentrations in rats and humans as

$$C_r = CQ_r / k_r \quad (\text{rats}) \quad (\text{B.2a})$$

$$C_h = CQ_h / k_h \quad (\text{humans}) \quad (\text{B.2b})$$

where C = concentration of Be^{++} in ingested water (mg/l),

Q_r = rate of water ingestion for rats (l/day),

Q_h = rate of water ingestion for humans (l/day),

k_r = rat kidney clearance rate (l/day),

k_h = human kidney clearance rate (l/day).

Noting that the dose rate (mg Be^{++} /kg/day) for rats is defined as

$$d = CQ_r / W_r \quad (\text{B.3})$$

where W_r = weight of rat (kg).

This and Eq. (B.2a) can be used to recast Eq. (B.1) as

$$\frac{P(d) - P(0)}{1 - P(0)} = \left[1 - \exp\left(\frac{-q_{1r}^* k_r}{W_r} C_r\right) \right] \quad (\text{rats})$$

Here q_1 has been set equal to the upper-bound value for rats, q_{1r}^* . The scaling assumption above states that $q_{1r}^* k_r / W_r$ also applies to humans provided the human concentration variable is the target tissue level C_h . Thus, for humans

$$\begin{aligned} \frac{P(d) - P(0)}{1 - P(0)} &= 1 - \exp\left(\frac{-q_{1r}^* k_r}{W_r} \frac{C Q_h}{k_h}\right) \\ &= 1 - \exp\left(\frac{-q_{1r}^* k_r}{W_r} \frac{W_h}{k_h} d\right) \end{aligned} \quad (\text{B.4})$$

where d is the human analog of Eq. (B.3) and W_h is the weight of the human. Equation (B.4) shows that the upper-bound human q_1 value in Eq. (B.1) is

$$q_1^* = q_{1r}^* \left(\frac{k_r}{k_h}\right) \left(\frac{W_h}{W_r}\right) \quad (\text{B.5})$$

Since all animal kidney clearances are proportional to $W^{2/3}$ (with the same constant of proportionality), Eq. (B.5) can be simplified to

$$q_1^* = q_{1r}^* \left(\frac{W_h}{W_r}\right)^{1/3} \quad (\text{B.6})$$

from which $q_1^* = 8.84 \text{ kg-d/mg.}^1$

The excess lifetime risk at the 95 percent confidence level, $A(d)$, of a human population exposed to an environmental (low) dose level of Be^{++} is

$$A(d) = \frac{P(d) - P(0)}{1 - P(0)} = q_1^* d \quad (\text{B.7})$$

(EPA, 1980c) where d generally represents the drinking water dose and other ingestion sources of Be^{++} .

Hence q_1^* may be substituted into this expression with $A(d)$ fixed at an acceptable risk of 10^{-5} (EPA, 1980c) to yield an allowable dose. With $q_1^* = 4.84$, the allowable dose was found to be $2.07 \times 10^{-6} \text{ mg/kg/d}$.

¹This value was subsequently changed to 4.84 by the EPA (1981) after they revised the amount of food and water consumed by the rats in the experiment. This revision changed the dose from 0.25 to 0.455 mg/kg/day.

The water quality criterion itself is derived from an EPA formula (EPA, 1980c) in which the allowable dose rate is related to the concentrations in ingested water and food. If one assumes that (1) a 70 kg (W_h) person ingests an average of two liters of water per day (Q_h) with a beryllium concentration C and ingests an average of 0.0065 kg (W_f) of fish per day, where the fish are drawn from the same source as the drinking water, and (2) fish bioconcentrate beryllium nineteenfold ($R = 19$ l/kg), then the allowable dose rate d is related to C by

$$d = \frac{C}{W_h}(RW_f + Q_h) = \frac{A(d)}{q_1^*}$$

where the last equality comes from Eq. (B.7). Numerically,

$$C\left(\frac{\mu\text{g}}{\text{l}}\right) = \frac{3.3 \times 10^{-1}}{q_1^*(\text{mg}/\text{kg}/\text{d})^{-1}}$$

The concentration C from this calculation was defined by EPA as the beryllium water quality criterion and was found to be 0.068 $\mu\text{g}/\text{l}$. Note that the magnitude of this criterion is determined by the amount of beryllium in the drinking water and not by the amount in fish. Fish intake amounts to only 6 percent of the drinking water intake and could easily be ignored in the derivation of the criterion, a bounding calculation at best.

Critique of the Criterion

The EPA water quality criterion has several weaknesses, most of which result from the lack of an extensive data base on ingestion-related beryllium toxicity. These weaknesses include (1) dependence on response data accounting for changes in the occurrence of gross tumors in exposed animals rather than malignant tumors of the type expected from intravenous and inhalation studies; (2) derivation from a bounding calculation on statistically insignificant response data; and (3) dependence on an incomplete pharmacokinetic description of beryllium absorption from the gastrointestinal tract. We have discussed each of these weaknesses in Bolten et al. (1986) and have also suggested approaches to improving the beryllium water quality criterion. Until better dose-response data become available, however, we must use the approach adopted by the EPA.

HEALTH EFFECTS OF SELENIUM

Evidence regarding the acute and chronic toxicities of above normal intake of selenium has been discussed thoroughly in recent studies (NAS, 1976; EPA, 1980a). Much of this information has been summarized by P. F. Morrison in Bolten et al. (1985). The material in this section has been taken from that summary, which considers some of the primary chronic health effects and the basis for our calculation of the risk associated with the increase in selenium exposure caused by plant emissions.

Mammalian Toxicity

Selenosis and Dietary Deficiency. Lengthy reviews on human and mammalian health effects are available in the literature, especially *Selenium* (NAS, 1976) and the *Ambient Water Quality Criteria for Selenium* (EPA, 1980a). Acute and chronic toxicities associated with above normal intake of selenium are discussed in these documents, with particular emphasis on the alkali disease and blind staggers syndromes of hoofed animals² and acute toxicity of the vasculature in man. In addition, these reports also show that toxicity may arise in animals if they receive too little selenium. It is an essential nutrient for rats, sheep, chicks, and quail and very probably for swine, cattle, squirrel monkeys, and man. Depending on the species, selenium deficiency leads to nutritional myopathy (musculoskeletal weakness or myocardial necrosis), hepatic necrosis, gastroesophageal ulceration, yellow fat disease, poor growth, and impaired reproduction. It is suspected, although not yet proven, that selenium normally plays a role in the maintenance of the cardiovascular system (e.g., maintaining vascular membranes) and that failure to meet this role during dietary insufficiency may underlie several of the above-mentioned clinical manifestations. Selenium salts have been used as a nutritional additive since 1957, when they (along with cystine and vitamin E) were found to prevent liver necrosis in certain dietary formulations.

Concentrations of selenium in the diet (water or food) that lead to toxicity because of dietary excess (selenosis) depend strongly on the chemical form of the element. Elemental selenium, for example, is

²Alkali disease is a chronic form of selenium poisoning of livestock characterized by emaciation, lethargy, and separation of the hoof. Newborns exhibit deformed feet, abnormal eyes, myopathies, and atrophy and cirrhosis of internal organs. Blind staggers is a more acute form of poisoning that is characterized by diminished vision and perception of distance and, in its final stages, by paralysis and respiratory failure. It has been suggested that this latter disease may actually be caused by other compounds in selenium-bearing feed plants (NAS, 1976).

virtually insoluble in water and therefore has no significant toxicity. In well-aerated alkaline soils, however, the element may be oxidized to soluble selenites (SeO_3^{-2}) and selenates (SeO_4^{-2}). If later taken up by plants, these forms of the element may become a toxic threat to the animals or humans who eat them (NAS, 1976; Fishbein, 1977). Because plants and animals appear to take up the soluble selenites and selenates over other forms, most laboratory toxicology reports give results for these chemical species.

Some of the most significant selenosis results are shown in Table B.1, where threshold concentrations for various end-points of selenosis are presented. These concentrations depend on the number of animals observed and end-point criteria used, but chronic toxicity generally occurs above 0.5 ppm of selenium in the daily food. This is the EPA lowest-observed-effect level (LOEL) for rats (EPA, 1980a).

Unfortunately, translating from the rat to a standard 70 kg man involves a significant amount of uncertainty. If one assumes that the rat consumes 10 percent of its body weight as wet food and water per day, this concentration is equivalent to a human dose of $50 \mu\text{g Se/kg body wt/day}$ or $3,500 \mu\text{g Se/day}$. On the other hand, if the rat consumes 20 percent of its weight per day, the corresponding human dose becomes $100 \mu\text{g Se/kg body wt/day}$ or $7,000 \mu\text{g Se/day}$. Because of the uncertainty in the calculations, we will use both threshold levels in our calculation of health effects. However, the EPA has chosen the higher threshold ($7,000 \mu\text{g Se/day}$) as its standard level in deriving the current recommended water quality criteria for selenium (EPA, 1980a).

Several of the newborn toxicities associated with selenosis are teratogenicities. As discussed by the National Academy of Sciences (1976) and Rosenfeld and Beath (1964), the chick embryo is extremely sensitive to the effects of selenium and has been used as a leading indicator for the presence of selenosis in other animals on farms. The toxicity of selenium initially shows up as a much reduced rate of egg hatchability. The cause for reduced hatchability is not a reduced rate of fertility but rather deformations of the embryo. Of the eggs that do hatch, many more fail to survive. The abnormalities include the malformation or absence of the legs, toes, wings, beak, and eyes. Chickens fed grain containing 5 ppm of selenium exhibited about a 25 percent reduction in hatchability; others fed 10 ppm exhibited nearly a 100 percent reduction.

Mammals (e.g., sheep) exhibit abnormalities very similar to the chicken, especially the occurrence of grossly deformed eyes, joints, and feet. Some animals exhibit turned feet, suggestive of certain bilateral clubfoot observations in man thought by Robertson (1970) possibly to be associated with a human teratogenic effect of selenium. Teratogenic

Table B.1

THRESHOLD SELENIUM CONCENTRATIONS FOR SELENOSIS

(ppm in feed)^a

Concentration	Species	Toxicity End-Point	Chemical Form	Source
5-40 ^b	Multiple species	Alkali disease ^c	Selenite naturally present in feed, presumably selenite/selenate	(1),(2)
10.0	Rat	Longevity	Selenite/selenate	(3)
4.8 ^d	Rat	Growth rate (over 6 week period)	Selenite	(4)
2.0 ^e	Chicken	Egg production, hatchability, body weight, survival rate; growth of progeny (over 105 week maximum period)	Selenite	(5),(6)
1.25	Rat (severely vitamin E deficient)	Optimal growth rate (as function of selenium dose)	Selenite	(7)
0.75	Rat	Longevity; lesions of heart, kidney, spleen	Selenite ^f	(8)
0.5	Rat	Maximum body weight	Selenite/selenate	(3)

SOURCES: (1) Moxon (1937); (2) Rosenfeld and Beath (1964); (3) Tinsley et al. (1967); (4) Halverson et al. (1966); (5) Tharpar et al. (1969); (6) Arnold et al. (1973); (7) Witting and Horwitt (1964); (8) Harr and Muth (1972)

^aEquivalent to 0.05 mg of selenium per kg body weight per day for a 200 g rat consuming 10 g of feed per day.

^bVarious thresholds reported for various species.

^cSee first footnote of section.

^dAbove this threshold, other toxic effects were noted including increased mortality, increased spleen and pancreas sizes, reduced liver to body weight ratios, and reduced hemoglobin concentration.

^eAt 8 ppm, all toxicity end-points were observed; at 2 ppm, none.

^fPresumed form.

dose-response data for mammals is lacking, although it is apparent that teratogenicities generally occur at higher doses than those of the chick. On the other hand, 10 ppm in the diet of swine will lead to other reproductive effects including a lowered conception rate, lowered number of survivors, and reduced weights at 56 days of age. Hamsters (Holmberg and Ferm, 1969) exhibited no teratogenic effects at near lethal concentrations of sodium selenite nor, unlike rats, pigs, sheep, or cattle, embryotoxic effects.

At the other end of the toxicity spectrum from selenosis is dietary insufficiency. The diseases resulting from deficiency of this micronutrient—especially lowered reproduction rates, myopathy, and vascular disease—have been reported for a wide range of animals. Deficiency disease occurring during the raising of livestock focused attention on the nutrient role of selenium in sheep, swine, and cattle and other laboratory work confirmed the micronutrient role in rats, mice, and monkeys (Muth et al., 1971; NAS, 1971).³ No direct evidence of selenium deficiency in man is available, but its need can be inferred from the animal results and some epidemiological studies of early cardiovascular mortality (Frost et al., 1972). Lowered reproduction rates in animals are attributable primarily to effects induced in the female leading to elevated embryonic losses and loss of fertility. The latter may also be related to decreased sperm motility. The most well-known myopathy is the white muscle disease of sheep and cattle, a degeneration of the muscle fibers. The vascular diseases are associated with altered capillary permeability.

The amount of selenium required to avoid deficiency diseases has been determined as about 0.02 μg Se/g food for rats eating 10 percent of their body weight/day (McCoy and Weswig, 1969; Witting and Horwitt, 1964), 0.02–0.04 μg Se/g food for chicks (Combs and Scott, 1974; Noguchi et al., 1973), slightly less than 0.1 μg Se/g food for sheep (Muth et al., 1971), and 0.03 to 0.04 μg Se/g dry food for trout (Poston et al., 1976). If a value of 0.04 μg Se/g food is assumed to apply to humans, then about 48 μg Se/day is the required dietary intake (this assumes that a 70 kg human consumes 1.2 kg (wet) food per day). The average U.S. resident consumes an amount of selenium only slightly in excess of this, about 132 μg /day (EPA, 1980a). Hence nationwide selenium intakes are much closer to the dietary deficiency limit (48 μg /day) than to the EPA selenosis limit (7000 μg /day). In fact, in the case of low seleniferous regions, dietary deficiencies are probably avoided because foodstuffs are not just grown locally but originate in diverse regions of the nation.

³Other species shown to require dietary selenium are the chicken, turkey, Japanese quail (NAS, 1971), and rainbow trout (Poston et al., 1976).

Dental Caries. The effects of selenium on the incidence of dental caries have been reviewed by both the NAS (1976) and the EPA (1980a). They pointed out that Hadjimarkos and coworkers proposed that selenium could lead to an increased incidence of dental caries. This proposal was based upon observations that urinary selenium in children with an increased incidence of caries was higher than in children exhibiting normal incidence rates. However, other epidemiologic studies have shown either no correlation or a weak correlation between selenium and dental caries (e.g., Muhleman and Konig, 1964; Ludwig and Bibby, 1969). Schwartz (1967) noted that many of the epidemiological studies were not adequately controlled for such variables as economic status, so that it is difficult to reach definitive conclusions about the cariogenesis role of selenium.

Experimental animal studies are mixed. Several found no effect of selenium on the incidence of dental caries. On the other hand, Buttner (1961) reported elevated numbers of dental caries in rats exposed to selenium after weaning or in rats born to dams given selenium during pregnancy and lactation; and Bowen (1972) reported chalky teeth in monkeys fed selenium during tooth development.

The NAS (1976) concluded: "Unfortunately, the work with experimental animals has dealt with selenium added to water or feed at levels that are quite toxic and well above what the general population would be expected to consume. It is dangerous and unrealistic to extrapolate the findings to man. This matter is worthy of additional study, but at present there seems no reason to suspect that selenium is important to cariogenesis in man."

Carcinogenicity and Anticarcinogenicity. In addition to dietary deficiency and the selenosis toxicities of reduced growth, impaired reproduction, and vascular impairment, there has also been some question as to whether selenium has a carcinogenic or anticarcinogenic role. The principal report of carcinogenic activity is Nelson et al. (1943) who found hepatic cell adenomas and carcinomas in rats fed organic selenium in corn and wheat or inorganic selenide in a mixed solution. However, eight subsequent efforts have not reported the induction of neoplasia when rats and mice were exposed to various selenium salts, including sodium selenite and selenate (Cherkes et al., 1962; Harr et al., 1972; Schroeder and Mitchener, 1971 and 1972; Shamberger and Frost, 1969; Tinsley et al., 1967; Volgarev and Tscherkes, 1967). Some of the experiments also involved feeding schedules where selenium was given in combination with a variety of additives whose metabolism is linked to that of selenium. As quoted by EPA (1980a), the discrepancy between the Nelson et al. and later reports may be due to contamination of the feeds with other unknown carcinogens, an extreme

sensitivity of the particular rat strain used by Nelson et al., and the resemblance and mis-scoring of "hobnail" appearing hyperplastic foci as tumors. EPA (1980a), discussing unpublished 1978 data from an NCI-supported Hazleton Laboratory study, reported that female rats, exposed to high doses of selenium disulfide, exhibited significantly increased rates of hepatocellular carcinoma. This is the only recent positive evidence and is not considered sufficiently strong by the FDA, National Academy of Sciences, International Agency for Research on Cancer (IARC), or EPA to conclude that selenium is a carcinogen.

In fact some evidence suggests that selenium has anticarcinogenic characteristics. Demographic studies have described an inverse relationship between cancer mortality and the geographic incidence of selenium or its concentration in cancer patients' serum (Shamberger and Frost, 1969; Shamberger and Willis, 1971; Shamberger et al., 1973). Other studies report that animals exposed to known carcinogens (e.g., nitrosamines and 1,2-dimethylhydrazine) exhibit greater cancer incidence rates than normal when they are selenium deficient (Johnson, 1974) and lesser rates when simultaneously exposed to selenium supplements (Harr et al., 1972).

Because the evidence is so inconclusive at present, we have therefore not considered carcinogenicity as one of the end-points in our case study. Rather embryotoxicity, teratogenicity, growth, and survival (as measured by the selenosis thresholds in Table B.1) seem to be the end-points of primary concern.

Exposure Rates and Toxicity to the Regional Population

The people living in central and southern Pennsylvania are exposed to normal levels of selenium through their consumption of drinking water and imported and locally grown foods. Although some exposure also occurs through inhalation of selenium in the atmosphere, this exposure is negligible relative to the ingestion rates for selenium considered below.

This assertion can be substantiated by considering the inhalation exposure associated with operation of the 500 MWe coal-fired power plant in the case study of Bolten et al. (1985). The highest concentration of airborne selenium reported in municipalities and communities is $0.0097 \mu\text{g}/\text{m}^3$ (EPA, 1980a). Our previous case study assumed that the background concentration of selenium in the atmosphere was $0.005 \mu\text{g}/\text{m}^3$. An adult male inhales $21.9 \text{ m}^3/\text{day}$ (EPA, 1980a), corresponding to a maximum background intake of $0.11 \mu\text{g Se}/\text{day}$. The atmospheric transport results from that study found that the maximum incremental selenium concentrations caused by plant emissions would be about one thousand times smaller than the background levels.

Selenium emitted from the power plant and converted to soluble selenates and unbound selenites will add to background intake as these soluble forms enter the water supplies and are taken up by plants and animals. Although the average selenium intake level may be low, compared to the selenosis threshold, certain subgroups of people, particularly those who eat large quantities of vegetables, may be at a higher risk than the average person. Some of those may be sufficiently near the toxic limit to conceivably be pushed over the limit by additional exposure to power plant selenium. Hence we need to know the distribution of dietary intake of selenium to estimate this increase in toxic response. The selenium dietary distribution can be crudely estimated from the urinary selenium concentration measurements of Smith and Westfall (1937),⁴ assuming that urinary output of the element reflects ingestion rates. These data, if fit to a log normal distribution and rescaled to a mean $\langle \log I \rangle$, where I is the ingestion rate, are characterized by a standard deviation of $\sigma_{\log I} = 0.3238$. Accordingly, the probability, P , that an individual will have a dietary intake exceeding the toxic threshold, $I_t = 3,500$ or $7,000 \mu\text{g/day}$, is

$$P = 1 - \frac{1}{\sqrt{2\pi}} \int_{-\infty}^{(\log I_t - \langle \log I \rangle) / \sigma} e^{-z^2/2} dz$$

where $z = (\log I - \langle \log I \rangle) / \sigma_{\log I}$.

If the average $I = 129.3 \mu\text{g/day}$ is used to estimate $\langle \log I \rangle$, then about 4.9×10 percent of the people might be expected to consume selenium in excess of the $3,500 \mu\text{g/day}$ threshold rate and 3.9×10^{-6} percent in excess of the $7,000 \mu\text{g/day}$ threshold. Note that this average background exposure level found in our analysis is almost equal to the estimated average consumption rate ($132 \mu\text{g/day}$) for the region (EPA, 1980a). This is a consequence of the assumptions about background selenium concentrations in the food and water consumed in the region. At these low selenium levels, the number of people who might exceed the EPA toxicity threshold becomes vanishingly small.

⁴This analysis would benefit significantly from current, or at least more recent, data on the consumption of selenium and the consequent urinary selenium concentrations. Selenium consumption rates have probably fallen in recent years because of increased food processing and wider distribution.

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