ASSESSMENT OF RISK FROM URANIUM MINING IN VIRGINIA

Prepared for

The Coal and Energy Commission
Commonwealth of Virginia

September 1984

SENES CONSULTANTS LIMITED

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SENES Consultants Limited
499 McNicoll Avenue
Willowdale (Toronto), Ontario
Canada M2H 2C9
Tel; (416) 499-5030

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SUMMARY

This report presents an evaluation of the potential radiological risks associated with uranium development in Virginia. Included in the report is a discussion of how radiological risks can be evaluated, a discussion of the results of an evaluation of the proposed Swanson project prepared by the proponents, Marline and Union Carbide (now Umetco), and a discussion of various considerations relevant to setting radiation protection standards for uranium development in Virginia.

A uranium development facility typically includes a mine, a mill, and a tailings (waste) management area. Each of these components can affect the levels of radioactivity in the environment. The possible effects that such releases of radioactivity might have on the public can be assessed by radiation pathways analysis. Such an analysis makes use of models to estimate the exposure that members of the public (in this case persons most likely to receive the highest exposures) could receive by all of the possible pathways of exposure. Examples of potential exposure pathways include drinking water, consuming fish from nearby waters, eating locally grown vegetables and inhaling air, all of which may contain radioactivity released from the facility.

The models used in radiation pathways analysis incorporate basic scientific principles, the experience gained at similar projects, characteristics of the specific site and project being studied, and information about the lifestyles of potentially exposed people. The various models needed to represent the different pathways are often combined into computer codes. Several generic codes are currently available which users can modify to study specific scenarios or facilities.

The pathways analysis prepared by the proponents for the Swanson project utilized two codes: MILDOS and PABLM. The MILDOS code was used to estimate exposures from airborne emissions while the PABLM code was used to evaluate exposures from radionuclides released to the surface waters and subsequently transported via aquatic and/or terrestrial pathways. Exposures were estimated for several locations. The estimates indicated that the air pathways would be the dominant route of exposure at most locations, releases from the mine would contribute a large portion of the total dose, exposure would decrease rapidly with distance from the sources, and exposures would markedly drop upon project close-out.

Table S.1 presents exposure estimates of particular interest together with some comparative values that can be used to put the estimates into perspective. The estimates indicate that under normal operating conditions the most exposed hypothetical off-site receptor would receive an incremental (above background) exposure of about 7.8 mrem per year taking into account all sources and pathways modelled. (The rem is a unit used to express the amount of radiation exposure. A millirem or mrem is one one-thousandth of a rem.) The population dose to all persons living within 50 miles of the project site was also calculated. This calculation indicated that an average member of the population within this 50 mile radius would receive an incremental dose of about 0.04 mrem per year, which is nearly 200 times smaller than that predicted for the most exposed off-site individual.

One way to assess the exposures predicted for the Swanson project is to compare them with current regulations. The two federal agencies largely responsible for setting radiation protection standards are the Environmental Protection Agency (EPA) and the Nuclear Regulatory Commission (NRC). The EPA currently requires that exposures to the public not be more than 25 mrem per year excluding exposures due to background radiation, releases from mines, or any exposure to radon gas and its short-lived daughters. The NRC requires that exposure to any individual in an unrestricted area not exceed 500 mrem per year. This regulation does include exposure from radon and its daughters but, like the EPA standard, does not apply to any releases from mines or background radiation. The Swanson exposure estimates noted above and shown in Table S.1 were based upon releases from a fully developed facility including the mine, mill, and tailings facility.

A second way to assess the predicted incremental exposures is to compare them to existing levels of background (naturally occurring) radiation. Based on measurements taken at ten outdoor locations around the site in 1983, background radiation levels amount to approximately 90 mrem per year of external whole body exposure and approximately 120 mrem per year of radon daughter exposure (calculated on an equivalent risk basis). Additional exposures would result from taking naturally occurring radionuclides into the body through eating and breathing.

A third way to assess incremental radiation exposures is to compare the risks resulting from such exposures to the levels of risk associated with other activities. All activities present some element of risk. For many activities, the levels of risk are so small that no thought is given to avoiding those activities. In this context, a risk of death in the order of about one-in-a-million is in the range of risks commonly considered to be insignificant (de minimis). For instance, travelling 50 miles by car incurs a risk of death of one-in-a-million. The maximum predicted dose of 7.8 mrem per year to a hypothetical off-site receptor is within this range of risks and the risk to an average receptor living within a 50 mile radius of the site is lower still. This approach to assessing radiation exposures is illustrated by the data provided in Table S.1 where the lifetime risk arising from one year of exposure at the levels shown are presented.

TABLE S.1

Summary Comparative Dose and Risk

Receptor/Characteristics	Annual Whole Body Dose	Risk per Million Persons**
NRC limit for general population (excluding background exposure and any release from mines)	500 mrem	50
Exposure to local residents from natural background radiation in vicinity of project prior to mining activity (dose equivalent due to external radiation and inhaled radon daughters)	210 mrem	21
Coles Hill property (on mining site)	16.4 mrem	1.6
Hypothetical off-site receptor with the largest potential exposure* (the location is currently unoccupied)	7.8 mrem	0.78
Hypothetical receptor living at Cedar Hill Hunt Club*	3.5 mrem	0.35
Hypothetical receptor living in Halifax*	0.15 mrem	0.015
Dose to hypothetical average receptor of the population currently living within 50 miles of project*	0.04 mrem	0.004
Average risk of dying from cancer in the U.S.	not applicable	180,000

Note

* Exposure estimates for hypothetical receptor, the Cedar Hunt Club resident and typical Halifax resident include contributions of all radionuclides released from all sources. Federal regulations exclude some sources and radionuclides.

** lifetime risk

$$= \text{annual dose (mrem)} \times \frac{1}{1{,}000}\frac{\text{rem}}{\text{mrem}} \times \frac{1}{10{,}000}\ \frac{\text{lifetime risk}}{\text{rem}}$$

Our review of the pathways analysis prepared by the proponents of the Swanson project suggests that their estimates are likely indicative of expected exposures. Refinement of the analysis can reasonably be left to later stages of project development. The analysis prepared by the proponents also indicates that the maximum predicted exposures are well below federal requirements and represent only a few percent of the natural background levels in the Swanson area. The total annual dose equivalent for all of the people living within 50 miles of the project during the 13 years of operations translates into a lifetime risk of about 0.04 additional fatal cancers. This can be put in perspective by noting that the current incidence of cancer-related mortality in the U.S. (approximately 18%) indicates that over a lifetime more than 140,000 cancer fatalities can be expected to occur in the population living within a 50 mile radius of the site irrespective of whether or not the Swanson project were developed.

Based on this risk assessment the following suggestions should be considered in the establishment of radiation protection standards for uranium mining in Virginia:

- all sources and pathways should be considered in assessing potential exposures
- the prime standard should be a maximum annual whole body dose consistent with a level of risk considered to be acceptable in Virginia
- secondary criteria (such as concentrations in air and water) and procedures for determining compliance need to be developed by State authorities
- efforts should be made to ensure that all doses are kept as far below the maximum dose limit as reasonably achievable, social and economic factors taken into account (ALARA).

1.0 INTRODUCTION

1.1 Study Objectives and Scope

The term risk has been associated with the possibility of many kinds of harm, including among others financial loss, impaired health and loss of life. In this report, risk is most often used to refer to the radiological risks to members of the public associated with uranium mining, milling and tailings management, should these activities take place in Virginia. More specifically risk is discussed in terms of potential increases in the amounts of radiation and radioactivity that uranium development could bring about and the associated radiation-induced cancers.

The risk from any human activity can never be totally eliminated unless the activity itself is either stopped or not undertaken in the first place. Kaplan and Garrick (1981) have expressed this idea symbolically in terms of an equation

$$\text{risk} = \frac{\text{hazard}}{\text{safeguards}}$$

As these authors point out, "This equation also brings out the thought that we may make risk as small as we like by increasing the safeguards but may never, as a matter of principle, bring it to zero. Risk is never zero, but it can be small."

What is needed is an appreciation of the magnitude of the risks. The objective of this report is to provide such an appreciation through an analysis (assessment) of the risks from uranium development in the Virginia context. This is done by first describing the nature of the risks, and then attempting to provide a framework or risk perspective which hopefully will provide the Uranium Task Force with a basis for developing recommendations concerning uranium development in Virginia.

It must be emphasized that what is being described in this report is a risk assessment and not a risk management study. Responsibility for preparation of the latter rests with the government of Virginia or its surrogate. The risk assessment described in this report is only one of several activities being carried out by or for the Uranium Task Force, all of which will be considered in the risk management evaluation.

Finally, it must be noted that this study utilized available information concerning uranium development in Virginia, the most extensive of which are the studies conducted by Marline Uranium Corporation and Union Carbide Corporation (now Umetco) in 1983 and 1984 for the Swanson Project (MUC 1983, MUC 1984). That analysis of the project provided the basis for illustrative calculations and no new environmental modelling was carried out for this study.

Similarly, while certain key issues in the area of health risk estimation are noted, this report does not pretend to attempt a resolution of such issues. Rather they are simply acknowledged as appropriate in the discussion of risks.

1.2 Report Outline

The general nature of uranium mining, milling and tailings management and the ways in which these activities can affect man are discussed in Chapter 2. Also identified are the types of data required for the analysis of potential radiological effects.

Various concepts related to the assessment of radiological risk are discussed in Chapter 3, including the way risk is estimated and various perspectives on the acceptability of risk.

Chapter 4 presents a case study, namely the Swanson Uranium Project. This example is used to illustrate some of the data requirements and calculation procedures used to assess a uranium mine/mill complex. Both individual and cumulative (population) doses are discussed and perspective on the levels of risk associated with the predicted exposures is provided.

Finally, in Chapter 5, various considerations related to the risk assessement process are discussed in terms of how they might affect the setting of standards in Virginia.

2.0 PATHWAYS OF RADIATION EXPOSURE

Mining and milling of uranium ores has been carried out for many years in the United States and elsewhere in the world. However, as reported by the Environmental Protection Agency in its Environmental Standards for Uranium and Thorium Mill Tailings at Licensed Commercial Processing Sites; Final Rule, "all current U.S. uranium mills are located in arid and semi-arid areas, and that we have less experience with many of the control measures needed to comply with the standards under wet than under dry conditions" (EPA 1983b). Although the EPA has indicated in this rule that it feels the standards are adequate for both wet and dry areas, this aspect is of particular interest to Virginia where there is a net water surplus (i.e. annual average precipitation exceeds annual average evaporation and water taken up by plants).

The following brief discussion identifies some of the factors required to assess the surface water pathways, in the event releases to the surface water occur. It is worth noting that while there is no experience with this type of uranium mining/milling facility in the United States, there is considerable experience elsewhere in the world, notably Canada, Australia and France.

Before providing an overview of the factors that should be considered in a pathways analysis (see Section 2.2) some background considerations (generic in nature) of uranium mining, milling and tailings management concepts is provided. The exposures calculated in a pathways analysis can subsequently be related to risks and the risks compared to those of other activities (see Chapter 3.0).

2.1 Background Concepts

2.1.1 Uranium Mining

The type of mining method used at any location is influenced by many factors including the nature and grade of the orebody, its depth, the geological state of the surrounding rock, and the presence or absence of groundwater. Underground and open pit methods are the most common approaches to mining. In situ leaching is also used to recover uranium from certain kinds of deposits.

Environmental considerations that must be addressed are associated with each mining operation. Many of these concerns, such as the storage of waste rock and control of minewater, are common to mines for other elements and minerals.

Potential radiological concerns from uranium mining include:

- discharging minewater that contains radioactive contaminants
- releasing radon and dust in the exhaust air from underground mines
- emitting radon and dust from open pit mines.

In many cases the minewater is used in the mill. If not, contaminated minewater should be treated before being discharged.

2.1.2 Uranium Milling

Mill processing typically involves grinding the ore to a very fine, sand-like consistency. The ground ore is then subjected to a leaching process in either highly acidic or alkaline solutions (depending upon the characteristics of the ore) which cause the uranium to dissolve. This separates the uranium from the solid rock particles. The uranium is concentrated by ion exchange or solvent extraction and precipitated and dried into a product called yellowcake.

Selection of optimal milling techniques require extensive laboratory analysis to determine such matters as leachability of ore, uranium recovery rate, product specifications, the amount of water used, and the degree of recycle.

Because of the nature of the milling operation, the majority of the potential environmental and radiological contaminants end up in the tailings management area.

The amounts of radon and dust released from the milling operation are generally small in comparison to those associated with the tailings management area.

2.1.3 Uranium Tailings Management

Tailings are the waste materials produced during the milling of ore. Tailings consist of ground rock particles, water and various amounts of mill chemicals. Some sort of facility is usually required in which the tailings are managed.

The selection of a tailings management facility and the methods of its operation evolves from the consideration of numerous factors including:

Location

- climate
- local topography and geomorphology
- ore characteristics

Environment

- liquid effluent treatment
- atmospheric emissions control
- seepage control
- public access restrictions

Economics

- slurry versus filtered tailings
- distance from mill to tailings basin
- size
- close-out considerations

At a specific site, local features and environmental conditions determine which of these factors is most important.

The liquid portion in the tailings area is commonly referred to as tailings water. It is predominantly water with relatively high levels (when compared to background) of some dissolved constitutents including radionuclides, and a number of other elements which are present in small amounts in most types of rock. If the operation of the area requires a liquid discharge (as, for example, is the case at all Canadian uranium mines), then treatment facilities are provided. Treatment plants can include chemical addition equipment, a mixing area, and a solids separation facility.

Concern is often expressed over the possible effects of radionuclides released from uranium tailings. One radionuclide, radon, can be emitted as a gas, while other radionuclides can enter the environment in liquid releases or as windborne dust particles. Since uranium tailings remain radioactive for long time periods, the manner in which the tailings management facility would be decommissioned is also of concern.

Table 2.1 presents examples of uranium tailings management practices in use throughout the world. These reflect the effect of different site conditions, economics, and regulations on the development of tailings management practices.

2.2 Pathways Analysis

The possible effects that radioactivity released from a uranium mining area might have on the public is assessed by undertaking a pathways analysis. Such an analysis is used to estimate the exposure or dose that certain critical groups (those most likely to receive the highest exposures) could receive by all of the possible pathways of exposure. Potential exposure pathways include drinking water, consuming fish from nearby waters, eating locally grown vegetables and inhalating air, all of which may contain radioactivity released from the facility. The results of these calculations can be compared with the appropriate standards or other points of reference such as natural variability in background radiation levels.

It is important to recognize that all such calculations are inherently uncertain; for example, the mathematical models (such as those that predict air dispersion of gaseous or suspended particulate matter), are at best only approximations. Moreover, the parameters used in the models are often poorly defined. In view of such uncertainties there is a tendency to incorporate conservative measures which lead to overestimates of exposure in pathways analysis. Care must be taken to avoid introducing so large a bias that the predicted dose indicates an inappropriate measure of risk (harm) from the proposed activity.

Radioactive materials released to the environment may result in exposure to man through various physical, chemical and biological processes. Such processes include the movement or transportation of radionuclides in the environment (usually via air or water), the uptake or bioaccumulation of radionuclides by local plants and animals, and the living conditions or lifestyle of the exposed individual (usually referred to as a receptor). The relative influences of these processes on predicted exposures vary from

Table 2.1

Examples of Tailings Management Practices

Mining Area	Management Practice: Containment Method	Management Practice: Tailings Form	Management Practice: Effluent Treatment
Elliot Lake, Ontario	land/water	slurry	yes
Beaverlodge, Saskatchewan	underwater	slurry	yes
Key Lake, Saskatchewan	semi-dry (layered)	slurry	yes
Southwestern United States	dry	slurry	no
La Bernardan, France	dry	filtered	yes
Nabarlek, Australia	underwater	slurry	no
South Africa	dry	thickened slurry	no

location to location and person to person.

Radiological exposures resulting from the development of a uranium mine/mill facility can be calculated through the use of mathematical models that simulate the behaviour of radionuclides in the environment. Figure 2.1 illustrates several of the potential pathways (routes that lead to exposure) that might be incorporated into a model. Various models have been developed for pathways analysis.

Models can be adapted to suit the situation being studied. Pathways that are not appropriate for a specific situation can be eliminated from an analysis. For instance, if the source term to the surface water is known to <u>not</u> occur (as is often the case in dry areas) the pathway related to this release need not be modelled. Conversely, extra effort can be directed to those pathways known or suspected of being particularly relevant to a specific situation. For example, when it is known that an individual of interest hunts locally for meat, this pathway can be added to the model and information gathered about hunting habits.

Model selection, modification, and application requires information about the type and nature of the source terms, the surrounding environment, and the location and lifestyle of the individuals most likely to be exposed.

Based on their information requirements, the models commonly used in radiological risk assessment for uranium mine/mill developments can be divided into four components: source characteristics, environmental distribution (dispersion and uptake), receptor considerations and dose calculations.

2.2.1 Source Characteristics

Source term characterization typically is based on the following information:

- physical, chemical and radiological features of the ore and host environment
- design of the mine
- design of the mill
- design of tailings management area
- chemical and biochemical reactions occurring in the tailings
- scheduling information, operating procedures
- emission controls
- climate and meteorological conditions
- laboratory feasibility studies
- experience in other operating areas.

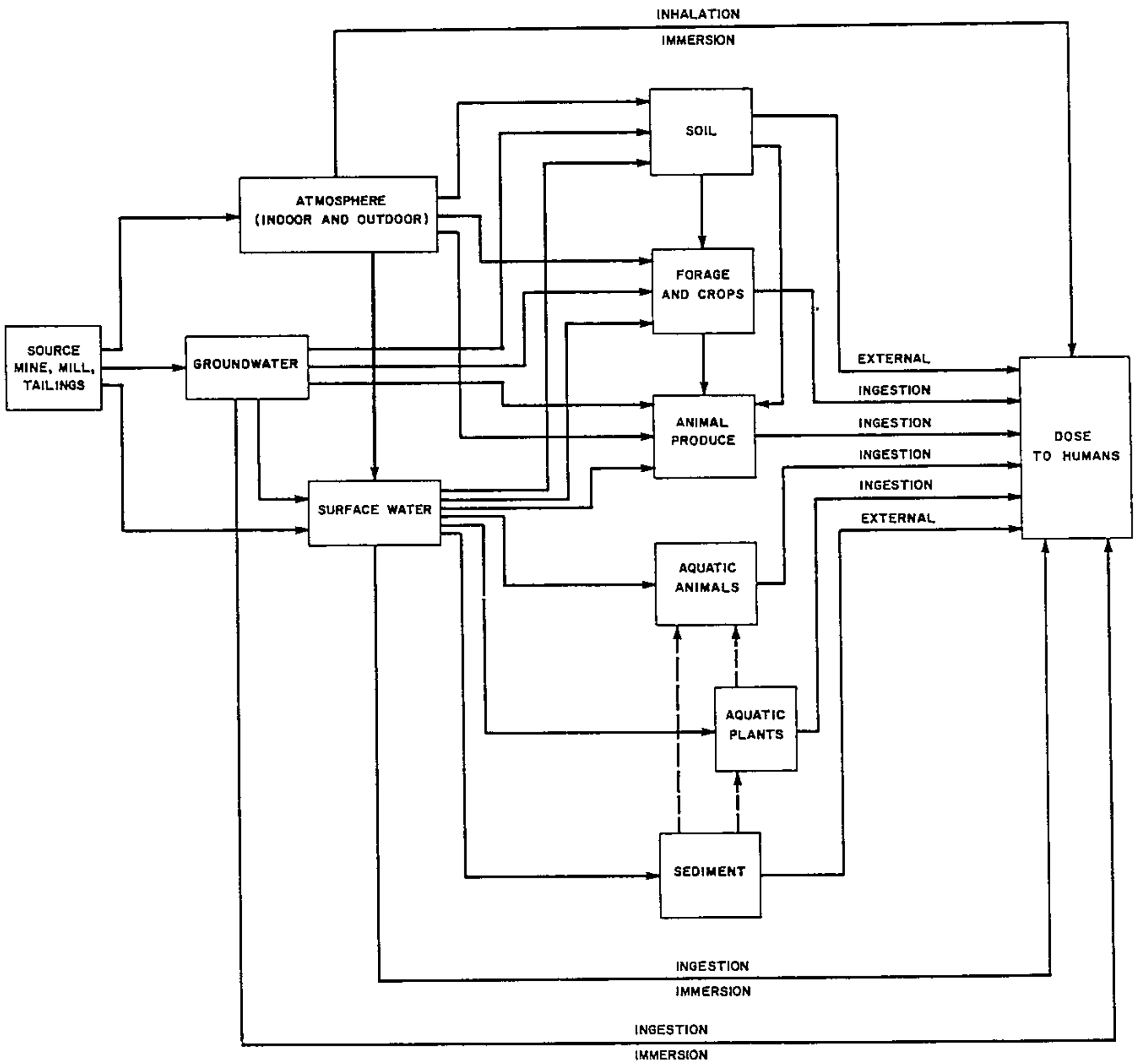
2.2.2 Environmental Distribution

Dispersion involves the calculation of radionuclide concentrations in the air, surface water and ground water at receptor locations. The following information is used to make a model site specific:

- physical, chemical and radiological features of the surface water and ground water environment

FIGURE 2·1

GENERALIZED SOURCE, ENVIRONMENTAL TRANSFER AND DOSE MODEL



NOTE:

1. DOTTED LINES INDICATE PATHWAYS THAT ARE NOT USUALLY EXPICITLY MODELLED, BUT ARE IMPLICITLY INCLUDED IN OTHER MODELLING.

- climate and meteorological conditions
- information on certain aspects of the sources (i.e. stack height, flow rates, terrain)
- experience in other operating areas
- chemical and physical features of the emissions
- the types and locations of potential receptors.

Uptake refers to the incorporation of radionuclides into the terrestrial and aquatic environments at the receptor location. The required types of information include:

- the media in which the uptake occurs
- transfer and uptake factors from site specific or other studies (i.e. literature values)
- local agricultural practices.

2.2.3 Receptor Considerations

The dose (and hence the risks) associated with a facility are usually the principle consideration in assessing whether or not that facility presents a suitably low hazard (risk). Moreover, by controlling the dose to the critical receptor (the individual(s) most likely to receive the largest exposure), it can normally be demonstrated that people living further away will be subjected to a much smaller risk.

Depending on site conditions and individual characteristics, the presence of a particular radionuclide in a critical pathway will result in varying doses to different members of the public. For dose estimation purposes, one or more groups of people are identified that are likely, on average, to receive higher doses than other groups of the public. The identification of such groups, called critical groups, requires consideration of lifestyle characteristics as well as releases to the environment, and local environmental conditions. Lifestyle characteristics include:

- recreational habits (e.g. time spent hunting near the site, swimming in nearby waters, etc.)
- diet, including all foods locally raised (gardens, livestock, etc) or caught (fish, deer, wild fowl,etc.) and sources of drinking water or water for irrigation
- fraction of time spent at home and time spent indoors versus outdoors
- shielding factors (e.g. building construction, methods and materials)
- population density and distribution (to estimate cumulative dose).

2.2.4 Dose Calculations

The final modelling component is a procedure to calculate external exposures (due to direct exposure from the site or levels of radionuclides in the air, water and soil) and internal exposure (due to intakes through inhalation and ingestion). This usually incorporates dose conversion factors (both external and internal) for age groups as appropriate for various body organs. Many of these considerations are illustrated in the Swanson Case Study discussed in Chapter 4.

3.0 INCORPORATING RISK INTO THE EVALUATION OF RADIATION EXPOSURE

Risk is defined in different ways in different disciplines. In this report, as noted in Chapter 1, risk refers to the incremental radiological risk to members of the public associated with the release of radioactivity to the environment through uranium mining, milling and tailings management practices. In particular, the focus of the discussion in this report is the incremental risk of mortality (from cancer) to persons so exposed. In this chapter we examine, on a generic basis, the various factors which were considered relevant to this risk assessment study.

3.1 General Characteristics of Risk

All activities entail some risk; that is, there exists a chance of experiencing some form of hazard, harm, or loss associated with participating in any type of activity. Studies of the incidence of mishap or detriment have quantified the risks of some activities.

Most activities present levels of risk that are sufficiently small that little or no thought is given to the risk involved. Other activities are known or perceived to be relatively hazardous and there may be a tendency to avoid such activities. Attitudes toward risk and the willingness to accept risk are highly variable. Factors that influence attitudes include the attractiveness of the benefit to be gained, whether the activity undertaken is voluntary or involuntary, and whether the decision to participate is made by an individual or by a group.

Many studies have investigated the statistical aspects of voluntary risks, such as those resulting from occupations (assuming that the person has a choice of occupation) and recreational activities. For example, the estimated annual risk of death from being a commercial fisherman is three-in-a-thousand, while skiing for 100 hours presents a risk of seven-in-one-hundred thousand (Webb and McLean 1977).

Involuntary risks are those associated with activities that are determined not by an individual but by "guardians of public health" such as the agencies charged with setting construction codes, designing highways, or providing public water supplies. People will generally tolerate greater voluntary risks than those imposed upon them by others (Oser 1978).

Risks are often expressed in terms of probability of occurrence. Often there is an element of time involved. For activities that occur over relatively short periods of time, the risk is usually stated as a simple probability. For example, there is about a one-in-a-million probability of death from driving for 1 hour in a car.

For activities that occur regularly or constantly over longer periods of time, the risk may be expressed in annual terms. For example, Crouch and Wilson (1982 Table 7-2) indicate that in the United States the average annual per capita risk of death from a motor vehicle accident is in the order of 240 deaths per million people per year. If this risk to an average individual were applied to the U.S. population of about 200 million, it represents a cumulative annual mortality of about 48,000 persons per year.

Many of the risks in this discussion are presented as annual risks. Over still longer periods of time, risks may be expressed in terms of the probability (of an event) occurring over a lifetime.

Risks from exposure to hazards can also be expressed in terms of days of life expectancy lost. Such risks consider the age at which exposure occurs and the expected length of a life. Any activity that presents an annual risk of one-in-a-million over a lifetime of 70 years, would reduce the average lifetime by approximately one day. Similarly, a one-in-a-thousand annual risk experienced from birth would reduce the average lifetime by approximately 1000 days.

In matters pertaining to health, estimates of risk can be concerned with the occurrence of death or can be related to the occurrence of a specific disease or group of similar diseases. The following discussion largely uses mortality statistics to illustrate various aspects of risk.

3.2 Relating Risk to Radiation Exposure

An enormous amount of research into the effects of exposure to ionizing radiation has been carried out, particularly over the past forty years. Various national and international commissions who publish reports and recommendations routinely examine the scientific literature in this area. These bodies include: the National Council on Radiation Protection and Measurement (NCRP); the International Commission on Radiological Protection (ICRP); the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR),; and the National Academy of Sciences (NAS) Committee on the Biological Effects of Ionizing Radiation (BEIR) (eg. NCRP 78 1984b, ICRP 26 1977, UNSCEAR 1977 and 1982, and NAS BEIR 1980).

While there is no conclusive evidence that low doses of radiation result in a health detriment (in this case an increased incidence of cancer), it is commonly assumed that there is some risk associated with any dose no matter how small (ICRP 1977). While the precise nature of the dose-effect relationship is not known, for radiation protection purposes the effects at low levels of exposure (those of interest to this study) are assumed to be directly proportional to the dose received. This linear, no threshold model provides a basis for estimating risks of exposure to ionizing radiation but needs to be used with caution, as discussed below.

Based on evidence from many biological systems and theoretical calculations, most radiobiologists believe that the linear, no threshold model leads to an overestimation of the health effects of small doses, especially for low linear-energy-transfer radiation such as gamma rays or low energy beta particles. In the case of high linear-energy-transfer radiation, such as alpha particles from the decay of the short-lived radon daughters, the dose-response relation is more likely to be linear. The nature of the dose-response relationship is controversial and widely discussed in the literature. However, regardless of the model selected, the effects are quite small at low doses. As a result, it seem unlikely that the "true" shape of the dose-response curve at low doses will ever be known (eg. NAS BEIR 1980).

It is worth noting that "man and lower forms of life have developed in the presence of such natural sources (of radioactivity) in spite of any radiation damage that may have been present" (FRC 1960 para. 4.10). Indeed, there is even speculation by some scientists that low levels of radiation may have a beneficial effect (the concept of hormesis, Luckey 1980).

The risks of low-dose, low-dose-rate exposures are often expressed as the lifetime chance that a detrimental effect, such as cancer, will be caused by an increment of dose. The fact that this is a calculation of a probability must be emphasized, since the potential effects occur only randomly among an exposed population.

How large is the risk of exposure to low levels of radiation? The ICRP concludes that the mortality risk factor for whole body radiation is about 1 in 10,000 per rem (ICRP 26 1977). This means that if a person is exposed to 1 rem of radiation (above natural background), his chances of dying from cancer are increased by 1 in 10,000. Or, if a million people are each exposed to 1 rem of radiation, about 100 excess deaths from cancer would be expected. (where excess is relative to the normal number of cancer-related deaths.) In a population of one million, approximately 180,000 persons will die of cancer (American Cancer Society 1984). Thus the 100 excess cancer deaths due to the irradiation of 1 million people with 1 rem each is not likely to be detected due to normal variations in cancer deaths in the general population. An individual would normally be exposed to about 7 rems of radiation over his lifetime from natural sources of radiation (excluding radon). This may vary by a factor in the range of 2-5 depending upon where one lives, and one's lifestyle and occupation.

The risk of lung cancer from exposure to radon daughters is perhaps more controversial. The estimation of risk from exposure to radon daughters is largely based on epidemiological studies of persons exposed to very high concentrations of radon daughters in a workplace setting, principally underground uranium mines. These studies are confounded by a variety of factors including the concurrent exposure of miners to other workplace contaminants which may themselves be carcinogenic or alter the miner's response to exposure to radon daughters (measured in working level months, WLM), high or unknown cigarette smoking patterns, and uncertain estimates of the miner's cumulative exposure.

A recent NCRP report estimated a lifetime risk of between 1 to 2 cases per 10,000 persons exposed to 1 working level month (i.e. (1-2) x 10^{-4} per WLM) depending on age and duration of exposure (NCRP 78 1984b). Evans et al (1981) examined the risk from environmental exposure to radon and its daughters and estimated an upper limit of about 1 x 10^{-4} per WLM for members of the general public.

For the purposes of this report, a lifetime risk of about 1 x 10^{-4} per WLM is assumed. On this basis, the lifetime risk of mortality from cancer from 1 rem of whole body radiation is comparable to the lifetime risk of mortality from lung cancer from 1 WLM of radon daughter exposure.

Radon daughter exposures are highly variable. In the U.S. the average exposure is about 0.2 WLM per year. Largely because of local soil characteristics and/or local building materials, about 0.14% of Americans (nearly 300,000 people) receive exposures in excess of 4 WLM per year (NCRP 77 1984a), the federal occupational standard for exposure to radon daughters.

On the basis of a life expectancy in the order of 75 years and assuming uniform exposure and uniform risk (convenient simplifications) an estimate of the lifetime risk from exposure to 0.2 WLM per year is about 1.5×10^{-3} (75 years x 0.2 WLM per year x 10^{-4} per WLM) or about 0.15%. This is approximately 2.5% of the normal lung cancer fatality rate for all Americans and about one-fifth of the lung cancer fatality rate for non-smoking Americans.

3.3 Summation of Risks from Exposure to Radiation

As discussed in Chapter 2, there are a number of pathways through which people may be exposed to radiation in the environment. These include the inhalation of radon daughters in the atmosphere, gamma radiation from radionuclides in soil and rocks, and the ingestion of water and food containing radionuclides. Each pathway produces varied radiation doses to different organs of the body. These doses are not strictly additive but one might wish to consider some form of summation to evaluate the total effect of mixed exposures and to provide a basis for comparing risks from radiation exposure to other risks.

In practice, permissible limits of risk or exposure to radiation can be met by expressing each exposure as a fraction of a permissible limit and requiring that the sum of these fractions not exceed unity (the so-called "summation rule"). For example, the recommended ICRP dose limit for occupational exposures is 5 rem/year and the limit for exposure to radon daughters is 4.8 WLM/year. According to the ICRP, both of these limits represent equivalent risks. That is, the total combined risk of external exposure (say X rem/year effective whole body dose), internal exposure (say Y rem/year effective whole body dose) and radon daughter exposure (say Z WLM/year) will not exceed permissible limits provided that:

$$\frac{X + Y}{5} + \frac{Z}{4.8} \leq 1$$

Current 10 CFR 20 regulations (NRC 1977) are based on the recommendations of ICRP Publication 2 published in 1959. ICRP 2 used the concept of "critical organ" or tissue where the critical organ is defined as the body organ receiving the largest committed dose from an intake of a radionuclide. ICRP 2 recommended that the intake of a radionuclide in a given year be controlled so as to limit the committed dose to a maximum permissible dose for that organ. Current EPA regulations also use the concept of critical organ. The ICRP now bases its recommended limits on the total effective whole body dose (ICRP 1977), a practice convenient for risk assessment purposes.

A possible summation method has been proposed by the ICRP in its Publication 26 (1977). By using weighting factors set in proportion to the health risk or sensitivity to exposure of the individual organs, the ICRP converts doses to individual organs to effective whole body doses which are then additive because they represent equal risks (of health effects). For example, the ICRP weighting factor for the lung is 0.12. This means that a dose of 100 mrem (1 rem = 1000 mrem) to the lung resulting from the inhalation of radioactive materials is equivalent in risk, to a dose of 100 mrem x 0.12 = 12 mrem received by the entire body. If a person so exposed also receives a dose of say 50 mrem to his entire body as a result of some source of external radiation, his total effective whole body dose would be:

50 mrem + 12 mrem = 62 mrem

In this manner, the doses to various organs through various pathways can be added on the common basis of risk and the total resultant dose (risk) can be examined to see if it is below permissible (acceptable) limits.

For purposes of illustration, the existing level of risk from natural radiation exposure in the vicinity of the Swanson project can be estimated from the data presented in the 1983 Swanson submission. Measurements of direct gamma radiation levels were reported for several locations. Ten of these locations were at outdoor air quality monitoring stations. The average value of those measurements was reported to be 1.69 mrem/week or approximately 90 mrem/yr. At the same ten locations, measurements of radon concentrations produced a geometric mean value of 0.48 pCi/L, which presents a risk roughly equivalent to a whole body gamma radiation exposure of 120 mrem (see Table 5.2b for conversion between pCi/L of radon and dose). The gamma and radon daughters exposures represent annual risks of about 9×10^{-6} and 12×10^{-6}, respectively, and a total annual risk of about 21×10^{-6}.

3.4 When Risks Become Insignificant (De Minimis)

The concept of acceptability is a key component in the study of risk. Being a personal and subjective matter, acceptability is not directly measureable nor is it easily quantified. However by evaluating activities that present progressively smaller levels of risk, a level can be reached that most people will deem acceptable. At even lower levels, risks become perceived as being so small as to no longer influence an individual's behaviour. Such levels of risk can be described as insignificant, trivial or negligible.

For health matters, this level is generally thought to be near the risk level of one-in-a-million per year of death (Webb and McLean 1977). Based on the work of several researchers (Oser 1978, Pochin 1978, Wilson 1979, Starr and Whipple 1980, Crouch and Wilson 1982), Table 3.1 lists activities that present annual risks of death of one-in-a-million.

The smallness of the one-in-a-million level of risk can be illustrated by comparing it with the risks of death from other causes. For example, the most risk-free U.S. males are those in the group between 5 and 9 years old. To an individual in that group, the annual risk of death from all causes is approximately 334 chances per million (U.S. Department of Health and Human Services 1979). For individuals in the 55 to 59 age bracket, the annual risk of death is approximately 14,900 per million. The annual risk of death averaged over all age groups for the U.S. male population is approximately

Table 3.1

Selected Activities with a Risk of Death of One in a Million

Activity	Cause of Death
Travelling 50 miles by car	Accident
A pedestrian being hit by a motor vehicle during a nine-day period	Accident
Travelling 10 miles by bicycle	Accident
Living 2 months in an average stone or brick house	Cancer from the radioactivity of the building materials
Home accidents during a three-day period	Falls, electrocution, etc.
Being struck by lightning during a two-year period	Electrocution
Dying from air pollution during a two-day period	Various causes
Dying in a flood, or tornado during a two-year period	Various causes
Living 2 months with a cigarette smoker	Cancer, heart disease
Smoking 1-2 cigarettes	Cancer, heart disease
Drinking 0.6 ounces of beer per day for a year	Cancer (alcohol-related)
Drinking 2 ounces of milk per day for a year	Cancer (aflatoxin-related)
Living 20 minutes at the age of 60	All causes

8,800 per million.

In discussions of radiation protection, the upper level of exposure at which an individual's health and welfare is not significantly changed by its presence or absence is commonly referred to as the de minimis level. The term de minimis can be traced back to the Latin phrase de minimis non curat lex, the law does not concern itself with trifles. As noted by Dunster (1982), the de minimis concept was invented to avoid society's squandering its efforts on seeking legal solutions to problems of no significance. Long recognized in law, the concept is incorporated into many administrative and regulatory practices in the form of "cut-off" levels, below which authorities are not concerned.

Alternative De Minimis Proposals

The International Commission on Radiological Protection (ICRP) and other recognized scientific organizations have determined that the lifetime risk of deleterious health effects (such as cancer induction) from a radiation dose of one rem to the whole body is approximately one-in-ten-thousand. (For comparison, the lifetime risk from one year of exposure to 500 mrem, the current federal limit for a member of the public, would be 50 per million.) From that estimate, the linear hypothesis indicates that a one-in-a-million level of lifetime risk corresponds to the risk of an exposure of 0.01 rem (or 10 mrem). Annual exposure at 10 mrem has been proposed as a de minimis level for exposures to individuals (Webb and McLean 1977).

Chatterjee et al (1982) suggest that to allow for uncertainties in the number of radiation sources to which an individual may be exposed and to ensure that doses corresponding to this magnitude of risk will indeed be regarded by most as trivial, an annual risk of one-in-ten-million might be an appropriate de minimis level. This risk translates into a dose to an individual of 1 mrem/yr.

Announcements from the U.S. NRC indicate the favouring of a de minimis level based on lifetime risk rather than annual risk. Giving consideration to the many factors that influence exposure such as environmental dispersion and bio-availability, a de minimis value of 1 mrem/yr for individuals has tentatively been suggested by the NRC staff (Cunningham 1982).

Other researchers have suggested that a de minimis level should be derived from background radiation information. One approach proposes that the variation in dose that is experienced as a result of geographical differences in natural background levels should be considered to be de minimis (Eisenbud 1980). This method suggests a de minimis level in the range of 20 to 100 mrem/yr.

A second approach using background radiation levels would make use of the standard deviation of the background exposure rate, weighted with the exposed population (Adler and Weinberg 1978). Analysis of background radiation exposure data for the United States indicates that the standard deviation of external radiation exposure is approximately 20 mrem/yr.

A third approach to deriving the de minimis level from background radiation data is that the de minimis value should be a small percentage (say 30%) of natural background (Rossi 1980). Since background varies with location, the de minimis value would depend on the area being studied. The calculated world average for external background radiation of 80 to 90 mrem/yr (NIH 1980) would thus indicate a de minimis level of about 25 to 30 mrem/yr.

Figure 3.1 presents several de minimis values that have been proposed together with some other low levels of radiation for comparative purposes.

A trivial level of 0.1 mrem/yr for collective doses to large populations has been suggested by staff of the Atomic Energy Control Board of Canada. (Board staff have arbitrarily defined large populations as those of more than 100,000 persons). Use of such a level limits the size of the population and the time over which collective doses need to be considered. This eliminates the mathematical integrations of exposure over vast expanses of space and time that some researchers have used to indicate large numbers of potential effects, but which actually represent upper limits and reveal very little about the actual situations, other than that the actual effects lie somewhere between zero and the numbers calculated. Other inconsistencies in broad-scale integrations are evident when it is considered that the collective dose is calculated to double when the population doubles. Yet the individual dose and risk remain the same, while presumably all of the benefits and resources available to society are apt to double as well (Davis 1981).

3.5 The ALARA Principle

The ICRP provides guidance to many countries on matters of radiation protection, the safe use of radiation, and the development of radiological protection regulations. The system of dose limitation recommended by the ICRP is enunciated in Publication 26 (ICRP 1977) and based upon the following three principles:

"(a) no practice shall be adopted unless its introduction produces a positive net benefit;

(b) all exposures shall be kept as low as reasonably achievable, economic and social factors being taken into account; and

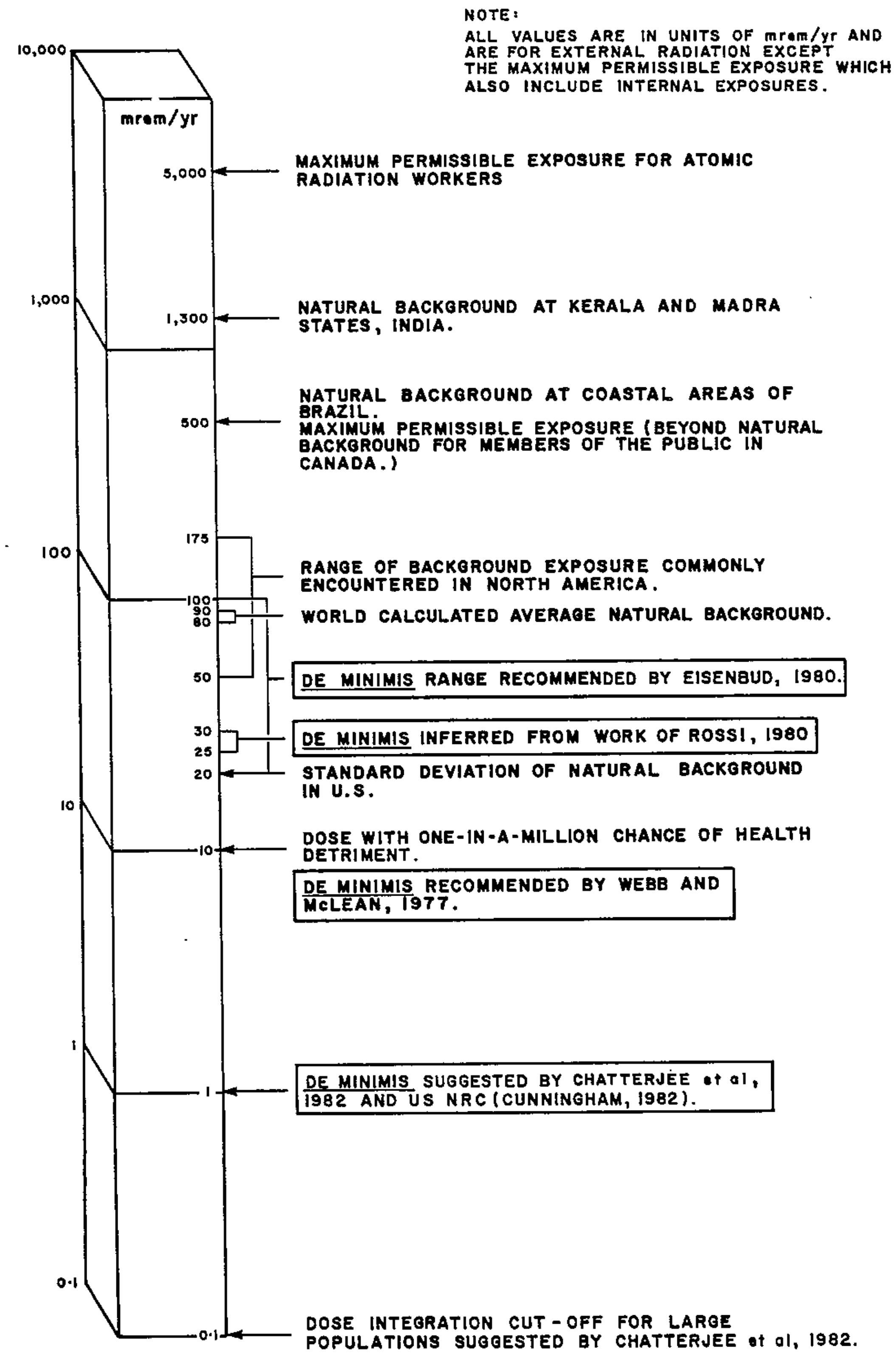
(c) the dose equivalent to individuals shall not exceed the limits recommended for the appropriate circumstances by the Commission."

The first principle is commonly referred to as the "justification" principle. The second principle is often cited as the "optimization" principle and is commonly referred to as "ALARA" (an acronym for as low as reasonably achievable, social and economic factors taken into account). The third principle calls for individual dose limits which the Commission has articulated in Publication 26 and subsequently expanded upon in Publication 30 (ICRP 1979) and Publication 32 (ICRP 1981).

The common interpretation of ALARA is that there is an appropriate degree of dose reduction, below the recommended individual dose limits, which should be determined by some form of cost-benefit (or risk-benefit) analysis.

FIGURE 3·1

SUMMARY OF SUGGESTED DE MINIMIS LEVELS AND SELECTED LOW LEVEL RADIATION EXPOSURES



Theoretically, the costs required to achieve incremental dose reductions are weighed against the benefits that may be obtained. The benefits are primarily in the form of reduced risks of adverse health effects in the exposed population. The theoretical optimum is achieved when "... the increase in the cost of protection per unit dose equivalent balances the reduction in detriment per unit dose equivalent" (ICRP 1977).

The ALARA principle has become a major objective of the practical application of radiological protection programs in many countries including the United States (at least in the sense of cost-benefit analysis). The ALARA concept recognizes the generally accepted assumption that there may be no dose, no matter how small, that does not entail some increased risk. Since a totally risk-free threshold dose cannot be identified, the optimization process represents a technique for determining an appropriate control level below the maximum individual dose limits, as shown schematically in Figure 3.2.

3.6 Risk-Benefit Analysis

At the heart of risk-benefit (or cost-benefit) analysis is the comparison of the risks (or costs) and benefits of a particular project, program, or event to determine if a net benefit will ensue. Indication of a net benefit can be used to support a decision to proceed while a net cost suggests that an alternative approach should be considered.

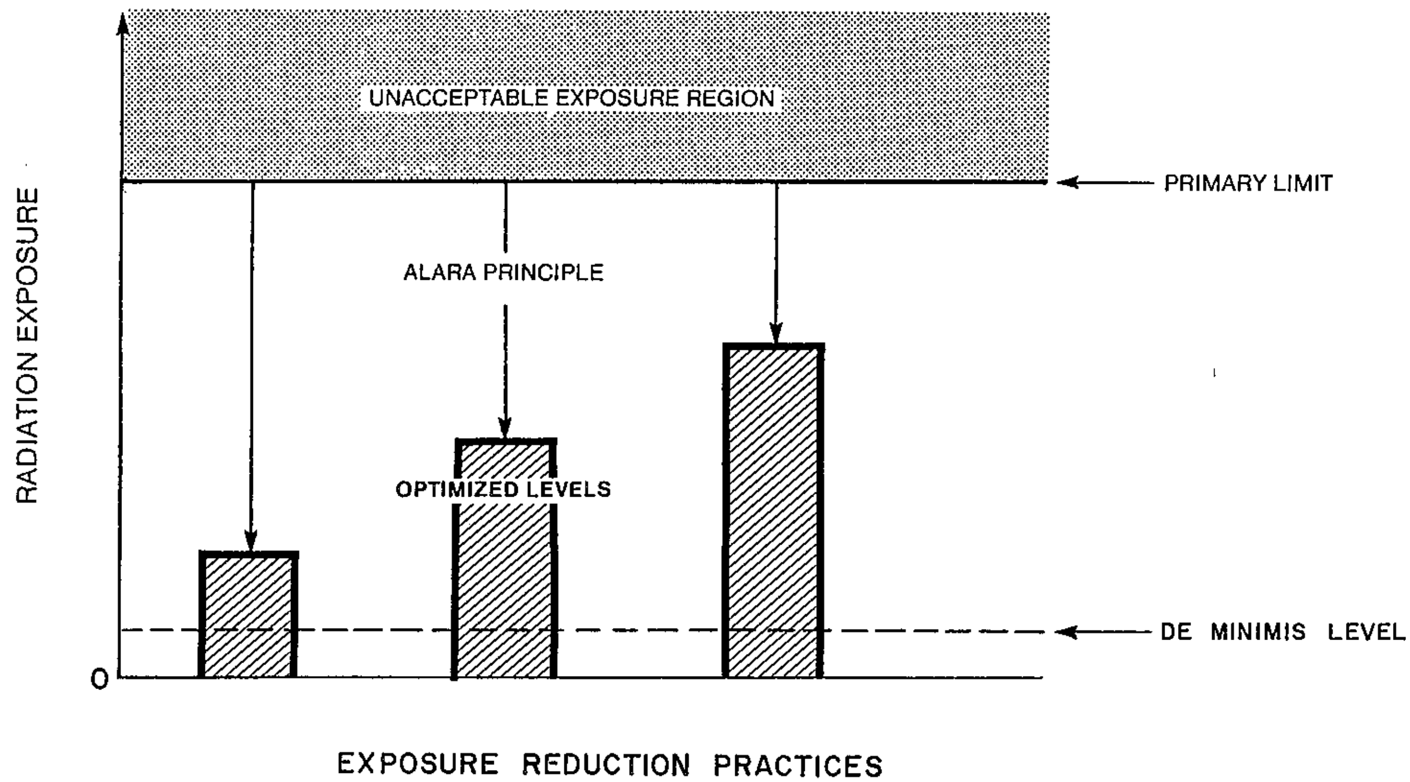
Concepts such as ALARA are directed not just to determining whether a net benefit or net cost will occur, but to extending beneficial actions until the point is reached where the next incremental increase of benefit is balanced by an equal increase in cost. For example, a section of a hypothetical highway is particularly treacherous and the scene of frequent accidents. The posting of an appropriate warning can be achieved at very little cost and will produce a net benefit even if only one accident is prevented. Conversely, the construction of an alternative section of highway could greatly reduce the accident rate but the costs of construction might be so large as to far exceed the costs of many years of accidents. Construction will also require funds that otherwise could be used for other community services that provide more benefit per dollar spent. Obviously there is an optimum solution somewhere between these two extremes, and it is such an optimum that ALARA and similar concepts are intended to identify.

Although not blatantly obvious, individuals use a modified, unstructured form of risk-benefit analysis regularly. Deciding to drive faster than the speed limit to reach a destination sooner, enjoying a cigarette or foregoing a check-up at the dentist all imply that some sort of risk-benefit analysis has been made. Because the personal resources of each individual are finite, decisions are constantly required of each of us to optimize their use and maximize our "quality of life".

On a broader scale (such as the preceding highway example), groups of individuals (usually as government bodies) also use cost-benefit analysis to help decide courses of actions and resource allotment. In recent years, cost-benefit analysis has been increasingly advocated as a method of assisting decision-makers. While the philosophy of optimization is appealing, ALARA and

FIGURE 3·2

DIFFERENCE BETWEEN BASIC LIMITS AND OPTIMIZED EXPOSURE LEVELS (AFTER JACOBI, 1981)



similar concepts have proven to be difficult to apply in practice. These difficulties stem from weaknesses encountered in accurately quantifying the costs and benefits to both individuals and groups.

ALARA requires an assessment of the costs of radiological protection and radiation-induced health detriment so that the sum of these costs may be minimized (Clark et al 1981). The level of sophistication and corresponding degree of effort given to this assessment should reflect the scale of the problem being considered.

The estimation of radiation protection costs is in principle straight-forward, although the assessment may in practice require professional judgement, especially when detailed costs of equipment, materials, energy and labour have to be considered. In general terms, the costs of radiological protection will typically involve an initial capital investment, followed by operating and maintenance costs over the lifetime of a facility or equipment. Methods commonly used for this purpose are present-worth evaluations and annualization techniques.

The assignment of costs to health detriment involves several types of judgements. Risk factors can be used to predict the statistically expected number of health effects which may occur in exposed populations. The problems of costing the impact on society of such statistical health effects are many and are not unique to radiological protection. Such valuations are needed, and made, in all decisions involving health and safety, although the valuations may not be explicitly obvious to either the decision-maker or those affected. For example, certain chemicals are used to minimize food spoilage since some forms of spoilage can pose serious health problems (for example, botulism). Conversely, many of the chemicals used to minimize spoilage are known or suspected of being hazardous if ingested in sufficient quantities. The decision must then be made as to whether or not it is preferable to risk the incidence of health problems related to food spoilage or assume the possiblility of other problems that might result from intake of certain chemicals.

Nonetheless, to carry out an ALARA optimization analysis, some value for the health detriment or cost per unit of exposure has to be adopted. To date, these values have usually been expressed in units of dollars per man-rem where one man-rem occurs when the exposures of a population total one rem (i.e. dose per person x number of people). For example, if 20 individuals have an average exposure of 50 mrem, or if 100 invididuals are each exposed to 10 mrem, the total exposure is one man-rem.

Dollar Value of a Man-Rem

Different approaches reported in the literature suggest dollar values that extend over several orders of magnitude. The ICRP (1973) has noted estimates that range from $10 to $250 per man-rem (in 1966 to 1972 dollars) with a median of about $50 per man-rem. A literature survey by the International Atomic Energy Agency showed a range from $10 to $1000 per man-rem (Ahmed and Daw 1980). The U.S. NRC has used $1000 per man-rem as a guideline in selecting waste disposal mechanisms for high level reactor waste (NRC 1976);

however, that figure arose in the context of a reactor rule-making proceeding and involved vastly different considerations than those encountered within the context of uranium mines and mills (NRC 1980).

A study of the societal costs of radiation exposure in the U.S. suggested that a value of $3000 per man-sievert ($30/man-rem) is an appropriate value to use in determining whether dose reduction actions are reasonably achievable (Voilleque and Pavlick, 1982).

A recent socio-economic impact analysis for the regulation of the chemcial compound chlorobiphenyl in Canada provided a range of estimates for the economic value of a life (Canadian EPS 1981). The authors defined the economic value of life as the worth that administrators place on life when making decisions to allocate resources to increase longevity. The values (in 1982 Canadian dollars) cited in that report ranged from $57,000 to $4,741,000. The National Radiological Protection Board (U.K.) has suggested a unit cost of about 20 pounds per man-rem ($40 to $50 per man-rem in 1980 U.S. prices) for optimization purposes of $400,000 to $500,000 per fatality (Clark et al 1981).

The EPA has discussed cost-benefit analysis in assessing the regulatory impact of standards for uranium mill tailings (EPA 1983a). The EPA has estimated, from studies of market compensation for small risks, that people would be willing to pay from 0.3 to 2.5 million dollars to save a life (EPA 1983a). The upper limit of 2.5 million dollars to save a life was used to suggest that control methods costing more than this value would not be justified on a cost-benefit basis (EPA 1983a).

Using a risk factor of one-in-ten-thousand per man-rem, a value of $100 per man-rem corresponds to a cost of $1,000,000 per statistical life saved (i.e. for a risk of one-in-ten-thousand per rem, ten thousand persons need to be exposed to one rem to produce one fatal cancer; 10,000 man-rem x $100 per man-rem = $1,000,000). To gain perspective on the costs of radiation protection, Siddall (1981) has catalogued the cost per statistical life saved for a large number of activities. For example, spending $1,000,000 on vaccinating children in underdeveloped countries would save 40,000 lives at a cost of only $25 per life. Installing smoke detectors in all homes in the United States could save 8,000 lives at a cost of $80,000 per life saved. Relative to these figures, the $100 per man-rem is considered a justifiably conservative (i.e. high) value.

3.7 The Influence of Time Upon Risk/Benefit Analysis

Society is becoming increasingly concerned with the risks of radiation exposures arising from situations that are not amenable to quantitative optimization. These situations generally exhibit various combinations of the difficulties previously described in this section. Some sources of radiation exposure may not be measurable directly and must be estimated by modelling techniques. These sources may be natural radioactivity with exposures enhanced by technological activities (such as mining) but which are indistinguishable in character from the naturally occurring background radiation.

A difficulty stems from the variation of radiation pathways with time. Environmental pathways through which people are exposed to radiation are known only for current conditions and are usually evaluated as steady-state conditions. Although pathways through the geosphere may be realistically assumed to change very slowly, the same cannot be assumed for pathways through the biosphere. Dramatic changes in agricultural practices, food processing and distribution, community water supplies and treatment methods, and management of other natural resources have taken place within the last century, and even more radical changes are possible during the next century.

Health effects from radiation exposure cannot be confidently predicted for conditions in the future. Current knowledge of radiation health impacts has been obtained, to a large extent, from humans exposed to radiation in combination with a wide range of other deleterious factors found in the environment. Whether or not the risks would be the same under some different environmental conditions in the future is a matter of conjecture. Risks from radiation exposure are also subject to future modifications by direct human intervention. The risk of cancer induction and death are likely to be tempered by improvements in medical diagnosis and treatment.

A discussion of the various approaches which have been proposed for selecting an appropriate time interval for performing dose integrations for risk- or cost-benefit analyses is beyond the scope of the present study, other than to note they are controversial.

In summary, it is clear that the ALARA type concepts are best suited to evaluating situations in which the exposed individuals can be identified and their radiation doses measured or predicted with some degree of reliability. Unfortunately, many situations involving radiation exposure cannot be evaluated in such a precise manner.

Although there are many difficulties in the application of ALARA it is nevertheless considered that this approach to deciding risk-benefit issues ideally comes closer to maximizing net social benefits than any other approach (Starr and Whipple 1980). To set priorities subjectively is to risk doing less than is possible.

4.0 THE SWANSON CASE STUDY

4.1 Introduction

In this section, analyses of the Swanson project (MUC 1983, MUC 1984) are used to illustrate the application of pathways analysis to uranium mining in the Virginia setting. It is important to note that these analyses were carried out on the basis of a preliminary project description. Should the project proceed, details of the project, including those which are used in the pathways analysis, would evolve as the proponent developed more detailed engineering design and conducted the environmental studies required for a licence application.

While it is beyond the scope of this report to comment on the particulars of the pathways analyses that were carried out by the proponent for the Swanson project, an overall comment may be appropriate. Our review indicates that, while there is some uncertainty as to the values of various inputs and model parameters most appropriate to the Swanson setting, the results are likely to be of the right order. Refinement of the pathways analysis procedures and inputs can reasonably be left to later stages of project development.

The general nature of the models used in the Swanson analysis are described, as are key input data. More detailed descriptions of the models (MILDOS, PABLM) are presented in the above noted Swanson project reports and the appropriate user's guides (Strenge and Bander 1982; INTERA Environmental Consultants 1983).

This section provides a discussion of dose to individual receptors. This is followed by a brief discussion of the potential population exposure, which is the incremental radiation exposure occurring to all persons living within approximately a 50 mile radius of the Swanson site. Finally, an attempt is made to place the anticipated incremental exposure levels in a suitable context.

4.2 Dose Estimates for the Swanson Uranium Project

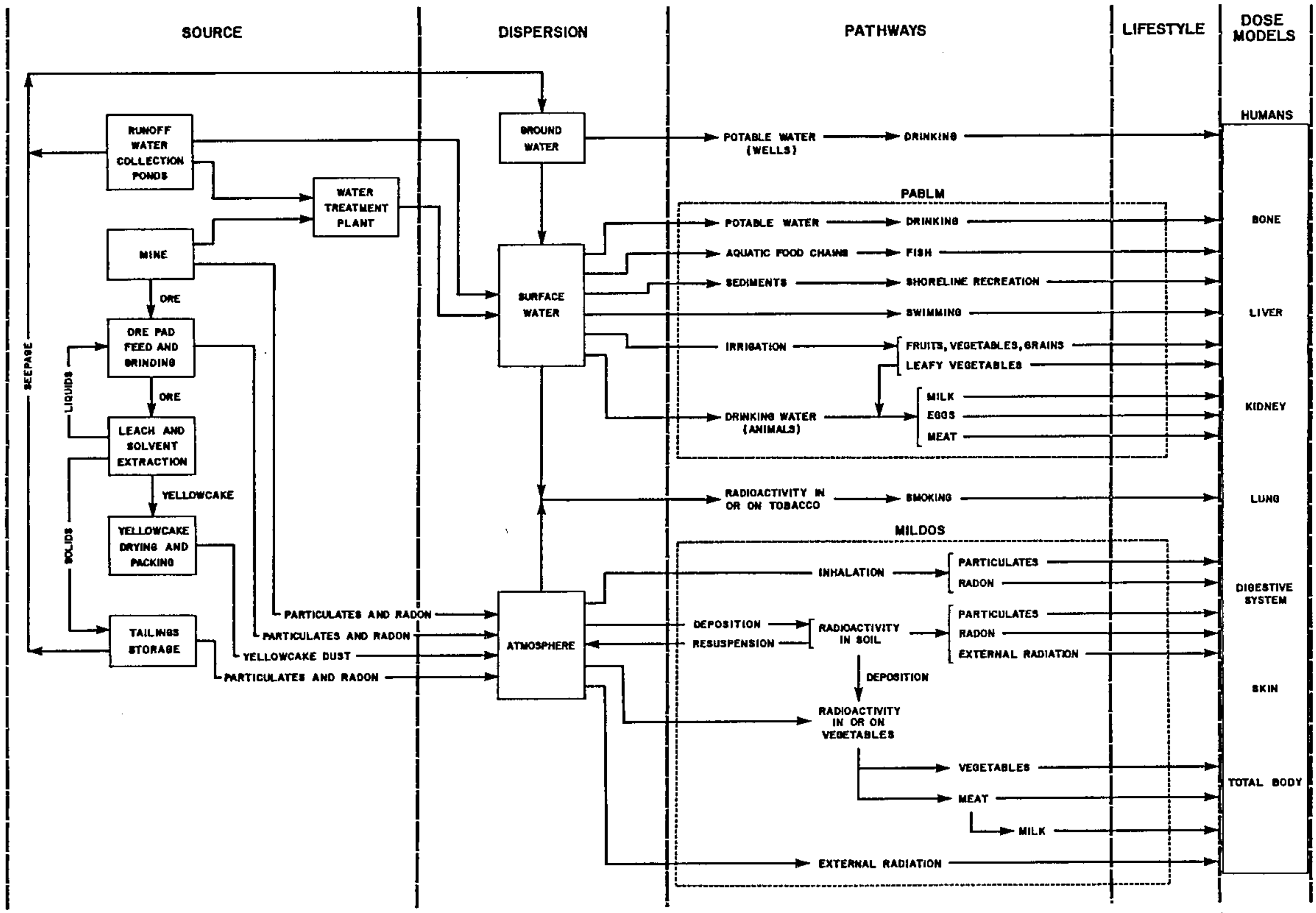
In the 1983 Swanson submission (MUC 1983), two computer codes were used to evaluate the potential population and critical individual dose that could result from the operation of the mine, mill and tailings facility. The first, MILDOS, was used to model the impact of releases of radionuclides to the air. The second, PABLM, was used to model the impact due to releases of radionuclides to the surface water (and indirectly the groundwater) environments. Figure 4.1 (taken from the 1983 Swanson submission) shows the potential public exposure pathways that the codes were used to evaluate.

A few comments on Figure 4.1 are required in that it implies that:

- the source and dispersion terms are not included in the programs. (This is not the case - both features are included in MILDOS and a form of dispersion (dilution) is included in PABLM.)
- these codes have identical lifestyle and dose models. (The codes were run with different assumptions.)

FIGURE 4·1

POTENTIAL EXPOSURE PATHWAYS FOR SWANSON CASE STUDY



ADAPTED FROM MUC 1983

- groundwater used for drinking water was included in the analysis. (The groundwater pathway was not evaluated. Groundwater flows containing seepage were assumed to enter a surface water stream prior to any consumption).

4.2.1 MILDOS

The MILDOS computer code was used to estimate the incremental radiological impact of airborne emissions from the Swanson uranium mine and mill. These impacts were presented as dose commitments to individuals and the regional population within a 50 mile radius of the facility. The pathways modelled using the MILDOS code are indicated in Figure 4.1.

The MILDOS code includes models that can be used to consider both point sources (stacks, vents) and area sources (mine, ore pads, tailings area, overburden storage). Releases of particulate matter are limited to consideration of uranium-238 (U-238), thorium-230 (Th-230), radium-226 (Ra-226) and lead-210 (Pb-210). Other radionuclides are implicitly accounted for by assuming secular equilibrium; that is, each radionuclide is present at the same activity as its parent. Gaseous releases are limited to consideration of radon-222 (Rn-222). Table 4.1 summarizes the source and emission data that were used to evaluate the Swanson project. In the MILDOS code, emissions are modelled using a straight-line, sector-averaged Gaussian plume dispersion model using Danville wind data (1950-1954 average). Annual average air concentrations are then computed for the special receptors shown on Figure 4.2 (for use in identifying the critical receptor) and for the midpoint of each spatial interval as defined by a circular grid centered on the facility and extending in 22.5° sectors and 10 kilometer intervals out to 80 km (50 miles) (for use in estimating the population dose).

The annual average air concentrations are used to compute a number of environmental concentrations. Ground surface concentrations are estimated from deposition buildup and ingrowth of radioactive daughters. The surface concentrations are modified by radioactive decay, weathering and other environmental processes. The concentrations of radionuclides on vegetation are calculated for the deposition of radionuclides from the air onto the plants and uptake from the ground surface for five categories of plants:

- edible above-ground vegetables
- potatoes
- other edible below-ground vegetables
- pasture grass
- hay

The two latter plant concentrations are used to calculate the concentration in meat and milk produced from animals feeding on these crops. All input parameters used in these calculations are default values; that is, values are provided in the MILDOS code which the modeller may modify if more appropriate values are known.

The pathways considered for individual dose commitments and for population impacts are:

FIGURE 4·2

RECEPTOR LOCATIONS AND PREDICTED TOTAL ANNUAL DOSE FOR MILDOS ASSESSMENT OF RADIOLOGICAL DOSE RATES FROM AIRBORNE PATHWAYS

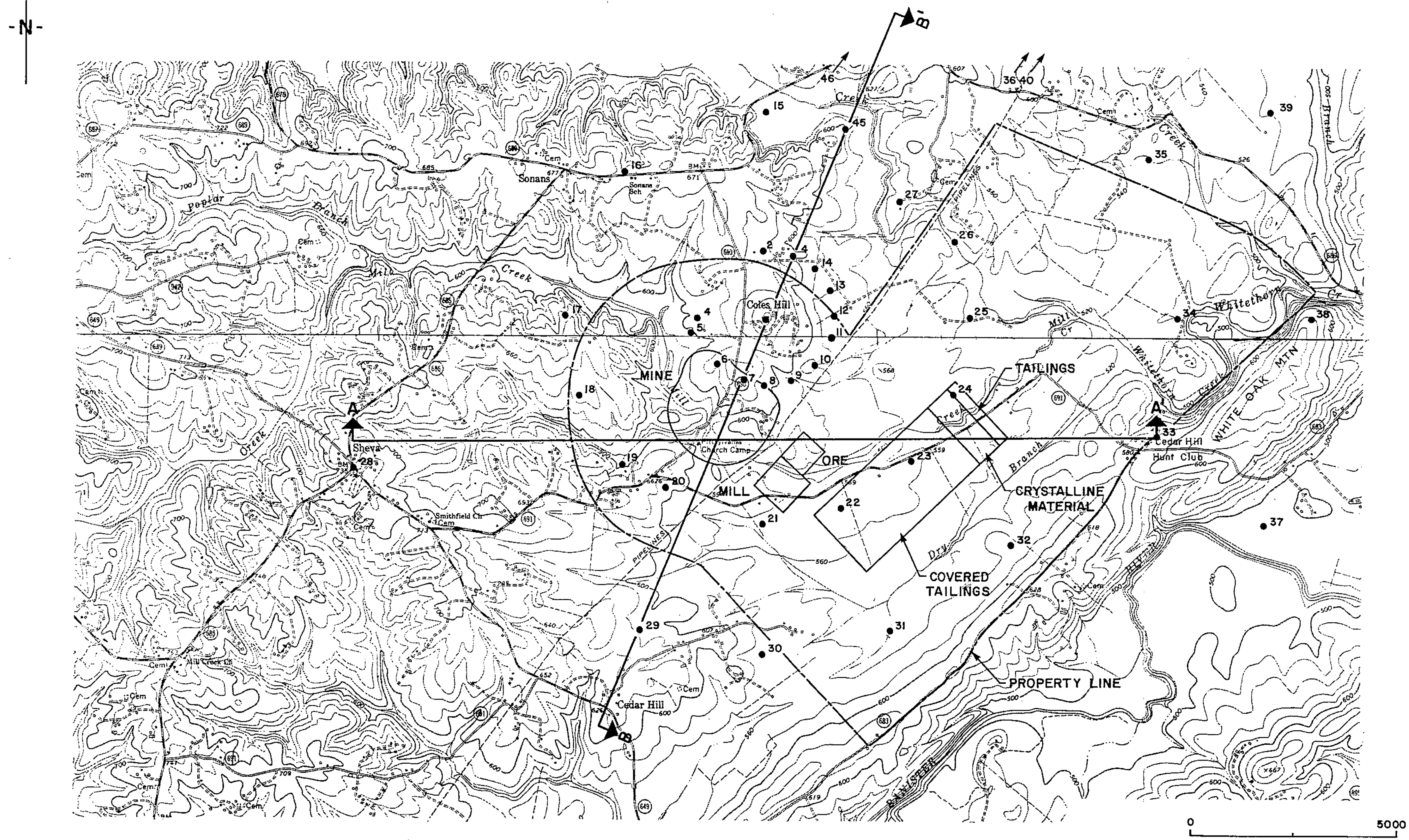


Table 4.1

Source and Emission Data used in MILDOS Runs (MUC 1983)

Source	Annual Emission* (Ci/year) U-238	Th-230	Ra-226	Pb-210	Rn-222	Area (km^2)	Comments
Overburden Storage	2.55×10^{-3}	2.55×10^{-3}	2.55×10^{-3}	2.55×10^{-3}	2.20×10^{2}	0.243	Dusting rates calculated by MILDOS
Open Pit Mine	4.83×10^{-2}	4.83×10^{-2}	4.83×10^{-2}	4.83×10^{-2}	4.34×10^{3}	0.445	Dusting rates calculated by MILDOS
Overburden on Tailings	1.27×10^{-3}	1.27×10^{-3}	1.27×10^{-3}	1.27×10^{-3}	1.09×10^{2}	0.121	Dusting rates calculated by MILDOS
Exposed Tailings	2.17×10^{-4}	4.34×10^{-3}	4.34×10^{-3}	4.34×10^{-3}	3.81×10^{2}	0.040	Dusting rates calculated by MILDOS
Ore Storage	6.62×10^{-3}	6.62×10^{-3}	6.62×10^{-3}	6.62×10^{-3}	1.21×10^{2}	0.061	Dusting rates calculated by
Yellowcake Stack	7.72×10^{-3}	3.86×10^{-4}	7.72×10^{-5}	7.72×10^{-5}	0	0	
Ore Crusher Releases	2.64×10^{-4}	2.64×10^{-4}	2.64×10^{-4}	2.64×10^{-4}	0	0	

Ref: MUC 1983, Vols. lA and 7.

* Based on time step 1 (operation) as discussed in MUC 1983

- inhalation
- external exposure from radionuclides deposited on the ground
- external exposure from cloud immersion
- ingestion of vegetables
- ingestion of meat
- ingestion of milk

Table 4.2 summarizes the assumptions upon which the dose calculations to individual receptors were based in the analysis of the Swanson project.

In the MILDOS code, the total population dose from ingestion pathways is calculated on the basis of regional agricultural productivity rather than population. This is because the total activity in the food determines the dose rather than the number of people exposed. Ingestion doses are calculated by the following procedure:

- the productivity rates are assigned as follows:

vegetables	1.72×10^3	$kg.yr^{-1}$ per m^2
meats	3.63×10^3	$kg.yr^{-1}$ per m^2
milk	7.34×10^3	$kg.yr^{-1}$ per m^2

- for each radial sector, the activity concentrations in each food type are calculated and multiplied by the production rate and the sector area to find the total activity in each food for the sector
- the total activity for the region is determined by summing all sectors
- population doses are determined by assuming all food produced in the region is consumed by a population with the same age distribution as the general U.S. population.

Dose conversion factors for both individual and population dose estimates are default values provided in the code. The internal exposures are based on the recommendations of the ICRP Task Group on Lung Dynamics (ICRP 1966, inhalation), the ICRP Publication 2 ingestion model (ICRP 1959, excluding Ra-226) and the ICRP Publication 10A dose ingestion model (ICRP 1971, Ra-226 only). External dose conversion factors are those presented in the Final Generic Environmental Impact Statement on Uranium Milling (NRC 1980).

The MILDOS code can be used to calculate individual dose totals which in turn can be used to evaluate compliance with EPA standards contained in 40 CFR 190 and NRC regulations presented in 10 CFR 20 (see Chapter 5.0). The 40 CFR 190 dose totals exclude all dose contributions from the short-lived radon daughters and all dose contributions from the long-lived radon daughters formed by the decay of released radon-222. The radon doses discussed in Section 4.3 were thus obtained by difference, using MILDOS data reported in MUC 1983.

The MILDOS code has the option to investigate more than one scenario. Two scenarios were studied during the Swanson assessment. The first scenario was for 13 years representing the maximum operational lifetime (doses calculated are for the 13th year), and the second was a two year time step representing a reclamation period (doses are calculated for the 15th year). Post-reclamation doses were not calculated.

Table 4.2

Assumptions* for Potential Critical Receptors (Adult) (MILDOS)

Exposure Pathway	Annual Intake	% Local	Comments
Inhalation	-	100	annual intake included in dose conversion factor
External (Air)	-	100	14 hours per day indoors
External (Ground)	-	100	14 hours per day indoors
Vegetables	105 kg	100	assumed to be 78.3% above ground vegetables, 19.6% potatoes, 2.1% other below ground vegetables
Meat	78.3 kg	100	
Milk	130 kg	100	

* MUC 1983

4.2.2 PABLM

The PABLM code was originally developed to evaluate the biosphere transport of radionuclides discharged to the biosphere from a geologic repository system and the subsequent dose to man. It has the capability to evaluate exposures from radionuclides released to the air or surface water and subsequently transported via aquatic and/or terrestrial pathways. However, for the Swanson project, PABLM was used to model only the liquid effluents in surface waters (in essence seepage was assumed to short-circuit the groundwater path and directly enter the local surface water system) and transfer to man.

PABLM does not model emissions to the surface water system. This information must be calculated outside the program. Table 4.3 summarizes the quality and quantity of the emissions that were assumed during the Swanson assessment for five source types: treated mine water, unattenuated tailings seepage, attenuated tailings seepage, overburden seepage and overburden runoff.

(A pond breach scenario which was also modelled is not included here as it pertains only to the accident analysis. Although beyond the scope of this study, it is appropriate to comment on the applicability of PABLM to assessing accidental releases. The PABLM models that are used to predict the behaviour of radionuclides in the environment are intended to simulate steady-state conditions with the source term assumed to be constant. Accidental releases involve a sudden release of radioactive material over short time intervals. Notwithstanding this difference, it is our view that PABLM provides a reasonable screening tool for evaluating doses from accidental releases when appropriate adjustments are made to take account of conditions anticipated during and following the release period.)

The treated mine water quality used in the Swanson analysis was based on laboratory leaching studies and treatability studies for radium-226.

The quality of the unattenuated seepage was based upon an alkaline tailings liquid simulated in the laboratory that is assumed to retain its full complement of radionuclides as seepage occurs through the clay liner beneath the tailings. The attenuated seepage was assumed to have its radionuclide content reduced by the various physico-chemical processes (e.g. absorption, adsorption) as it passes through the clay liner. The attenuated seepage scenario represents a somewhat optimistic expectation while the unattenuated seepage scenario represents a pessimistic situation. The real situation likely lies somewhere in between.

The overburden seepage and runoff concentrations were based on leaching tests conducted on relatively finely ground rock and are considered to be conservatively high values.

All stated quantities for the discharges were based on annual average flow rates.

Table 4.3 also summarizes the flow rates for the receiving streams. The proposed diversion of Mill Creek directs flows (via Dry Branch) into the Whitethorn Creek, which then discharges into the Banister River.

Table 4.3

Summary of PABLM Input Parameters (MUC 1984)

Source	Quality	Quantity	Receiving Stream
Treated Mine Water	U-nat - 0.5 mg/L Th-230 - 0.4 pCi/L Ra-226 - 3.0 pCi/L Pb-210 - 0.1 pCi/L Po-210 - 0.9 pCi/L	0.37 CFS	A. Mill Creek 10.5 CFS Whitethorn 30.0 CFS TOTAL 40.5 CFS B. Banister River 550 CFS
Tailings Seepage-Unattenuated	U-nat - 35 mg/L Th-230 - 162 pCi/L Ra-226 - 22 pCi/L Pb-210 - 7 pCi/L Po-210 - 1 pCi/L	0.203 CFS	Same as A and B above
Tailings Seepage - Attenuated	U-nat - 0.7 mg/L Th-230 - 1.0 pCi/L Ra-226 - 1.0 pCi/L Pb-210 - 2.0 pCi/L Po-210 - 1.0 pCi/L	0.203 CFS	Same as A and B above
Overburden Seepage	U-nat - 0.028 mg/L Th-230 - 0.203 pCi/L Ra-226 - 9.01 pCi/L Pb-210 - 0.187 pCi/L Po-210 - 0.116 pCi/L	Nonvegetated Overburden: 0.274 + Vegetated overburden: 0.540 TOTAL: 0.814 CFS	Same as A and B above
Overburden Runoff	U-nat - 0.014 mg/L Th-210 - 0.102 pCi/L Ra-226 - 4.505 pCi/L Pb-210 - 0.094 pCi/L Po-210 - 0.058 pCi/L	Nonvegetated Overburden: 0.002 CFS	Same as A and B above

PABLM then models incremental levels of radionuclides in agricultural soils, crops and forage (through sprinkler irrigation), aquatic foods and shoreline sediments.

Levels of radionuclides in agricultural soils are calculated by multiplying the irrigation rate by the deposition period. Radionuclides are assumed to be removed from the soil only by radioactive decay. Leaching from the soil and other removal mechanisms which could act to decrease exposure are not taken into account (a conservative assumption).

The concentrations of radionuclides in vegetation are calculated from the direct deposition of radionuclides onto plant surface from sprinkler irrigation and uptake from the ground surface for the following plants:

- leafy vegetables
- other above ground vegetables
- root vegetables (excluding potatoes)
- orchard fruit
- grain (other than wheat)
- forage

The latter is used to calculate the concentrations in eggs, milk, beef, pork and poultry produced from animals feeding on forage crops and on contaminated water. All input parameters used in these programs are available from Appendix D of the Swanson Report Technical Summary (MUC 1984) and are commonly used in the literature.

Radionuclide concentrations in fish are based on the radionuclide concentrations in the contaminated water.

Radionuclide concentrations in sediments are calculated assuming that there is a constant water concentration for each year of the release. The deposition rate to the sediment is assumed to be dependent only on the water concentration. Daughter radionuclides occur in the sediment from build up due to the decay of the parents in the sediment and direct deposition of daughter radionuclides in the water.

Radionuclide concentrations in drinking water obtained from the Halifax water treatment plant are calculated assuming element-specific removal rates in the treatment plant (see PABLM Documentation, INTERA Environmental Consultants 1983).

Using the above information, the following exposures can be calculated:

- external exposure while boating, swimming, standing on shore or farming on soils irrigated with contaminated water
- internal exposures from the consumption of river or treated water, fish, leafy vegetables, root vegetables, orchard fruit, grain, eggs, milk, beef, pork or poultry.

External doses from radionuclides deposited in farm fields are calculated assuming an infinite plane source model. For a person standing next to a body of contaminated water, the dose from radionuclides deposited in the shoreline sediments is calculated using the same model as that used for farm fields, modified to include a shore width factor. For persons swimming in contaminated water, the dose is calculated assuming the body of water is infinite in size. Persons boating on the water are assumed to be exposed to a dose rate half that to which swimmers are exposed.

Internal doses were calculated as a function of radionuclide concentration in food products, ingestion rates and radionuclide-specific dose-commitment factors. In the most recently reported runs (MUC 1984), the latter are reported to be based on ICRP Publication 10A for internally deposited radionuclides (ICRP 1971).

Annual doses in the first year of operation were calculated for two hypothetical individuals for each separate source. The first hypothetical individual was assumed to reside at the Cedar Hill Hunt Club, eat fish and use Whitethorn Creek water to irrigate his farm on which he produces 50% of his annual intake of vegetables, fruit, grains, eggs, milk, beef, pork and poultry. Although it is unlikely that people drink water from Whitethorn Creek a nominal consumption was used in the model to permit examination of this pathway. Table 4.4 summarizes the data used.

The second hypothetical individual was assumed to live in Halifax where the municipal water treatment plant draws from the Banister River. A typical resident was assumed to drink 440 L of this water per year. This individual also eats 2.2 kg of fish per year from the Banister River. Twenty hours per year are spent at the river's edge, ten hours swimming in the Banister and five hours boating on the Banister. All doses discussed are for the first year that all emissions reach the surface water environment.

4.3 Annual Dose Estimates for Individual Receptors

4.3.1 Pattern of Doses to Individuals Near the Facility

The MILDOS code was used to estimate annual exposures of individuals at 46 locations as shown on Figure 4.2 (MUC 1983). While the principal intent of this modelling was to identify a critical receptor (the individual(s) receiving the highest annual exposure), it is also useful to examine the results to obtain a feeling for how the doses vary with distance from the facility.

Figures 4.3 and 4.4 use MILDOS output to illustrate the way in which the total annual dose from releases to the air depends on the distance and direction from the different source types using two transects identified on Figure 4.2. It is clear from inspection of these figures that the dose decreases rapidly as one moves away from the source.

Similarly, the dose to individuals from emissions to the water pathways will decrease with distance from the source. This is simply due to dilution in the water course.

FIGURE 4·3

ANNUAL WHOLE BODY EXPOSURES TO INDIVIDUALS HAVING CHARACTERISTICS OF THE CRITICAL RECEPTOR DUE TO PARTICULATE AND RADON EMISSIONS FOR TRANSECT A-A'
(SEE FIGURE 4·2)

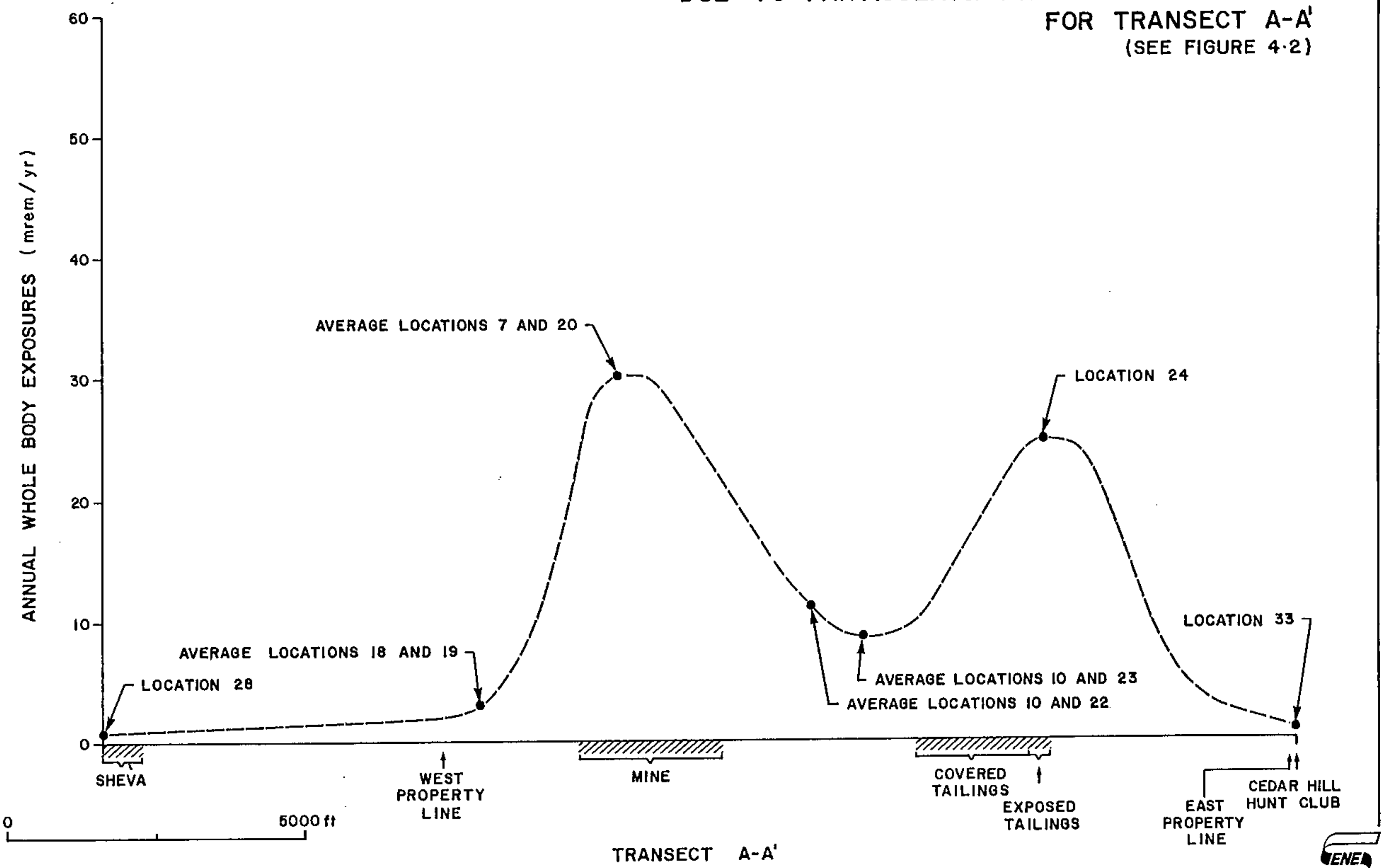


FIGURE 4·4

ANNUAL WHOLE BODY EXPOSURES TO INDIVIDUALS HAVING CHARACTERISTICS OF THE CRITICAL RECEPTOR DUE TO PARTICULATE AND RADON EMISSIONS FOR TRANSECT B-B'

(SEE FIGURE 4·2)

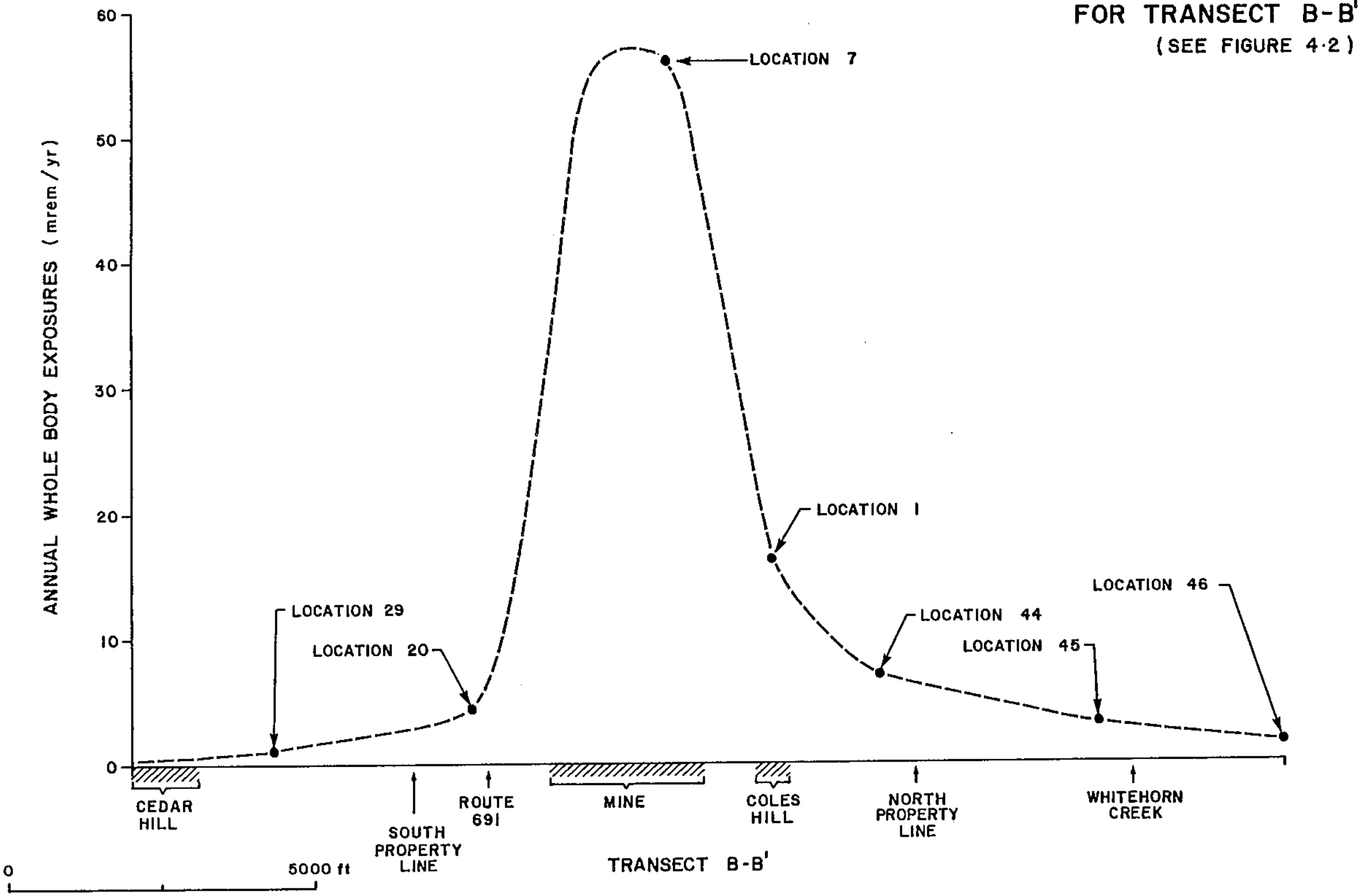


Table 4.4

Assumptions* Used for the Cedar Hill Hunt Club Receptor (Adult) (PABLM)

Ingestion	Annual Intake (kg/yr)
Leafy vegetables	7.5
Other above-ground vegetables	7.5
Other root vegetables	60
Orchard fruits	32
Grain (other than wheat)	40
Eggs	20
Milk	115
Beef	20
Pork	15
Poultry	4.25
Fish	2.2
Water	1.0 (L/yr)
External	
time spent on irrigated fields	4380 hours/yr

* MUC 1984

4.3.2 Identification of the Critical Receptor

One residence within the property boundary controlled by mining operations would be occupied. Residents at this location (Coles Hill property) would be exposed to a whole body dose of about 16.4 mrem/yr taking account of airborne release from the mine, mill and tailings facility and including the contribution of radon and its daughter products (MUC 1983).

Table 4.5 presents the annual dose estimates for the critical receptor as identified by the modelling discussed in Section 4.2 (location 14). Since a residence is not located at this location at this time Table 4.6 has been included to present information on the maximum annual exposure predicted for an existing residence (location 27). It should be noted that neither of these receptors are impacted by emissions to surface water.

Table 4.7 presents the results for a receptor assumed to reside at the Cedar Hill Hunt Club (location 33). While this is not the most exposed receptor, this individual does receive the maximum annual dose of any receptor exposed to both surface water emissions and air releases, and was included for this reason.

Table 4.8 presents the results for the hypothetical average resident of Halifax.

Details of the models and assumptions used were presented in Section 4.2. The implications of these dose estimates will be discussed in Section 4.5.

4.4 Population Dose Estimates

Table 4.9 presents the annual population doses (man-rem/year) from the combined facility for a radius of 50 miles (80 km) centered on the Swanson project.

The dose due to air emissions was calculated by MILDOS as discussed in Section 4.2.1.

The water doses were derived from the PABLM modelling discussed in Section 4.2.2 as follows:

- a ratio of the concentration of radionuclides in water in the Banister River at Halifax to the concentration of radionuclides in water in the Whitethorn at the Cedar Hill Hunt Club were developed
- the annual dose to an average individual using the Banister River at Halifax for fishing, irrigation and drinking water was estimated by multiplying the annual dose to the Cedar Hill Hunt Club critical receptor by this ratio and taking relative useage into account
- all persons living in the sector (generally to the south-east of the site to a distance of 50 miles) through which the Banister flows were assumed to be exposed at this average annual dose rate.

Table 4.5

Annual Dose Estimates for An Individual Receptor: Model Location 14 (Maximum Off Property, Unoccupied)

	Annual Doses (mrem/yr)			
	Air Releases[1] (Year 13)			
Source	Particulate	Radon	Water Releases[2] (Year 1*)	Total (Air Plus Water)
Mine	0.106	7.218	Mine Water Release: 0 Overburden Seepage: 0 Overburden Runoff: 0	7.3
Mill and Tailings	0.179	0.267	Attenuated Seepage: 0 Unattenuated Seepage: 0	0.5
Combined Facility	7.8		Attenuated Seepage: 0 Unattenuated Seepage: 0	7.8

[1] MUC 1983
[2] MUC 1984
* Water Releases Assumed Not to Impact on this Receptor

Table 4.6

Annual Dose Estimates for An Individual Receptor:
Model Location 27 (Maximum Off Property, Occupied)

	Annual Doses (mrem/yr)			
	Air Releases[1] (Year 13)			
Source	Particulate	Radon	Water Releases[2] (Year 1*)	Total (Air Plus Water)
Mine	0.727	2.27	Mine Water Release: 0 Overburden Seepage: 0 Overburden Runoff: 0	3.0
Mill and Tailings	0.181	0.321	Attenuated Seepage: 0 Unattenuated Seepage: 0	0.5
Combined Facility	3.5		Attenuated Seepage: 0 Unattenuated Seepage: 0	3.5

[1] MUC 1983
[2] MUC 1984
* Water Releases Assumed Not to Impact on this Receptor

Table 4.7

Annual Dose Estimates for An Individual Receptor: Model Location 33 (Cedar Hill Hunt Club)

Source	Annual Doses (mrem/yr) Air Releases[1] (Year 13) Particulate	Radon	Water Releases[2] (Year 1*)	Total (Air Plus Water)
Mine	0.171	0.803	Mine Water Release: 0.012 Overburden Seepage: 0.013 Overburden Runoff: 0.00001	1.0
Mill and Tailings	0.158	0.398	Attenuated Seepage: 0.009 Unattenuated Seepage: 0.410	Attenuated: 0.6 Unattenuated: 1.0
Combined Facility	1.5		Attenuated Seepage: 0.03 Unattenuated Seepage: 0.4	Attenuated: 1.6 Unattenuated: 2.0

1 MUC 1983
2 MUC 1984
* The first year that the seepage reaches the environment.

Table 4.8

Annual Dose Estimates for An Individual Receptor: Average Halifax Resident

	Annual Doses (mrem/yr)			
Source	Air Releases[1] (Year 13)** Particulate	Radon	Water Releases[2] (Year 1*)	Total (Air Plus Water)
Mine	0.001	0.027	Mine Water Release: 0.003 Overburden Seepage: 0.001 Overburden Runoff: 0.000002	0.03
Mill and Tailings	0.0005	0.008	Attenuated Seepage: 0.002 Unattenuated Seepage: 0.108	Attenuated: 0.01 Unattenuated: 0.12
Combined Facility	0.037		Attenuated Seepage: 0.006 Unattenuated Seepage: 0.11	Attenuated: 0.04 Unattenuated: 0.15

1 MUC 1983
2 MUC 1984
* The first year that the seepage reaches the environment
** Assumed to be the same as for Danville

Table 4.9

Annual Population Doses in man-rem/yr From the Mine, Mill and Tailings Area For the People Within 50 miles (80 km)

Source	Air	Water*	Total
Combined Facility	26.7	Attenuated: 0.3 Unattenuated: 4.2	Attenuated: 27.0 Unattentuated: 30.9
No. of exposed Persons	789,112	27,637	
Average Dose**	3.4×10^{-5}	Attenuated: 1.1×10^{-5} Unattenuated: 1.5×10^{-4}	Attenuated: 3.4×10^{-5} Unattenuated: 3.9×10^{-5}

Notes:

* the 27,637 people exposed via the water pathway and part of the regional population of 789,112.

** air: 26.7 rem divided by 789,112 people
water: 0.3 or 4.2 rem divided by 27,637 people
total: 27 or 30.9 rem divided by 789,112 people

4.5 **Discussion**

Individual Dose - Potential Critical Receptors

In Section 4.3, annual doses are presented for four possible critical receptors in terms of contributions to dose by source type (i.e. mine, mill and tailings and combined facility) and pathway (i.e. air and water). All dose contributions, including radon, are expressed in terms of millirem per year. Some of the uncertainties associated with expressing individual dose contributions in this fashion have already been discussed previously (see Section 3.2). Notwithstanding these difficulties, this mode of expression is convenient since it permits direct examination of the relative contributions of each source and pathway either in terms of dose or risk.

From a comparison of Tables 4.5, 4.6 and 4.7 it can be seen that the largest potential dose to an individual via the air pathway would occur at a location not currently occupied. Excluding radon, an individual living at this location (see Table 4.5) would receive an annual dose of about 0.3 mrem/yr from the combined facility, all air pathways taken into account. The dose equivalent from radon contributes about 96% of the total annual dose, about 7.5 mrem/yr. Interestingly, the airborne releases from the mine contribute nearly 94% of the total annual dose.

Similarly from Tables 4.6 and 4.7, releases from the mine can be seen to contribute a larger fraction of the dose via the air pathway than dose releases from the mill and tailings facility. Radon remains the largest contribution to dose.

Excluding the dose from radon, the largest annual dose via air pathways for the combined facility was estimated to be about 0.91 mrem/yr for location 27, well below the EPA criteria of 25 mrem/yr (EPA 1977).

The average annual dose due to natural external sources of radiation and (dose equivalent) due to the inhalation of naturally occurring radon at the Swanson site are in the order of 90 mrem/yr and 120 mrem/yr respectively. The maximum predicted incremental dose of 7.8 mrem/yr is thus only about 3.4% of natural background, and indeed is even smaller than the normal variability in natural background.

Only the Cedar Hill Hunt Club resident is impacted by releases to the water environment. In that case (see Table 4.7), the predicted total annual dose would be between 1.6 and 2.0 mrem/yr, for attenuated and unnattenuated seepage respectively. The mine is estimated to contribute about 50% of the total annual dose. Radon contributes about two-thirds of the dose via the air pathways for this individual.

Individual Dose - Average Member of the Population

From Table 4.9, it can be seen that the total annual dose to an "average" member of the population is predicted to be in the range of 0.034 to 0.039 mrem/yr for attenuated and unattenuated seepage respectively. These values represent about 2% of the dose estimated to be received by the receptor residing at the Cedar Hill Hunt Club. Thus the ratio between the maximally exposed individual and the "average" individual is large, about a factor of 50.

Population Dose

The cumulative annual population dose for releases from the combined facility was estimated to be between 27 and 31 man-rem/yr. This figure represents less than 0.02% of the dose the same population would receive every year from existing, naturally occurring sources of external radiation and from the inhalation of radon.

5.0 CONSIDERATIONS IN SETTING STANDARDS FOR VIRGINIA

The purpose of this section is not to advocate standards for uranium mining and milling in Virginia, rather it is to provide brief discussions of various factors which should be taken into account by decision makers in their deliberations as to whether uranium mining should take place in Virginia, and if so, under what conditions.

For a uranium mine, mill and tailings complex there are three possible routes of exposure to members of the public: external whole body radiation, ingestion of radionuclides in food or water, and the inhalation of either radon and its daughters or radionuclides in suspended dust (see Chapter 2). Experience suggests that external whole body radiation results in a much smaller dose than that from ingestion or inhalation. The doses arising from radionuclides taken into the body varies from radionuclide to radionuclide and are not directly additive. The ICRP in Publication 26 (1977) outlines a procedure for multiplying organ doses by weighting factors to convert organ doses to effective whole body doses which can then be added because they represent equivalent risks (see Section 3.3). In ICRP 32 (1981), the concept is extended to include exposure to radon and its daughters. Thus it is possible to add together the doses from the various radionuclides and modes of exposure. It is important to remember, however, that to carry out this summation, the different types of exposure must be first converted to a common denominator, namely risk.

The total regulation of uranium mining and milling is complex and involves numerous federal and state agencies. Given the preemptive nature of federal regulations, they are the first topic addressed in this chapter.

In establishing radiation protection standards, a balance is required between risk and benefit since reducing the (incremental) risk to zero would eliminate the opportunity to mine uranium and result in the loss of any benefit. The level of (radiological) risk associated with a given level of regulation is therefore of considerable interest. In Section 5.2, the risks from various levels of radiation exposure are placed in perspective by comparing them to risk from other hazards.

In Section 5.3 a number of specific factors which may affect either the selection of a radiation protection standard or the way it is applied are identified and briefly discussed.

Based on discussions and analysis presented earlier in this report, Section 5.4 outlines one possible approach to setting risk-based standards.

5.1 Federal Standards and Regulations

Two federal agencies are largely responsible for regulating the environmental aspects of uranium mills and tailings management operations: the Environmental Protection Agency (EPA) and the Nuclear Regulatory Commission (NRC). The EPA is responsible for setting standards that protect the public and the general environment. The NRC issues regulations intended to ensure that the EPA standards are realized and grants licenses to qualified operators.

When the EPA proposes new standards or amendments to existing standards that concern uranium mill tailings management or disposal, the NRC is required to suspend any provisions of its own regulations that could become unnecessary and to subsequently make the appropriate changes. As described below, such a procedure is currently in progress.

5.1.1 Environmental Protection Agency

In 1977, the EPA formally announced standards for radiation doses received by members of the public and for radioactive materials introduced into the general environment from nuclear fuel operations (EPA 1977). Issued as Part 190 to Title 40, Code of Federal Regulations (often referred to as 40 CFR Part 190), the standards apply to uranium milling operations but exclude mining.

The standards in Part 190 also apply to radioactive particulate matter emissions from tailings areas during operations but do not include exposures resulting from the release of radon or its decay products. In the final environmental statement for Part 190, the EPA indicated that "the problems associated with radon emissions are sufficiently different from those of other radioactive materials associated with the (nuclear) fuel cycle to warrant separate consideration" (EPA 1976).

Part 190 requires that the annual dose equivalent not exceed 25 mrem to the whole body, 75 mrem to the thyroid and 25 mrem to any other organ of any member of the public as a result of planned discharges from nuclear fuel cycle operations. In determining these values, measured and simulated impacts of radioactive material releases from nuclear fuel cycle operations were evaluated. The EPA concurred with the National Academy of Science committee (NAS BEIR 1972) conclusion that the weight of scientific evidence strongly supports the continued use of a linear, non-threshold model for assessing the implications (risks) of low doses or dose rates and that it remains an appropriate method for setting standards.

The analysis of health impacts used by the EPA to develop its regulations considered the radiation doses committed to local, regional, national, and worldwide populations as well as doses due to the long-term persistence of some of the radioactive materials found in releases. In addition to large populations, doses to critical individuals were also evaluated.

The EPA concluded that the 25 mrem/yr standard can be satisfied by levels of control that are cost effective for the risk reduction achieved; can be readily achieved in practice; provides a reasonable margin of operating flexibility; and, provides an ample margin for areas where several operations are located. Critics have questioned whether the standard was based on a proper risk assessment or was the result of using the ALARA concept improperly. Various EPA statements and documents have been cited as indicating the latter rather than the former. For example, a senior EPA official publicly stated that the intent of the 25 mrem/yr standard was "to get the releases from (nuclear fuel cycle) facilities to as low as reasonably achievable level, and that is basically what the standard was based on " (Sjoblom 1982).

The EPA has also issued standards for releases to the environment from mills and tailings areas. In 1981, 40 CFR Part 440, Subpart E established standards for waste water quality from uranium mills (EPA 1981). In 1983, the EPA formally issued 40 CFR Part 192 which addressed the releases of radionuclides from tailings sites. The EPA concluded that during the active phase of a tailings site, the requirements embodied in the NRC regulations of 10 CFR Part 20 (described below) assured adequate control of radon releases (EPA 1983b). For tailings sites after closeout, 40 CFR Part 192 requires that the average radon release rate from the surface not exceed 20 $pCi/m^2/s$. The same criticism levelled at 40 CFR Part 190 (the use of ALARA rather than risk analysis to set the standards) has also been directed toward 40 CFR Part 192.

5.1.2 Nuclear Regulatory Commission

Regulations concerning protection against radiation hazards arising from activities under licenses issued by the NRC are contained in 10 CFR Part 20 (NRC 1977). Among other things, these regulations state that the level of radiation in unrestricted areas (that is, any area to which access is not controlled by the operator or licensee) must be such that any individual is unlikely to receive a dose to the whole body in any one calendar year in excess of 0.5 rem (500 mrem). Also stipulated are maximum radiation exposure levels for one hour and one week. Requirements are also presented in 10 CFR Part 20 for the maximum permissible concentrations of individual radionuclides (including radon-222) being released to unrestricted areas. These maximum permissible concentrations, to be determined on an annual average basis, apply to incremental (that is, above natural background) levels. The appropriate number for the incremental concentration of radon is 3 pCi/L in unrestricted areas.

Other Parts of 10 CFR address licensing policy and requirements and procedures for environmental protection. In accordance with these responsibilities, NRC has issued various regulatory guides and impact statements. The NRC had determined that radon emissions from buried tailings should be limited to 2 $pCi/m^2/s$ (NRC 1980). Shortly following the issuance of 40 CFR Part 192, the NRC announced that several of its regulations pertaining to tailings management, including the 2 $pCi/m^2/s$ requirement, had been suspended. The NRC is currently determining the modifications that its regulations require. These modifications are expected to be announced shortly although definite dates have not been identified.

5.2 Risk Perspective

In earlier sections of this report, the desirability of a risk-based standard has been referenced. In this section, the level of risk associated with alternative levels of regulation are discussed.

Moeller et al (1983) examined the literature on criteria for dose limits to the public. In Table 2 of their paper, they summarize various dose rate limits for the general public as recommended by the ICRP, NCRP and the Federal Radiation Council (FRC). All of these groups recommend an individual dose limit of 500 mrem/yr. An abbreviated form of this table is presented in Table 5.1

Table 5.1

Individual Dose Limits and Their Implications for the General Public

	Principal Source of the Recommendation	General Population: Critical Group Maximum	General Population: Critical Group Average	General Population: Average**
Dose or Dose Limit	ICRP 26 (1977)	500 mrem/yr	100 mrem/yr*	$\leq$50 mrem/yr
	NCRP 39 (1971)	500 mrem/yr	170 mrem/yr	$\leq$100 mrem/yr
	FRC (1960)	500 mrem/yr	170 mrem/yr	-
Approximate Increment in Mortality Risk		50×10^{-6}/yr	10×10^{-6}/yr to 17×10^{-6}/yr	5×10^{-6}/yr to 10×10^{-6}/yr

Notes:

(1) Adapted from Moeller et al (1983)

(2) The mortality risks are based upon a lifetime risk coefficient for fatal cancers of 1×10^{-4}/rem.

(3) The risk estimates are very crude and do not take into account variations in sensitivity, duration of exposures, remaining life expectancy, etc.

* Averaged over a lifetime.

**Average member of the public.

The annual dose limits shown in Table 5.1 indicate that on the basis of a linear dose-effect relationship and a risk factor of 1×10^{-4} per rem, the most exposed individual would experience an annual risk of about 50 per million per year. On average, members of the general population would be exposed to an annual risk of 5 per million per year or less. The latter level of risk has been suggested by various people as being trivial or negligible (de minimis, see Section 3.4). It should be remembered that the ICRP, NCRP and FRC all indicate that the dose limits for average exposures to members of the public are guides and that higher average exposures could possibly be justified on the basis of a cost-benefit analysis.

In any event, Virginia will have to decide what it considers to be an acceptable level of risk for exposure to either individual dose components or combined exposures.

The level of risk associated with continuous (i.e. lifetime) exposure to various annual levels of radiation exposure is illustrated in Table 5.2. In Table 5.2a, annual dose equivalent can be interpreted in terms of whole body external radiation, or the whole body dose equivalent from radionuclides taken into the body, or the dose which would result in a risk equal to the risk from a given radon daughter exposure, or any combination thereof. Table 5.2b summarizes the risk from different ambient levels of radon-222. The risk levels shown in Table 5.2 can be compared to the everyday risks shown on Table 3.1.

5.3 Some Additional Factors Relevant to Risk Management

Various scientific and philosophical considerations related to the derivation of radiation protection standards have been discussed in previous sections. Evaluation of the potential radiological risk from uranium development has been the focus of this report. Because a precise evaluation of risk is not possible, a variety of approaches have been used to try to place the level of risk in a reasonable perspective. As noted previously, this risk assessment study will form one of the bases that the decision makers use to determine whether or not uranium mining should proceed in Virginia, and if so under what conditions. This type of risk management decision requires a balancing of potential risks and potential benefits.

In this section, a number of concepts which are helpful in interpreting a risk assessment in order to develop a risk management strategy are discussed.

Critical Group

As discussed in previous sections, the actual dose received by an individual member of the public will depend on many factors including individual habits and local environmental conditions. However, it is usually possible to identify population groups with characteristics causing them to be more exposed than other members of the public. The ICRP suggest that when the dose is extended in time (as in the case of uranium development), it is "the maximum of the average dose equivalent in the critical groups that should be compared with the corresponding dose-equivalent limit" (ICRP 26 para 217 1977).

Table 5.2

Risk of Radiation Exposure

a. From Whole Body Radiation

Annual Dose Equivalent (mrem/yr)	Annual Risk Per Million Persons Exposed At The Specified Level
500	50
170	17
100	10
25	2.5
10	1
1	0.1

b. From Inhalation of Radon

Radon Level (pCi/L)	Annual Risk Per Million Persons Exposed at The Specified Level
1.92	50
0.65	17
0.38	10
0.096	2.5
0.0038	1

Notes:

1) A conversion factor relating pCi/L of radon to an equivalent (on a risk basis) whole body exposure can be derived as follows for continuous exposure conditions:

$$1 \text{ pCi/L} = 1 \text{ pCi/L} \times \frac{0.5 \quad \text{WL}}{100 \quad \text{pCi/L}} \times \frac{8760 \text{ hours/year}}{170 \text{ hours/working month}} = 0.26 \text{ WLM/year}$$

This assumes an average radon daughter/radon equilibrium factor of 0.5. On the basis that the risk of mortality from exposure to 1 WLM is approximately equal to the risk of mortality from exposure to 1 rem of whole body radiation, the risk from continuous exposure to 1 pCi/L of radon-222 is approximately equal to the risk from whole body exposure to 260 mrem per year, or 1 pCi/L = 260 mrem per year on a risk basis.

2) Existing indoor and outdoor radon levels in vicinity of Swanson Project are 1.6 $\pm$ 1.9 pCi/L and 0.58 $\pm$ 0.2 pCi/L, respectively (MUC 1983).

Finally, it is worth noting that the ICRP suggests that "when several practices may contribute significantly to the exposure of the same exposed population, either simultaneously or successively, the definition of these critical groups must take account of these separate contributions" (ICRP 26 para 216 1977).

Maximum and Average Exposure

As noted in Section 5.2, the dose limit suggested for individual members of the public (i.e. a member of a critical group) is 500 mrem/yr as recommended by the ICRP (Publication 26 1977), the NCRP (Publication 39 1971) and the FRC (1960).

The ICRP suggest that for continuous lifelong exposure to an individual member of the public, a dose of about 100 mrem/yr would be appropriate (ICRP 26 para 119 1977). Moreover, the ICRP suggest that the application of an annual dose equivalent limit of 500 mrem/yr to individual members of the public is likely to result in average dose equivalent of less than 50 mrem/yr (i.e. a factor of ten lower). Indeed, for the Swanson project (see Chapter 4) the most exposed individual, living outside the property boundary controlled by the mining operations, was estimated to receive a total dose equivalent exposure of about 7.8 mrem/yr (Table 4.5), compared to a total dose equivalent of less than 0.04 mrem/yr for an average member of the regional population (Table 4.9), or about one-two hundredth of the dose to the most exposed individual.

The FRC (1960) and the NCRP (1971) both incorporate a factor of three between the limit for the maximum and the average individual from the critical group. They justified this factor on the basis that the majority of individuals do not vary from the average by more than a factor of about three (FRC Para 5.4 1960).

All three groups (ICRP, NCRP, FRC) encourage the keeping of radiation doses as low as practicable (or specifically in the case of the ICRP, the use of ALARA - As Low As Reasonably Achievable, social and economic factors taken into account).

Natural Radiation

In comparing individual doses to criteria levels or standards, the doses from natural radiation are not included (e.g. ICRP 26 para 212 1977). The ICRP, NCRP and FRC provide various philosophical bases for this position and the interested reader is referred to reports of the three agencies which have been previously cited for details. One key consideration is the observation that man has evolved in the presence of natural radiation. The FRC provides the following comment which may provide insight in this area: "We believe that the current population exposure resulting from background radiation is a most important starting point in the establishment of Radiation Protection Guides for the general population. This exposure has been present throughout the history of mankind, and the human race has demonstrated an ability to survive in spite of any deleterious effects that may result. Radiation exposures received by different individuals as a result of natural background are subject to appreciable variation. Yet, any differences in effects that may result have not been sufficiently great to lead to attempts to control background radiation or to select our environment with background radiation in mind." (FRC 1960 para 5.2).

Risk Versus Benefits

To carry out a risk-benefit analysis (or an ALARA analysis - see Section 3.5 of this report) it is necessary to make an evaluation of the total risk from a particular operation. The difficulty is in how to value the collective dose commitment and associated risks or costs derived from estimation of ever smaller doses to increasingly large populations as larger and larger areas are included in the dose integrations. One suggested resolution to the problem has been the application of a de minimis dose concept to provide a limit to the integration of collective dose.

From the Swanson case study discussed in Chapter 4, the maximum annual dose equivalent commitment for an individual off-site receptor was estimated to be approximately 7.8 mrem/yr, and the annual dose equivalent commitment for an average resident of the region within 50 miles (80 km) of the site was estimated to be less than 0.04 mrem/yr. As discussed in Section 3.4, even the maximum annual dose is within the range of de minimis values which have been proposed and the average dose is at the lower end of the de minimis values. Thus the collective dose estimated for all persons living within an 50 mile radius of the site would seem to provide a reasonable upper value of dose (and hence risk) for use in risk-benefit analyses which are based on annualized benefit and risks.

5.4 Approach to Regulation

Regardless of whether or not Virginia chooses to become an agreement state, if uranium development is permitted it will be necessary, as a minimum, to comply with federal regulations. This would mean that no individual in an unrestricted area should receive an annual whole body dose in excess of 500 mrem/yr (NRC) and, excluding the dose from radon, no member of the public should be exposed to a dose equivalent in excess of 25 mrem/yr (EPA). Both of the federal standards apply to radiation and radioactivity from uranium mills and tailings but specifically exclude the effect of releases from mining.

In the Swanson case study discussed in Chapter 4, it is clear that radioactive releases from the mine (radon in particular) contribute a large fraction to the annual total whole body dose equivalent. In the case of the most exposed hypothetical off-site receptor, releases from the mine would contribute nearly 94% of the total annual dose. While the fractional contribution of the mine to the total dose changes from receptor to receptor, at least for the Swanson project, it is always a relatively large fraction of the total dose. Thus it seems that it is appropriate to consider potential dose contributions from all sources (i.e. mine, mill and tailings) when evaluating the potential risk from a uranium mine/mill complex.

In Virginia there is a net water surplus. This suggests that it is important to consider the potential effect of releases to the water environment. In the case of the Swanson Case Study, the hypothetical receptor assumed to live at the Cedar Hill Hunt Club (see Table 4.7) was estimated to receive a total exposure of between 1.6 and 2.0 mrem/yr for attenuated and unattenuated seepage, respectively. This example illustrates the importance of ensuring that both air and water pathways of exposure are identified and evaluated in a risk assessment.

In Section 5.2 various levels of regulations were discussed. A 500 mrem/yr whole body dose equivalent is suggested by the ICRP, among others, as being an appropriate limit for the most exposed individual. The ICRP also suggest that by restricting the maximum individual exposure to 500 mrem/yr the exposure to an average individual will be much less, perhaps a factor of ten smaller.

In the case of the Swanson case study, the annual dose equivalent to an average individual living within a 50 mile radius of the site was was about one-two hundredth of the maximum individual dose. It may be of interest to note that the maximum individual dose of 7.8 mrem/yr standard for the Swanson project is of the same order as the maximum individual exposures which have been estimated for uranium facilities elsewhere in the U.S., Canada and overseas.

A total annual dose equivalent of between 27 and 31 man-rem per year of operation has been estimated for the general population living within a 50 mile radius of the Swanson site. This one year of operation corresponds to an eventual (i.e. over the lifetime of those exposed) risk of mortality of at most

$$31 \text{ man-rem} \times \frac{1}{10{,}000} \; \frac{\text{lifetime risk}}{\text{man-rem}} = 0.003$$

For the 13 year operating life of the project, this risk value of 0.003 increases by a factor of 13 to 0.04 excess cancers. This statistical excess of 0.04 fatal cancers for releases from the full thirteen years of operation can be compared to the natural risk of fatal cancers in the same population. Since approximately 18% of all Americans will, at today's rates, eventually die of cancer, about 140,000 cancer deaths would naturally be expected in the 789,112 exposed persons.

Following closeout, the major dose contributor, radon, will largely be eliminated and other dose contributions will be reduced below operational levels. It is clear that releases from a well designed and operated facility should result in only an extremely small incremental risk.

With the foregoing in mind, the following considerations appear relevant to setting radiation protection standards for uranium mining in Virginia:

- The prime standard should be a maximum annual dose consistent with a risk considered acceptable by Virginia.

- In assessing the dose to an exposed individual, all sources and potentially significant pathways should be considered.

- The average dose should be lower than the maximum dose, perhaps by a factor of 3 to 10.

- Efforts should be made to ensure that all doses be kept as far below the maximum dose limit as practicable. In the context of radiation protection we suggest the application of ALARA - that is, keeping doses as low as reasonably achievable, social and economic factors taken into account. While such an objective is desirable, the way in which an ALARA analysis should be carried out is not well defined at present and a self-consistent approach would need to be developed by the responsible Virginia agencies.

- A lead agency responsible for coordinating all state input should be selected.

- Once a prime standard has been selected, secondary criteria (eg. concentrations in air and water) and procedures for determining compliance (eg. monitoring and modelling) need to be developed by state authorities.

REFERENCES

Adler, H.L. and Weinberg, A.M., 1978. "An Approach to Setting Radiation Standards." Health Physics, Vol. 34, pp. 719-720, June 1978.

Ahmed, J.U. and Daw, H.T., 1980. "Cost-Benefit Analysis and Radiation Protection." International Atomic Energy Agency Bulletin, Vol. 22, No. 5/6, pp. 13-22, October 1980.

American Cancer Society Inc., 1984. "Cancer Facts and Figures. Revised - 2/84." American Cancer Society, New York.

Canadian Environmental Protection Service (EPS), 1981. "Socio-Economic Impact Analysis for Chlorobiphenyl Regulations No. 2 (Products) and Regulation No. 3 (Release)." Prepared by the Environmental Strategies Directorate, Environment Canada.

Chatterjee, R.M., Coady, J.R. and Wagstaff, K.P., 1982. "An Approach to the Exemption of Materials from Regulation as Radioactive Wastes." Atomic Energy Control Board Report INFO-0086.

Crouch, E.A.C. and Wilson, R., 1982. "Risk/Benefit Analysis." Ballinger Publishing Company, Cambridge.

Cunningham, G.H., 1982. "The De Minimis Concept From a Regulatory Standpoint."

Davis, J.P., 1981. "The Feasibility of Establishing a De Minimis Level of Radiation Dose and a Regulatory Cut-off Policy for Nuclear Regulation." Prepared by General Physics Corporation for Edison Electric Institute.

Dunster, J.H., 1982. "ALARA or De Minimis?" Health Physics, Vol. 43(3), p. 438.

Eisenbud, M., 1980. "The Concept of De Minimis Dose." Paper presented at the National Council of Radiation Protection Meeting, 2 April 1980.

Environmental Protection Agency, Office of Radiation Programs, 1976. "40 CFR 190 Environmental Radiation Protection Requirements for Normal Operation of Activities in the Uranium Fuel Cycle: Final Environmental Statement." EPA 520/4-76-016.

Environmental Protection Agency, 1977. "40 CFR Part 190 - Environmental Radiation Protection Standards for Nuclear Power Operations." Code of Federal Regulations.

Environmental Protection Agency, 1981. "40 CFR Part 440. Ore Mining and Dressing Point Source Category. Subpart E - Uranium, Radium and Vanadium Ores Subcategory." Code of Federal Regulations.

Environmental Protection Agency, 1983a. "Regulatory Impact Analysis of Environmental Standards for Uranium Mill Tailings at Active Sites." EPA 520/1-82-023, March 1983.

Environmental Protection Agency, 1983b. "40 CFR Part 192. Environmental Standards for Uranium and Thorium Mill Tailings at Licensed Commercial Processing Sites. Final Rule." October 1983. Code of Federal Regulations.

Evans, R.D., Harley, J.H., Jacobi, W., McLean, S., Mills, W.A. and Stewart, C.G., 1981. "Estimate of Risk from Environmental Exposure to Radon-222 and Its Decay Products." Nature, Vol. 290, pp. 98-100.

Federal Radiation Council, 1960. "Background Material for the Development of Radiation Protection Standards." Report No. 1. May 13, 1960. Reprinted by the U.S. Department of Health, Education and Welfare, May 1965.

INTERA Environmental Consultants Inc., 1983. "PABLM: A Computer Code to Compute Accumulated Radiation Doses from Radionuclides Transported to Aquatic and Terrestrial Pathways in the Biosphere." ONWI-446.

International Commission on Radiological Protection, 1959. "Report of the ICRP Committee II on Permissible Dose for Internal Radiation." ICRP Publication 2, Pergamon Press, New York.

International Commission on Radiological Protection, 1966. "Deposition and Retention Models for Internal Dosimetry of the Human Respiratory Tract." Health Physics, Vol. 12, pp. 173-207.

International Commission on Radiological Protection, 1971. "Recommendations of the International Commission on Radiological Protection." ICRP Publication 10A, Pergamon Press, New York.

International Commission on Radiological Protection, 1973. "Implications of Commission Recommendations That Doses Be Kept As Low As Readily Achievable." ICRP Publication 22, Pergamon Press, New York.

International Commission on Radiological Protection, 1977. "Recommendations of the International Commission on Radiological Protection." ICRP Publication 26, Pergamon Press, New York.

International Commission on Radiological Protection, 1979. "Limits for Intakes of Radionuclides by Workers." ICRP Publication 30, Parts 1 (1979), 2 (1980), 3 (1981) and Supplements, Pergamon Press, New York.

International Commission on Radiological Protection, 1981. "Limits for Inhalation of Radon Daughters by Workers." ICRP Publication 32, Pergamon Press, New York.

Kaplan, S. and Garrick, B.J., 1981. "On the Quantitative Definition of Risk." Risk Analysis Vol. 1, No. 1, pp. 11-27.

Luckey, T.D., 1980. "Hormesis with Ionizing Radiation." CRC Press, Boca Raton, Florida.

Marline Uranium Corporation and Union Carbide Corporation, 1983. "An Evaluation of Uranium Development in Pittsylvania County. (Vol. 1-9)." Prepared by Dravo Engineers Inc. October 15, 1983.

Marline Uranium Corporation and Umetco, 1984. "Technical Summary and 1984 Supplement with Supporting Technical Memoranda. Final Draft (Vol. 1-3)." Prepared by Dravo Engineers Inc., August 1984.

Moeller, D.W., McKone, T.E. and Evans, J.S., 1983. "Criteria for Dose Limits to the Public." in Environmental Radioactivity: Proceedings of the Nineteenth Annual Meeting of the National Council on Radiation Protection and Measurements. April 1983. National Council on Radiation Protection and Measurements, Bethesda, Md.

National Academy of Sciences, Committee on the Biological Effects of Ionizing Radiations, 1972. "The Effects on Populations of Exposure to Low Levels of Ionizing Radiation." National Research Council.

National Academy of Sciences, Committee on the Biological Effects of Ionizing Radiations, 1980. "The Effects on Populations of Exposure to Low Levels of Ionizing Radiation: 1980." National Academy Press.

National Council on Radiation Protection and Measurements, 1971. "Basic Radiation Protection Criteria." NCRP Report No. 39, National Council on Radiation Protection and Measurements, Washington, D.C.

National Council on Radiation Protection and Measurements, 1984a. "Exposures from the Uranium Series with Emphasis on Radon and its Daughters." NCRP Report No. 77, National Council on Radiation Protection and Measurements, Washington, D.C., March 1984.

National Council on Radiation Protection and Measurements, 1984b. "Evaluation of Occupational and Environmental Exposures to Radon and Radon Daughters in the United States." NCRP Report No. 78, National Council on Radiation Protection and Measurements, Washington, D.C., May 1984.

National Institute of Health, 1980. NIH Publication 80-2087.

Nuclear Regulatory Commission, 1976. "Cost Benefit Analysis for Radwaste Systems for Light-Water Cooled Nuclear Power Reactors." NRC Regulatory Guide 1.110.

Nuclear Regulatory Commission, 1977. "10 CFR 20. Standards for Protection Against Radiation." Code of Federal Regulations.

Nuclear Regulatory Commission, 1980. "Final Generic Environmental Impact Statement on Uranium Milling." Report NUREG-0706. Office of Nuclear Material Safety and Safeguards.

Oser, B.L., 1978. "Benefit/Risk: Whose? What? How Much?" Food Technology, Vol. 32, pp. 55-58.

Pochin, E.E., 1978. "Why be Quantitative About Radiation Risk Estimates." Lecture No. 2 in the L.S. Taylor Lecture Series in Radiation Protection and Measurement, National Council on Radiation Protection and Measurement, Washington, DC.

Rossi, H.H., 1980. "What are the Limits of ALARA?" Health Physics, Vol. 39, pp. 370-371.

Siddall, E., 1981. "Risk, Fear and Public Safety." Atomic Energy of Canada Limited Report AECL-7404, April 1981.

Sjoblom, G. (Director, Office of Radiation Program, EPA), 1982. As quoted in a private communication from L.D. Hamilton, BNL to T.F. Yosie, EPA. March 26, 1984.

Starr, C., and Whipple, C., 1980. "Risks of Risk Decisions." Science, Vol. 208, pp. 114-119.

Strenge, D.L. and Bander, T.J., 1981. "MILDOS - A Computer Program for Calculating Environmental Radiation Doses from Uranium Recovery Operations." NUREG/CR-2011, PNL-3767. U.S. Nuclear Regulatory Commission, Washington, D.C.

United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR), 1977. "Sources and Effects of Ionizing Radiation." United Nations, New York.

United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR), 1982. "Ionizing Radiation: Sources and Biological Effects." United Nations, New York.

U.S. Department of Health and Human Services, 1979. "Vital Statistics of the U.S. 1978; Volume II, Mortality - Part A."

Voilleque, P.G., and Pavlick, R.A., 1982. "Societal Cost of Radiation Exposure." Health Physics, Vol. 43, No. 3, pp. 405-409.

Webb, G.A.M. and McLean, A.S., 1977. "Insignificant Levels of Dose: A Practical Suggestion for Decision Making." Prepared by the National Radiological Protection Board, Report R62, April 1977.

Wilson, R., 1979. "Analyzing the Daily Risks of Life." Technology Review, February 1979.