Environmental Radiation Dose Commitment: An Application To The Nuclear Power Industry

ENVIRONMENTAL PROTECTION AGENCY
Office of Radiation Programs

ENVIRONMENTAL RADIATION DOSE COMMITMENT: AN APPLICATION TO THE NUCLEAR POWER INDUSTRY

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FOREWORD

All levels of government and industry are faced with the challenge of responding affirmatively and fairly to the reality of public demands for action to improve the quality of the environment and the quality of life. These demands come from a new general awareness of the degradation we have inflicted on our surroundings and from a fear we may destroy ourselves if degrading trends are not reversed. Concern for quality of life also stems from an increasing emphasis on human and social values, and it is clear that a quality of life ethic has become deeply ingrained in our society and that a growing demand for maintaining and improving the quality of the environment is a principal component of that ethic.

The Office of Radiation Programs presents this report in the hope that the concept of "environmental dose commitment" will serve as a useful tool to assist evaluation of the potential environmental impact of alternative energy sources. It should be noted, however, that such discussion is beyond the scope of the present report. A comprehensive analyses of alternative energy sources will require, in addition to assessments of the impacts of the variety of environmental releases associated with normal operations (such as the long-lived radionuclides addressed here); assessments of the impacts due to mining and waste disposal, and of safety - both public and occupational. Comments on

this report are welcomed. These should be sent to the Director, Criteria and Standards Division of the Office of Radiation Programs (HM-560).

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PREFACE

An analysis of the consequences of the discharge and dispersal of long-lived radionuclides into the general environment is one of a variety of considerations required in the setting of standards for radiation protection. By virtue of the long persistence of these materials these consequences may extend over many generations and, in this respect, these discharges can represent irreversible public health commitments.

We have developed the concept of "environmental dose commitment" to encompass the radiation doses to populations implied by this irreversibility, extended it to include the calculation of resultant potential adverse health effects, and applied it to the specific case of the potential consequences of the next 50 years of normal operations of the United States nuclear power industry. Only the potential impact of the release of four types of long-lived radionuclides, namely tritium, krypton-85, iodine-129, and the actinides has been considered, and therefore the report does not purport to provide an evaluation of the overall impact of the industry. In addition, although these radionuclides have half-lives ranging from a decade to millions of years and can be projected to migrate over large areas, on the basis of present knowledge we cannot meaningfully project their persistence in the biosphere for periods much longer than a number of decades.

Therefore the potential consequences on health have been calculated only for the first 100-year period following release.

Two different viewpoints for preparing estimates of these potential future consequences have been utilized in the analysis. The first attempts to make an assessment of potential consequences giving due allowance for expected performance of current emission controls. The second attempts to establish estimates of upper limits of potential adverse consequences that are useful for public health and safety planning purposes, such as in assessing the adequacy of the margin of safety provided by the controls assumed in the first viewpoint. Obviously, these numerical estimates of projected impact are subject to considerable uncertainty; this is due both to the variability associated with all projections and the currently indeterminate character of some of the important parameters in the analysis. Expanded research efforts to better define the possible environmental pathways and health impact of these radionuclides are needed.

The report projects, by the end of the 50-year period considered, upper estimates for some of the radionuclides considered of as many as 5,000 to 25,000 committed potential health effects over the succeeding 100 years. To provide a perspective any such potential health impact can be viewed in the light of the many-fold greater number of health effects attributable to natural background radiation. The National Academy of Sciences, in its report entitled "The Effects on Populations of Exposure to Low Levels of Ionizing Radiation," has given a most likely estimate of approximately 3,000 to 4,000 cancer deaths annually

as attributable to levels of natural background radiation in the U.S. This is equivalent to roughly one percent of the spontaneous cancer deaths per year. Effects attributable to natural background radiation exposures are estimated on an annual basis and therefore for any comparison to the projections made in the report the time period covered would have to be taken into account.

Unlike the situation with respect to natural background radiation exposures, however, most of the projected potential impact of long-lived radionuclides from the nuclear power industry can be avoided. The timely imposition of controls, which considers the environmental dose commitment concept, can minimize the potential effects attributable to release of these materials. It is concluded, therefore, that the overall environmental dose commitment resulting from the release of these long-lived radionuclides by normal operations of the United States nuclear power industry for the next 50 years can be relatively small. A summary of the major findings of the report will be found in section IV.

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ABSTRACT

The concept of environmental dose commitment is developed and illustrated by application to projected releases of selected radionuclides from the nuclear power industry over the next fifty years. The concept encompasses the total projected radiation dose to populations committed by the irreversible release of long-lived radionuclides to the environment, and forms a basis for estimating the total potential consequences on public health of such environmental releases. Because of the difficulty of making projections of radionuclide transport on the basis of present knowledge, these potential consequences have been calculated only for the first one hundred-year period following release. The particular radionuclides considered are tritium, krypton-85, iodine-129, and the actinides.

ENVIRONMENTAL RADIATION DOSE COMMITMENT:

AN APPLICATION TO THE NUCLEAR POWER INDUSTRY

I. INTRODUCTION

In recent years mankind has become aware that decisions made to achieve short-term gains must take into account their impact on future generations. Contamination of the environment due to the use of such materials as pesticides, mercury, lead, and a variety of other toxic substances which persist for long periods of time is well known. These substances, even though discharged at low rates, can over a period of years gradually build up to undesirable levels. Since there are usually no practical methods to remove these materials from the environment, their introduction represents, in fact, an irreversible commitment ameliorated only by natural decomposition or occlusion.

Current and projected technologies for the utilization of nuclear energy introduce a variety of radioactive materials to the environment. Most of these materials are short-lived, due to radioactive decay, and have their primary impact near the sources of their discharge. A number, however, are long-lived and represent a long-term potential source of exposure of a large number of people. In general, no methods are available to effectively remove such materials from the environment once they have been released, and such releases thus imply irreversible commitments for exposure of future generations, except for natural occlusion in environmental sinks. In cases where these materials have physical or chemical properties which allow their widespread dispersal

through one or more environmental media, the impact of these commitments may be significantly enhanced.

The current contamination of the general environment by nuclear weapons fallout from tests conducted in the 1950's and early 1960's is a prime example of general environmental contamination by radioactive materials that is now irreversible. This source of radiation is worldwide and, next to natural background and medical exposure, it is the largest component of man's current radiation exposure. The recognition that fallout represents a general population risk (not primarily an individual one) and the associated public reaction which occurred were strong factors in the movement leading to cessation of atmospheric nuclear weapons tests by the major world powers in 1962.

It is generally accepted that no threshold can be assumed for health effects due to radiation exposure; therefore, the perspective that must govern discharges of these materials to the environment is that all doses which accrue to exposed populations result in some increment of risk to these populations. The perspective that all radiation dose results in some risk to the individuals exposed, or to their progeny, plus the fact that projected large-scale use of nuclear energy will produce large quantities of long-lived radionuclides, some of which may be discharged to the environment, make it especially important to consider the consequences of irreversible commitment of these discharges to the environment before they have occurred.

This paper develops general concepts for calculating the cumulative consequences of release to the environment of such long-lived

radionuclides and illustrates these with an analysis of the potential environmental consequences of projected releases of certain of these long-lived radionuclides associated with operations of the nuclear power industry for the next 50 years. These consequences are developed through calculation of the entire commitment of doses (i.e., dose equivalent) to populations implied by an environmental release, a quantity defined here as the "environmental dose commitment." Projections of this type are particularly important because the impact of these releases on populations continues over a long period of time. Since control must be instituted long before the impacts associated with these releases occur, projection of anticipated potential health effects which could result from the release of these radionuclides constitutes a necessary basis for decisions concerning the need for institution of control over their release.

Future decisions ought to consider these dose commitments with respect to both the types of development that should occur and the choice of controls that should be imposed. This analysis attempts to develop some of the factors associated with the perspective provided by environmental dose commitment and to illustrate how it may provide results which apply to the nuclear power industry as it embarks upon an anticipated period of accelerated growth over the next several decades.

II. ENVIRONMENTAL DOSE COMMITMENT - GENERAL CONSIDERATIONS

The impact of radioactive effluents on man can be considered from three different perspectives. The first of these is in terms of the maximum dose to individuals. This measure has been the one traditionally used for radiation impact analyses, and existing radiation protection guidelines are usually expressed as limits on annual doses to individuals. Although this approach may be adequate when the primary objective is to limit risk to specific individuals, as in the case of occupational limits for radiation workers, it is not adequate for use in limiting the impact of long-lived radioactive effluents on large populations. These materials typically deliver exposures over many generations and the exposure of specific individuals is usually very small. Doses received by members of the public due to the radionuclides considered later in this report are in general several orders of magnitude below existing Federal Guides limiting annual individual doses. The impact of such materials can be large, not because there are substantial risks to specific individuals, but because there are substantial numbers of people at low levels of risk, and because the potential for exposure may persist for a substantial period of time.

A second perspective is provided by summing the individual doses to each of the members of a population to obtain an index of the total population impact. This sum is generally expressed in person-rems, and is commonly estimated by forming the product of the total number of persons exposed and their average dose. This population dose is usually expressed on an annual basis.

A third perspective of the environmental impact of radioactive effluents includes the additional impact in subsequent years due to the buildup and persistence of long-lived radionuclides. This perspective is termed the "environmental dose commitment" and, simply defined, is the sum of all doses to individuals over the entire time period the material persists in the environment in a state available for interaction with humans. The unit of measure for this total population dose is person-rems of environmental dose commitment. It is calculated for a specific release at a specific time and is obtained by summing the person-rems delivered in each of the years following release to the environment until dose increments are inconsequential as the result of either radioactive decay or removal from the biosphere by other means.

The impact of this dose commitment can be expressed in terms of cumulative potential health effects. The terminology "cumulative potential health effects" is used here to describe the sum of projected deaths and diseases, including birth defects, that may be attributable to environmental releases from a given radiation source over a specified time period. The qualifying adjective "potential" is added to emphasize that the incidence of specific effects is based on extrapolations from information derived at higher levels of dose than those actually expected, using a linear, non-threshold dose-effect assumption. In addition, these effects will not be demonstrable since they are distributed on a statistical basis throughout the entire exposed population and are not different in kind from health effects occurring from other causes. Health effects are here defined as radiation-induced

somatic effects such as lung, thyroid, or skin cancers, plus certain serious genetic effects in future generations. Relevant aspects of health effect considerations are discussed in appendix D.

The idea of dose commitment is inherent in the internal dose models of the International Commission of Radiological Protection used to compute the maximum dose an individual can receive from internally deposited radioactive materials which have long physical and biological half-lives. Doses arising from radium and strontium deposition in bone are examples of this application of the concept of dose commitment. In addition, the United Nation's Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) has discussed basic concepts which pertain to calculating the dose committed by long-term exposures due to environmental contamination by radionuclides, but these calculations focus on the maximum potential individual dose rather than on the total impact of a given release on populations over extended periods of time. The concept of environmental dose commitment developed here extends these concepts to incorporate the total population dose implied by the environmental release of a radionuclide. Even though many of the principles involved have been previously enunciated, no significant application of the concept of dose commitment to evaluations of the total impact on populations of environmental releases of radionuclides appears to have been previously reported.

The determination of doses committed by the release of a radioactive material to the environment involves a multiplicity of factors. Environmental dose commitment, however, is particularly dependent upon

the radioactive half-life of the nuclide under consideration, as it will determine the availability for widespread dispersion in the environment and hence the total number of persons potentially exposed over both space and time. For the short-lived radionuclides, the environmental dose commitment will usually consist only of the short-term exposure of a limited population group. There is effectively no environmental buildup of these radionuclides because the actual amount available at any time represents a balance between incremental additions and incremental removals by radioactive decay (i.e., equilibrium) that is achieved in a short span of time. For long-lived radionuclides such an equilibrium condition will not be reached for many generations. These radionuclides continue to accumulate in the environment and, even if all further additions are stopped at some point in time, will persist for extended time spans as a potential source of cumulative exposures to successive generations. For some radionuclides this time period may be of the order of tens of thousands or even millions of years. In theory, their total impact should be evaluated over this entire time period. In practice, it is difficult, if not impossible, to make predictions over such extremely long time periods and some reasonable cutoff must be used. For this analysis this cutoff has been arbitrarily chosen as 100 years. This time span includes very nearly the entire potential impact of radionuclides with half-lives of the order of 10 years (such as tritium and krypton-85) and it provides at least an evaluation of the impact over a defined time period for the much longer-lived radionuclides such as plutonium-239 and iodine-129.

A schematic mathematical representation of the cumulative population dose resulting from a given environmental release, i.e., the "environmental dose commitment," is given by:

$$D_i(t) = \sum_{n=0}^{\infty} Q_i(t) e^{-\lambda_i n} T_i(n) F_i P(t+n) ,$$

where $D_i(t)$ = cumulative population dose resulting from the release of radionuclide i in calendar year t.

$Q_i(t)$ = quantity of radionuclide i released in the year t.

λ_i = radioactive decay factor for radionuclide i.

$T_i(n)$ = pathway model conversion factor relating quantity of radionuclide i released to its concentration in the medium at the location of interest n years following release.

F_i = dosimetry conversion factor relating concentration of radionuclide in the medium to resultant dose to individuals exposed.

$P(t+n)$ = number of persons exposed in calendar year $(t + n)$.

t = calendar year of release.

n = number of years from year t.

This illustrative expression is necessarily simplified. In real applications, the complications introduced by the multiplicity of environmental pathways, differences in doses to various organs and the spatial dependence of both the pathway model and population must be

considered explicitly. This expression applies specifically to the situation in which exposure of all individuals is uniform and the dose of interest is to the whole body, or to one organ, and from one medium only. Appendix A contains more detailed general equations which consider the complications introduced by the above and other factors.

In converting from environmental dose commitments to health effects it is necessary to define the particular types of health effects to be considered and the probabilities that they will be incurred as a function of the dose delivered. The analyses in this report are limited, as described earlier, to estimates of cancers and certain serious genetic effects, and the risk coefficients used have, in general, been derived from the recent (November, 1972) report of the Advisory Committee on the Biological Effects of Ionizing Radiations of the National Academy of Sciences - National Research Council, entitled "The Effects on Populations of Exposure to Low Levels of Ionizing Radiation." Although in the future it may become possible to quantify some of the less serious effects of radiation exposure, it is not believed that this will substantially modify the inferences for health impacts derived on the bases used here.

III. APPLICATION TO SELECTED LONG-LIVED RADIONUCLIDES FROM THE NUCLEAR POWER INDUSTRY

The concept of environmental dose commitment is applied to releases of long-lived radionuclides from the nuclear power industry in order to illustrate the use of the concept for some specific cases of environmental releases. The nuclear power industry is only one of several possible sources of long-lived radionuclides. Others include testing and other applications of nuclear devices, space power supplies, and some medical and industrial applications. Because of the projected rapid growth of the nuclear power industry, releases resulting from the normal operations of the industry have been selected as the source for this calculation. Releases associated with normal operations are defined here to include all routine releases plus those unplanned releases resulting from minor accidents, such as equipment malfunction and human error, which can be expected on a recurring basis.

A. General Considerations

Two different viewpoints for preparing estimates of future consequences have been utilized in this analysis. The first of these attempts to make an assessment of potential consequences with due allowance for expected performance of current emission controls, with the objective of placing the impact of projected releases of a specific radionuclide in a realistic perspective. The second viewpoint attempts to establish plausible estimates of maximum potential consequences under conditions of less effective control and more adverse environmental behavior. Such a viewpoint is most useful for public health and safety

planning purposes. These projected maximum potential consequences are not actually expected to occur. Indeed, it is the purpose of such projections to develop a rational basis for the control measures that would prevent their occurrence.

To assess the magnitude of the potential environmental impact of the release of long-lived radionuclides, four different isotopes (or groups of related isotopes) have been chosen for this study. This selection was based primarily on estimated total public health impact. The radionuclides considered are tritium, krypton-85, iodine-129, and the actinides including plutonium-238, 239, 240, and 241, americium-241, and curium-242 and 244. A number of other long-lived radionuclides, such as strontium-90 and cesium-137, are also produced in substantial quantities by the nuclear power industry and their total impact could conceivably be significant compared to that of the illustrative radionuclides discussed here.

The general considerations included in making these assessments are outlined in figure 1. Quantities of the various radionuclides produced were derived from projections of United States nuclear power production through the year 2020 beginning with the year 1970 and are discussed in appendix B. Obviously, growth estimates of nuclear power are subject to differences of opinion and the results of these assessments will be proportional to the projections used. This appendix also describes projections used in this report for United States and world populations. Assumptions concerning control technology and environmental pathways are described in appendix C. Release of all of the radionuclides considered

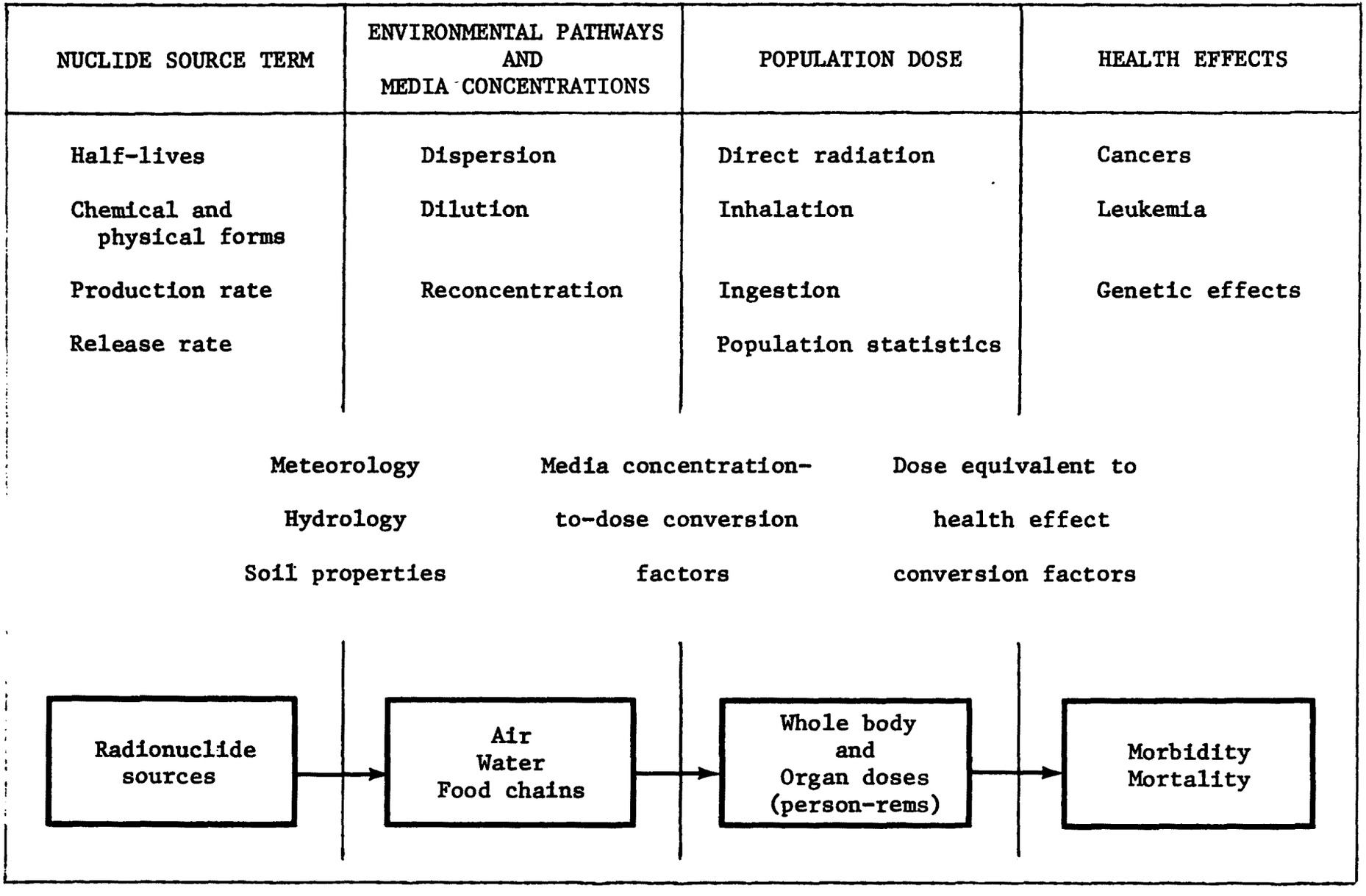

Figure 1 Model for estimating health effects from the nuclear power industry

is assumed to lead to initial short-term exposures of individuals within an 80 km (50 miles) radius. Iodine-129 and the actinides are then assumed to be ultimately uniformly distributed over, and confined to, large portions of the continental United States. Krypton-85 and tritium are assumed to be ultimately dispersed over the entire world. The concentration of radioactivity in food, water, air and other materials was converted to population dose and then to health effects. The conversion factors used are described in appendix D.

The calculated consequences of the release of the long-lived radionuclides are critically dependent on the assumptions made. A wide range of possible values for input parameters exists in addition to the normal uncertainties inherent in making any future projections. The estimated range of possible values for certain parameters and the actual values used in this report are shown in table 1. The rationale for the selection of these values is discussed below in section B. This table lists only the parameters for which different values were assumed in computing the dose commitments for each of the two viewpoints considered in this report. Using these parameters, the series of environmental dose commitments resulting from each of the annual releases attributable to operations of the nuclear power industry over the period 1970-2020 was calculated. Using these results, the cumulative numbers of health effects attributable to releases of each radionuclide through any given year were estimated. These effects represent the potential irreversible commitment due to releases through any given year, even if all future releases should cease at that point. The calculated environmental dose

Table 1

Numerical values for some significant parameters used in the analysis

Parameter	Range of possible values	Values used in study	
		Expected minimum performance	Public health planning
Actinide release fraction------	10^{-6} - 10^{-9}	10^{-7}	10^{-6}
Actinide resuspension (m^{-1})----	10^{-5} - 10^{-11}	10^{-8}	10^{-6}
Iodine-129 release fraction----	10^{-1} - 10^{-4}	10^{-3}	10^{-1}
Krypton-85 release fraction----	10^{0} - 10^{-3}	10^{0}	10^{0}
Tritium release fraction-------	10^{0} - 10^{-2}	10^{0}	10^{0}

commitments take into account all of the exposures occurring during the period of persistence of krypton-85 (half-life 10.7 years) and of tritium (half-life 12.3 years), but include only that fraction of the total environmental dose commitment occuring during the initial 100 years after release for iodine-129 (half-life 1.7×10^7 years) and the actinides (half-lives ranging up to 2.4×10^4 years).

B. Numerical Values of Key Parameters for Specific Radionuclides

Calculations of environmental dose commitments attributed to releases from operations incorporating emission controls are highly dependent upon the parameters chosen to characterize the effectiveness of these controls. In addition, certain of the radionuclides considered here undergo environmental transport processes which are not yet quantitatively well-defined. The existence of large uncertainties in these release and transport properties led to the choice of two viewpoints to estimate the potential impact of these radionuclides. The factors involved in the choice of numerical values for these key parameters for the two viewpoints are discussed below for each radionuclide considered.

1. Actinides:

Relatively small quantities of plutonium and the actinides have been produced by the nuclear power industry to date, and projections of future releases are subject to considerable uncertainties. Thus, it should be recognized initially that these estimates should be reviewed and revised as additional information is developed.

Current control practices for actinide releases at a single operation, such as nuclear fuel chemical reprocessing, are expected to restrict releases to the order of 10^{-8} to 10^{-9} of the total amount processed, and future experience may justify the assumption of even smaller release fractions. However, when allowance is made for inclusion of cumulative releases from the variety of fuel processing operations as well as transportation and handling throughout the entire fuel cycle, the fractional loss of plutonium and the actinides to the environment for the entire fuel cycle must be assumed to be greater than that from a single operation. In this context the fractional release of the actinides is not realistically expected to exceed 10^{-7} of the total amount handled in any given year. This value was used for projecting expected minimum performance of the industry. For public health planning purposes a more conservative viewpoint was adopted; a release fraction ten times greater was used.

Transport pathways for the actinides through the environment to man are not well defined. In this analysis the only pathway to man has been assumed to be inhalation of aerosol particles initially suspended in air and subsequently resuspended in the atmosphere after initial deposition. Additionally, the simplifying assumption was made that the fraction released was uniformly distributed over the continental United States. In view of the large uncertainties associated with the estimation of other factors in the analysis, these assumptions are not considered to represent a serious deficiency.

Resuspension - the ratio of material per unit volume of air to that per unit surface area of soil - is critically dependent upon a variety of parameters, including the depth profile of the deposited material, the size distribution of aerosol particulates, and especially upon local variations of topography, surface vegetation, and wind velocities. Resuspension factors ranging from a low of 10^{-8} per meter to a high of about 10^{-3} per meter have been reported for newly deposited plutonium, with most values clustered in the region of 10^{-5}. For undisturbed areas considerably lower values, ranging from 10^{-7} to 10^{-11} generally seem to apply for time periods ranging from a few months to several years after deposition. A value of about 10^{-9} appears to be a reasonable estimate of the average availability of plutonium deposited on soil for relatively long periods of time at the Nevada Test Site.

The difference between newly deposited sites and undisturbed areas is probably due to downward migration of these compounds through the soil, which depletes the quantity near the surface, rather than to a fundamental change in the physical characteristics of resuspension. It is expected that actual resuspension at specific locations decreases with time, and that migration through most soils represents a pseudo-sink for the actinides. Thus, the long-term impact of inhaled actinides may be overestimated by assuming a constant value for resuspension. However, adequate data are presently lacking, and a time-averaged resuspension factor appears appropriate for use at the present state of knowledge. Because of the availability of data for only a few isolated sites and the known somewhat greater tendency for larger resuspension of

particulates in populated urban areas, a numerical value of 10^{-8} was chosen as an appropriate numerical national average for the resuspension factor for purposes of estimating the anticipated impact of the industry. For public health planning purposes a more conservative (pessimistic) value of 10^{-6} was selected.

It must also be recognized that the uncertainties associated with human uptake pathways for the actinides give only limited validity to the applicability of the single pathway model assumed for these estimates and that this model must be updated as more information becomes available.

2. Iodine:

Releases of iodine-129 by the nuclear power industry can be expected to be almost exclusively restricted to fuel reprocessing facilities. Nearly all of the iodine produced by fission in the reactor is released when the fuel cladding barrier is destroyed and the spent fuel is dissolved for fuel reprocessing. Iodine control technology is becoming available which appears to be capable of restricting releases to the environment to the order of 10^{-3} to 10^{-4} of the total amount present in the fuel. If this degree of control is achieved a release fraction of 10^{-3} will be an achievable objective for the industry, even if allowance is made for possible additional losses in waste handling programs for this material. However this control equipment is designed primarily to control iodine-131, which has a much shorter half-life than iodine-129. For public health planning purposes a more pessimistic view of the

performance of this technology for iodine-129 was assumed; a release factor of 10^{-1} was used.

The pathway model for iodine-129 in the environment used in this analysis is subject to considerable uncertainty. Uniform distribution of iodine-129 over the entire eastern land area of the United States is an idealized concept, but this probably does not introduce significant error when used in the evaluation of health effects in national populations. However, the migration of iodine-129 in the environment, its pathways to man, and its ultimate disposition are not yet well established.

3. Krypton and Tritium:

Assumptions relative to the quantities of krypton-85 and tritium released and of possible environmental pathways are subject to considerably less uncertainty than those for plutonium and iodine-129. Krypton is an inert gas and tritium is found in the environment principally in the form of tritiated water (i.e., HTO). Control methods which have decontamination factors of 10^3 and 10^2 for krypton and tritium, respectively, have been proposed, but none is currently in use in the nuclear power industry. Therefore, release of all krypton-85 and tritium produced has been assumed.

Environmental pathway models for tritium are subject to somewhat larger uncertainties than those for krypton. Krypton-85 is assumed to expose local and regional populations via a finite cloud model and then to mix uniformly in the world's atmosphere. Pathway assumptions for tritium are somewhat less well established, and assumptions concerning

the initial regional dispersion of tritium prior to its entering the hydrological cycle are not well documented. The model chosen assumes rainout over the eastern United States of half of the tritium released followed by dilution and recirculation in the world's hydrological cycle of the entire amount released.

C. Expected Minimum Performance By Industry--A First Viewpoint

Table 2 lists the projected numbers of health effects resulting from projected releases from the United States nuclear power industry over the next 50 years; these are based on the parameters shown in table 1 for this viewpoint. These projections assume that presently anticipated performance of controls currently in use will obtain in the future. Releases of krypton-85 and tritium are currently not controlled. The imposition of controls which would reduce the fraction of radionuclides released to the environment would decrease the environmental dose commitment proportionately. However, this decrease would affect only the additional number of health effects attributable to the releases prevented.

The following is an example of how to use the data contained in table 2. If the present absence of control is assumed to continue and all krypton-85 produced through the year 2000 is released to the environment, the calculations indicate that an estimated 230 health effects (on a worldwide basis) will be committed by krypton-85 doses received prior to the end of the year 2000, and that an estimated additional 760 health effects will be caused by doses delivered after the year 2000 by krypton-85 remaining in the environment from all

Table 2

Projected numbers of health effects attributable to release of certain long-lived radionuclides by normal operation of the nuclear power industry (estimated for anticipated minimum performance by industry assuming current release practices)

Year(t)	Cumulative number of health effects							
	Iodine-129		Tritium		Krypton-85		Actinides	
	Past-present[a]	Future[b]	Past-present[a]	Future[b]	Past-present[a]	Future[b]	Past-present[a]	Future[b]
1970----------	0	0	0	0	0	0	0	0
1975----------	0	0	2	0.5	0.3	5	0	0
1980----------	0	0	11	3	3	26	0	0.1
1985----------	0	0	35	8	14	79	0	0.4
1990----------	0	0.1	88	21	42	190	0.1	1
1995----------	0	0.2	190	43	110	410	0.2	2
2000----------	0.1	0.3	360	81	230	760	0.4	4
2005----------	0.2	0.5	630	140	460	1,300	0.7	7
2010----------	0.3	0.8	1,000	230	830	2,100	1.2	10
2015----------	0.5	1.2	1,600	340	1,400	3,200	2	15
2020----------	0.8	1.7	2,300	500	2,300	4,600	3	21
	one-fourth fatal		two-thirds fatal		two-thirds fatal		all fatal	

[a]The number of health effects committed from doses received through year(t).

[b]The number of health effects committed from doses delivered after year(t) by radionuclide releases up through year(t) only.

releases prior to the end of the year 2000. Operations of the U.S. nuclear power industry through the year 2020 could result in a total worldwide population impact (i.e., cumulative potential health effects) of about 7,000 health effects attributable to the release of krypton-85 and as many as 10,000 health effects due to all the radionuclides considered here. This number is derived by summing all entries for the year 2020. An obvious conclusion from the results is that, under the conditions assumed for this part of the analysis, krypton and tritium are the radionuclides of major concern for the 100-year period considered.

D. Public Health Planning Projections -- A Second Viewpoint

The importance of developing projections for public health planning purposes is to gain a perspective of the maximum plausible impact man's activities may have on the total quality of life. Predictions made for this purpose must necessarily adopt a more conservative (i.e., pessimistic) perspective, especially for activities which may result in an irreversible deterioration of the environment and on which controls must be imposed long before an unacceptable level of impact is reached. The expectation is not that these conservative estimates will come to fruition, but rather that constant vigilance and effective application of technology can and must be utilized to prevent these estimates from being realized.

The values of input parameters which were used for these planning projections are shown in table 1. The choice of this set of numerical values resulted from a series of judgments at least as difficult to make

as those for the previous projections for expected minimum industry performance. The general approach taken in choosing the values of parameters for these second viewpoint projections was to attempt to avoid the use of worst case assumptions simultaneously for all of the variables. Calculated impacts resulting from the parameters used for this second viewpoint are listed in table 3 and displayed in figure 2. Results from the first viewpoint are also displayed in figure 2, for comparative purposes.

The most significant result of the public health planning viewpoint is the relatively large number of health effects attributable to releases of the actinides. Under these assumptions, by the year 2020 an additional commitment of 24,000 health effects is projected for normal operations of the United States nuclear power industry. The bulk of these effects are evenly distributed over the 100-year period following release for which the environmental dose commitment was calculated. Two points should be emphasized. First, the number of effects calculated is based on a highly conservative (pessimistic) set of assumptions and is expected to overestimate the actual impact of such releases over the 100-year period chosen for this analysis. Second, the actinides are, in general, very long-lived materials and their eventual total impact over many centuries may be many times that experienced during the first 100 years following release. Current knowledge does not permit estimation over such long time periods.

Until existing uncertainties in these projections are resolved, concern for protection of public health dictates that such estimates as

Table 3

Projected numbers of health effects attributable to release of certain long-lived radionuclides by normal operation of the nuclear power industry (estimated as maximum plausible projections for purposes of planning for adequate public health and safety considerations)

Year(t)	Cumulative number of health effects							
	Iodine-129		Tritium		Krypton-85		Actinides	
	Past-present[a]	Future[b]	Past-present[a]	Future[b]	Past-present[a]	Future[b]	Past-present[a]	Future[b]
1970	0	0	0	0	0	0	0	0
1975	0	0	2	0.5	0.3	5	2	26
1980	0	1	11	3	3	26	12	140
1985	1	4	35	8	14	79	38	440
1990	3	9	88	21	42	190	96	1,100
1995	6	17	190	43	110	410	210	2,200
2000	11	32	360	81	230	760	400	3,900
2005	21	53	630	140	460	1,300	720	6,500
2010	34	82	1,000	230	830	2,100	1,200	10,000
2015	53	120	1,600	340	1,400	3,200	1,900	15,000
2020	78	170	2,300	500	2,300	4,600	2,800	21,000
	one-fourth fatal		two-thirds fatal		two-thirds fatal		all fatal	

[a]The number of health effects committed from doses received through year(t).

[b]The number of health effects committed from doses delivered after year(t) by radionuclide releases up through year(t) only.

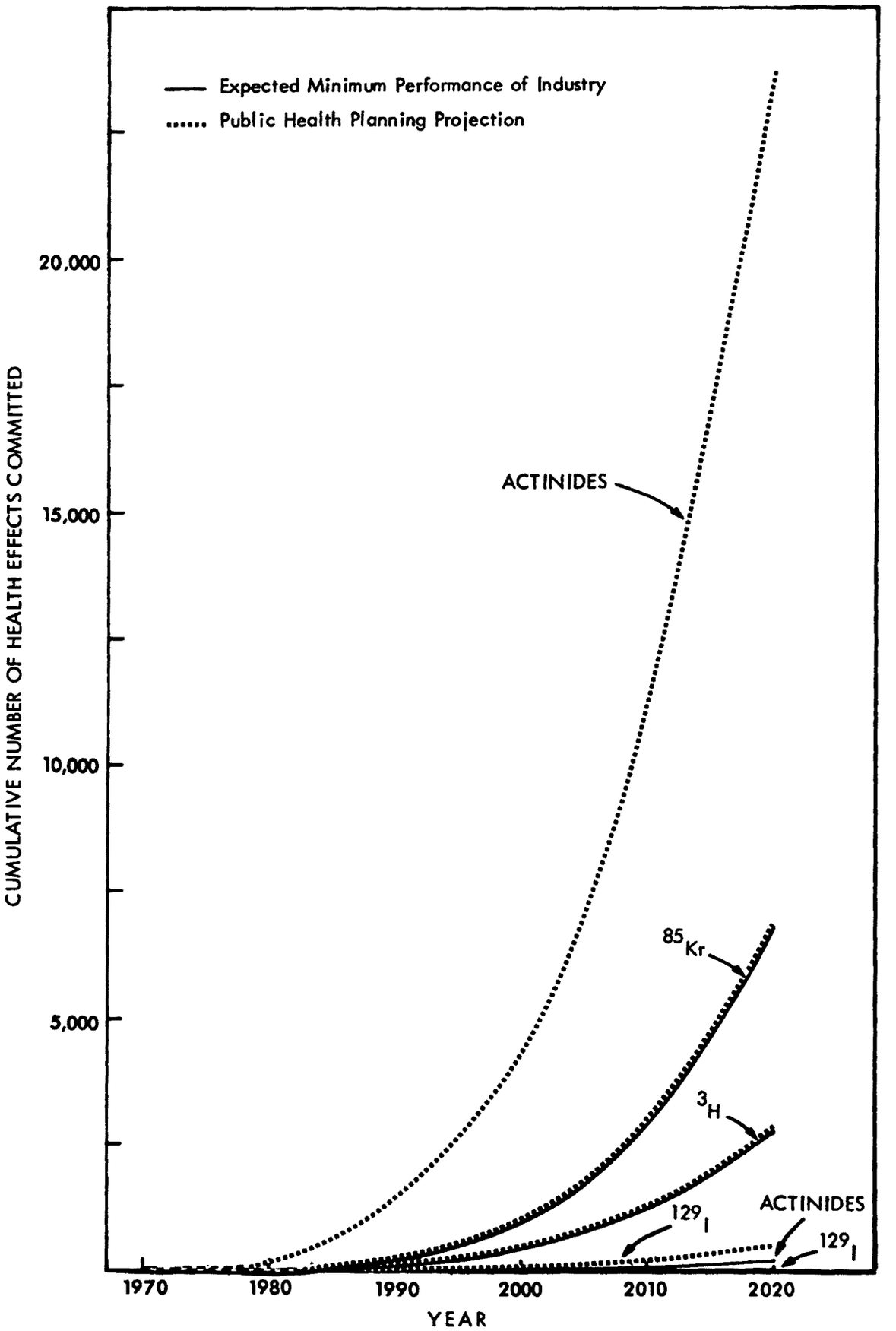

Figure 2. Estimated cumulative potential health effects committed by projected releases from the United States nuclear power industry

these of potential consequences be made in order to assure the public that uncertainties have been considered and that reasonable margins of protection will be provided. This approach is especially important in cases where current releases may be small but the cumulative potential impact of a rapidly expanding industry is significant. It is important to recognize that these large uncertainties exist because adequate data are not available, a condition that can be remedied only through additional research and monitoring efforts.

IV. SUMMARY AND CONCLUSIONS

This report has developed the concept of environmental dose commitment to examine the implications of the irreversible commitment of releases of long-lived radionuclides to the environment. It is found that a comprehensive assessment of the impact of such releases can be conducted within the analytical framework provided by this concept. An assessment of the impact of such materials on individuals or on local populations on the basis of annual exposure alone does not provide an adequate measure of the total impact of their release. The scope of analysis must include not only all members of the population initially exposed, but also all exposures during the entire time frame during which these radionuclides remain in the biosphere. Although not all of the projections for the radionuclides considered here satisfy both of these criteria completely, the perspective provided by even a partial calculation of environmental dose commitment is considerably more meaningful than such a traditional measure as annual individual dose.

The concept of environmental dose commitment was applied to the projected normal releases of several long-lived radionuclides over the next 50 years due to operations of the nuclear power industry in the United States. The results of the analysis are expressed in terms of numbers of potential health effects. Although the impact was evaluated on a worldwide basis, only the contribution of releases from the United States nuclear power program was considered. In view of uncertainties involved in projecting the impact of these releases, the results are presented both in terms of minimum expected industry performance as well

as in terms of estimates useful primarily for public health and safety planning purposes.

Application of the concept of environmental dose commitment leads to the conclusion that, in general, the impacts of projected future releases of the long-lived radionuclides considered here can be relatively small if appropriate and timely attention is given to their minimization.. Although these impact are particularly small for current levels of environmental releases, it is clear that future radiation guidance, standards, and regulations must address the implications of environmental dose commitments due to these materials. The perspective provided by this concept is essential in order to insure that proper attention is focused on minimizing the impact of man's rapidly expanding uses of these radioactive materials on future generations.

Numerical results of this study indicate two potential trends of significance. First, the potential future impact of the release of krypton-85, especially if other releases around the world are added to these estimates, is sufficiently large that active consideration should be given to controls to limit releases of this radionuclide. Second, the potential implications of release of the actinides are large. Additionally, the carrying out of this study made it clearly evident that there is a need for comprehensive research efforts to delineate release terms, environmental pathways, and biological effects of the radionuclides considered, and in particular for the actinides.

It should be recognized that any calculation of environmental dose commitment is subject to uncertainty. The projections presented here

are indicative only of current best estimates of possible consequences and cannot indicate more than potential future general trends. In order to reduce uncertainties and thereby be more useful for policy decisions, such analyses must be updated at frequent intervals as new information is developed. Fortunately, the vast majority of the adverse health effects estimated by application of the concept of environmental dose commitment are not yet committed. Because of this, and in spite of the inherent uncertainty of any projection, the concept of environmental dose commitment can provide a useful basis for dealing with the challenge of protecting the environment from avoidable and irreversible detriment.

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APPENDIX A

GENERAL EQUATIONS

FOR

ENVIRONMENTAL DOSE COMMITMENT

I. INTRODUCTION

The "environmental dose commitment" attributable to a release of radioactive material is defined as the sum of all doses to individuals over the entire time period the material persists in the environment in a state available for interaction with humans. It is calculated for a specific release occurring at a specific time. Environmental dose commitments are expressed in terms of the number of person-rems to the whole body or to specific body organs. The health impact of these commitments can be expressed in terms of the numbers of different kinds of health effects attributable to these doses by using a linear non-threshold dose-effect model. Finally, the total health impact of a release may be characterized by a single value derived by applying weighting factors to different categories of health effects and summing. In general, however, health impacts are most usefully expressed as separate categories of effect without such weighting.

In order to develop mathematical expressions for environmental dose commitment and their associated health impacts, a variety of terms must be coupled to the quantity (Q) of the radionuclide released to the environment. These are the pathway transfer function (T), dose conversion factor (F), population density (P), health effects conversion factor (R), and a weighting factor (W) that expresses the seriousness of a health effect. These terms may differ depending upon the radionuclide (i), pathway (j), body organ (k), type of health effect (l), and transfer medium (m) under consideration. In addition, the pathway

transfer function and population density will, in general, be functions of geographical location and time.

II. GENERAL EQUATIONS

To develop a general expression for environmental dose commitment we first consider an individual at some location, $\vec{r}$, away from the source of the release of a quantity, Q_i, of a radionuclide, i, that occurred at a specific calendar time, t_0. The dose rate, d_{ik}, that he is exposed to through various media, m (such as air, water, or food), will depend upon the pathways, j, of radionuclide i through the environment, including dilution and reconcentration mechanisms, radioactive decay, and intake and other biological factors which relate concentration in a medium to dose rate to a particular organ, k.

These relationships can be mathematically expressed as:

$$\dot{d}_{ik}(\vec{r},t_0,t) = Q_i(t_0) \sum_m \sum_j T_{ijm}(\vec{r},t)\ e^{-\lambda_i t}\ F_{imk}\ , \qquad (1)$$

where the first sum is over all media, and the second sum is over all possible pathways to the medium, m. The variable t is the time elapsed since release of the quantity $Q_i(t_0)$ at calendar time t_0, and the term $T_{ijm}(\vec{r},t)$ converts the quantity released to the concentration after a time t at the location $\vec{r}$ in medium m from the j^{th} pathway, and carries the units curies per unit volume per curie released. The pathway model, $T_{ijm}(\vec{r},t)$, must be considered a function of time as well as of location for two reasons: environmental sinks may have a time dependence quite apart from normal radioactive decay, and the model, if expressed as a function of

function of $\vec{r}$ only, can be multivalued due to recirculation in such environmental transport systems as the hydrological cycle or general atmospheric circulation. The exponential is the radioactive decay factor, where λ_i is the decay constant for radionuclide i. The concentration to which an individual is exposed through a particular medium is converted to dose rate to the whole body or to any organ or tissue of interest by the factor F_{imk}. We will use a generalized definition of organ that includes the whole body and skin as well as internal organs and tissues. For each specific nuclide it is, of course, necessary to determine which of these "organs" are of significance. This factor generally has no time or location dependence, although it is possible that individual uptake from some media could vary throughout the year, or from one location to another. In those situations where buildup of body burdens of internal emitters can occur, it will also be necessary to reflect the sum of all future doses to the individual committed by each incremental body burden in calculating the factor F_{imk}.

Equation (1) expresses the basis required for the calculation of individual exposures. In order to extend the calculation to populations an additional factor, $P(\vec{r},t_0+t)$, the population density as a function of location and calendar time must be introduced. In general, a number of different characteristics of population subgroups must be considered in order to properly calculate the effects of radiation exposure. These may include age-specific variation in uptake and organ size as well as additional variations in radiosensitivity due to age or sex. In cases

involving long-term (greater than a few years) commitments for exposure, however, it will usually be possible to avoid separate calculation for each population subgroup by using a suitably weighted average of the dose conversion factors appropriate to each population subgroup for the factor F_{imk} to represent the average for the entire population exposed. Similarly, in converting from dose to health effects in equation (3) below, an analogously constructed conversion factor can easily accommodate variations in sensitivity due to age or sex. For simplicity, therefore, we have not specified subgroups of the exposed population, although the extension for special situations requiring it is straightforward.

The population dose committed by a release at time t_0 is now computed by integrating equation (1) times the population density function over space and time:

$$D_{ik}(\vec{r}_1, t_0, t_1) = \int_{t_0}^{t_0+t_1} \int_0^{\vec{r}_1} \dot{d}(\vec{r}, t_0, t)\ P(\vec{r}, t_0+t)\ d\vec{r}dt\ , \qquad (2)$$

where $\vec{r}_1$ and t_1 are the limits of geographical area and of time for which the population dose is being calculated. Ideally, the calculation of dose committed by a release should consider all locations at which individuals may receive exposures, and all time until the exponential decay factor in equation (1) reduces the integrand to insignificant values. In practice this is often either not possible or not practical. The detailed examples elsewhere in the report examine the question of appropriate choices of these limits for some specific radionuclides.

The quantity D_{ik} given by equation (2) is the required *environmental dose commitment* and is specified in *person-rems* of *population dose* to the whole body or to any organ, k, attributable to release of a quantity, $Q_i(t_0)$, of a particular radionuclide i to the environment. It provides in a single value, or index, the means for comparative assessments of the impact of such environmental releases. However, two additional operations are required to transform D_{ik} into a more useful measure of the consequences of an environmental release. The first is to estimate the various health consequences or the total impact of such a dose due to a single release, and the second is to project the cumulative consequences of all of the projected releases over some future time period from the particular activity under examination.

The health consequences of a radiation dose may range all the way from inconsequential to lethal. We will assume that it is possible to categorize health effects into groups having similar importance and probability of occurrence in the exposed population. If this can be done, then the number of such health effects, H_{il}, of a particular category, l, due to an exposure from the release Q_i can be expressed as:

$$H_{il} = \sum_k D_{ik} R_{kl} , \qquad (3)$$

where R_{kl} is the probability of incurring an effect of category l in the population due to an exposure of organ k. If the desired endpoint of the analysis is the number of a particular category of effects that may be induced, such as the number of lethal effects, the sum over those

quantities, $H_{i\ell}$, which are lethal expresses this endpoint. If, however, it is desired to express the total impact of the release by a single result, then it is necessary to pursue the calculation one step further by introducing a weighting factor which expresses the relative severity of the various categories of health effects, as follows:

$$I_i = \sum_{\ell} H_{i\ell} W_{\ell} \ , \tag{4}$$

where I_i is a single index representing the total impact of the release, Q_i, expressed in some convenient unit, such as dollars or days or life shortening or discomforture, and W_i is an appropriately constructed weighting factor for the ℓ^{th} category of health effects.

Environmental dose commitments are calculated in order to make assessments of operations to be conducted, usually, over an extended period of time. To calculate the total projected consequences of conducting such an operation it is necessary to determine the impact of cumulative releases from the operation over a specified period of interest. Since the population exposed will, in general, vary with calendar time, the calculation must be performed by considering Q_i in equation (1) as a variable which expresses the rate of release as a function of time. The calculation of the cumulative dose commitment is then easily accomplished by performing an additional integration over time in equation (2) as follows:

$$\mathbb{D}_{ik}(\vec{r}_1, t_0, t_1, t_2) = \int_{t_0}^{t_2} D_{ik}\{\dot{Q}_i(t'), \vec{r}_1, t_0, t_1\}\, dt' \ , \tag{5}$$

where the integrand is given by the right-hand side of equation (2), but the time dependence of the release rate, $\dot{Q}_i$, is shown explicitly and t_2 is the end of the period over which releases due to the operation under investigation are included. The quantity $\mathbb{D}_{ik}$ is defined as the *cumulative environmental dose commitment* and is specified in *person-rems of population dose*, to the organ of interest, attributable to environmental releases from a particular operation over a specified period of time. The examples derived below and discussed elsewhere in this report estimate health effects as a function of t_2 (the year to which the activity continues) resulting from such cumulative environmental dose commitments. These health effects and their related impacts can be calculated from $\mathbb{D}_{ik}(\vec{r}_1, t_0, t_1, t_2)$ in a manner analogous to that shown in equations (3) and (4) for $D_i(\vec{r}_1, t_0, t_1)$.

The above dose equations involve three time perameters: t_0, the year of initial release of a given radionuclide; t_1, the period over which the dose commitment of each release is accumulated; and t_2, the final year for which releases contributing to cumulative environmental dose commitment are included. For purposes of the calculations in this report, the parameter t_0 is 1970, t_1 equals 100 years, and t_2 varies from the year 1970 to 2020.

APPENDIX B

ANNUAL RADIONUCLIDE INVENTORIES

AND

POPULATION PROJECTIONS

I. INTRODUCTION

This appendix is concerned with two important factors utilized in the calculations presented in this report: (a) projected annual inventories of radionuclides of interest, and (b) population projections. These projections are based on information in the literature (see attached bibliography).

II. ANNUAL RADIONUCLIDE INVENTORIES

For purposes of this study the annual quantities of the radionuclides of interest potentially available for release to the environment are assumed to be those quantities present in spent reactor fuel reprocessed each year. Only the U.S. nuclear power industry was considered. The number of metric tons of fuel to be reprocessed in any given year was estimated by using data on power generated 2 years earlier and assuming a thermal efficiency of 0.35 and a burnup of 33 gigawatt-days (thermal) per metric ton of fuel:

$$\frac{\text{metric tons}}{\text{year(t)}} = \frac{\text{GW(e)}}{\text{year(t-2)}} \times \frac{1\ \text{GW(th)}}{0.35\ \text{GW(e)}} \times \frac{1\ \text{metric ton}}{33\ \text{GW(th) days}} \times \frac{365\ \text{days}}{\text{year}} \quad .$$

The estimated nuclear power generation and the metric tons of fuel to be reprocessed per year are given in table B.1 for the expected mix of reactor types.

There are two types of radioactive material present in spent reactor fuel: fission products and activation products including actinide isotopes. The quantities of specific radionuclides present are

Table B.1

Estimated U.S. nuclear power production
and fuel reprocessing requirements

Year	Nuclear electric generation, GW(e)	Metric tons of fuel to be reprocessed annually[a]				
		LWR-U	LWR-Pu	LMFBR	HTGR	TOTAL
1970	2.6	25	0	0	0	25
1975	40	700	90	0	0	790
1980	110	1900	500	0	0	2,400
1985	220	2700	2600	0	100	5,400
1990	420	3700	3800	480	2,420	10,400
1995	650	4100	4100	2,600	6,600	17,400
2000	1000	3700	3800	11,500	7,800	26,800
2005	1360	3700	3700	20,600	10,000	38,000
2010	1780	4300	4400	32,800	10,000	51,500
2015	2220	5300	5400	43,800	10,000	64,500
2020	2700	6100	6100	58,000	8,800	79,000

[a]A burnup of 33 GW-day/MT to discharge was assumed for all fuel. Fuel burnup is a highly variable parameter and the value chosen represents an estimated design average for normal operation of current light water reactors. Fast breeder reactors are expected to have a design average fuel burnup up to 100 GW-days/MT. The value chosen may thus greatly overestimate the fuel discharges in later years, and the numbers shown here should not be considered the actual expected numbers. The resultant radionuclide inventories derived from these calculations, as used in this report, however, are largely independent of the burnup assumed and the results derived in this analysis are only slightly affected by this assumption.

determined primarily by fuel type, amount of burnup, and time of cooling (time between removal from the reactor and time of reprocessing).

Tables B.2 and B.3 show quantities of the potentially significant fission product and activation radionuclides present in one metric ton of spent fuel with 33 GW(t) days burnup and 150 days cooling time. These values are considered reasonably representative of all nuclear types. There is some possibility that cooling times shorter than 150 days may be used in the future, since faster recycling of the recovered fuel may result in a significant economic benefit. This would greatly increase the amounts of shorter-lived radionuclides in the fuel and available for release, but would not significantly affect the long-lived fission product inventories.

The amounts of actinides estimated to be present in uranium fuels and in plutonium-recycle fuels are given in table B.4. It is assumed that all fuels (including those used in HTGR's) other than uranium-235 fuels can be considered equivalent to plutonium-recycle fuels.

Based on the amounts of spent fuel to be processed, and on the estimated quantities of radionuclides per metric ton of spent fuel, the projected annual quantities of several of the most significant radionuclides in processed fuel were calculated and are presented in table B.5.

The release fractions applied to these annual inventories to determine the estimated amounts released to the environment are discussed in detail in the text of this report. In the computations carried out in this study, it was assumed that environmental releases of

Table B.2

Representative quantities of potentially significant fission products in spent reactor fuels

Isotope	Half-life (years)	Curies per metric ton	Grams per metric ton	Release state	Notes
^{3}H	12.3	800	0.083	Gas	> 95% released as HTO
^{85}Kr	10.7	10,500	27	Gas	
^{99}Tc	2.13×10^{5}	15	880	Semivolatile	Oxide b.p. 200° C
^{103}Ru	0.11	180,000	5.7	Semivolatile	Tetroxide b.p. 80° C
^{106}Ru	1.01	820,000	240	Semivolatile	^{103m}Rh + ^{106}Rh daughters
^{125m}Te	0.16	6,500	0.36	Semivolatile	Oxide b.p. 750° C
^{127m}Te	0.30	25,000	2.7	Semivolatile	^{127}Te daughter
^{129m}Te	0.09	13,000	0.42	Semivolatile	^{129}Te daughter
^{129}I	17×10^{6}	0.04	250	Volatile	b.p. 184° C
^{131}I	0.02	2.0	< 0.01	Volatile	b.p. 184° C
^{134}Cs	2.05	100,000	77	Semivolatile	Oxide b.p. 750° C
^{135}Cs	3×10^{6}	1.2	1400	Semivolatile	
^{137}Cs	30.2	106,000	1200	Semivolatile	^{137m}Ba daughter
^{89}Sr	0.14	100,000	3.5	Solid	
^{90}Sr	28.9	60,000	430	Solid	^{90}Y daughter
^{91}Y	0.16	190,000	7.8	Solid	
^{93}Zr	0.95×10^{6}	2	490	Solid	
^{95}Zr	0.18	400,000	19	Solid	^{95m}Nb + ^{95}Nb daughters
^{95}Nb	0.10	800,000	21	Solid	
^{125}Sb	2.73	13,000	12	Solid	
^{141}Ce	0.09	80,000	2.8	Solid	
^{144}Ce	0.78	800,000	250	Solid	^{144}Pr + ^{144}Nd daughters
^{147}Pm	2.62	200,000	220	Solid	
^{155}Eu	5.0	40,000	87	Solid	

Burnup = 33 GWd(t)/MT Cooling time = 150 days

Table B.3

Representative quantities of potentially significant activation products in spent reactor fuels

Isotope	Half-life (years)	Curies per metric ton	Grams per metric ton	Release state
^{54}Mn-----	0.86	30,000	3.9	Solid
^{55}Fe-----	2.7	20,000	8.3	Solid
^{59}Fe-----	0.12	500	<.01	Solid
^{58}Co-----	0.20	30,000	1.0	Solid
^{60}Co-----	5.26	2,000	1.8	Solid

Burnup = 33 GWd(t)/MT Cooling time = 150 days

Table B.4

Representative quantities of actinides present in spent reactor fuels

Isotope	Half-life (years)	Uranium fuels		Pu-recycle fuels	
		Ci/MT	g/MT	Ci/MT	g/MT
^{235}U-----	710×10^6	< 1	8,000	< 1	3,000
^{236}U-----	24×10^6	< 1	4,000	< 1	1,500
^{238}U-----	4510×10^8	< 1	950,000	< 1	950,000
^{237}Np----	2×10^6	< 1	600	< 1	200
^{238}Pu----	86	4,000	230	6,000	340
^{239}Pu----	24,400	500	8,100	750	12,000
^{240}Pu----	6,580	650	2,900	1,000	4,400
^{241}Pu----	13	150,000	1,300	300,000	2,600
^{242}Pu----	379,000	2	510	5	1,300
^{241}Am----	458	750	230	2,000	620
^{243}Am----	7,800	20	100	200	1,000
^{242}Cm----	0.45	35,000	10	250,000	75
^{244}Cm----	17.6	2,000	25	25,000	300
Total (excluding uranium)		193,000	14,000	585,000	23,000

Burnup = 33 GWd(t)/MT Cooling time = 150 days

Table B.5

Estimated annual inventories of selected nuclides in reprocessed fuels[a]
(curies)

Year	Fuel reprocessed (MT)	Tritium	Krypton-85	Iodine-129	Plutonium-239	Plutonium-241
1970----------	25	2.0×10^{4}	2.6×10^{5}	1.0	1.9×10^{4}	7.5×10^{6}
1975----------	790	6.3×10^{5}	8.3×10^{6}	3.2×10^{1}	5.9×10^{5}	2.4×10^{8}
1980----------	2,400	1.9×10^{6}	2.5×10^{7}	9.6×10^{1}	1.8×10^{6}	7.2×10^{8}
1985----------	5,400	4.3×10^{6}	5.7×10^{7}	2.2×10^{2}	4.1×10^{6}	1.6×10^{9}
1990----------	10,400	8.3×10^{6}	1.1×10^{8}	4.2×10^{2}	7.8×10^{6}	3.1×10^{9}
1995----------	17,400	1.4×10^{7}	1.8×10^{8}	7.0×10^{2}	1.3×10^{7}	5.2×10^{9}
2000----------	26,800	2.1×10^{7}	2.8×10^{8}	1.1×10^{3}	2.0×10^{7}	8.0×10^{9}
2005----------	38,000	3.0×10^{7}	4.0×10^{8}	1.5×10^{3}	2.9×10^{7}	1.1×10^{10}
2010----------	51,500	4.1×10^{7}	5.4×10^{8}	2.1×10^{3}	3.9×10^{7}	1.5×10^{10}
2015----------	64,500	5.2×10^{7}	6.8×10^{8}	2.6×10^{3}	4.8×10^{7}	1.9×10^{10}
2020----------	79,000	6.3×10^{7}	8.3×10^{8}	3.2×10^{3}	5.9×10^{7}	2.3×10^{10}

[a]Based on Pu-recycle fuels and reactor type distribution in table A.1. (33 GWd(t)/MT burnup and 150 days cooling period.)

tritium, krypton-85 and iodine-129 occurred only from reprocessing plants. Releases of the actinides were assumed to occur from other stages of the fuel cycle as well as from reprocessing plants.

III. POPULATION PROJECTIONS

A. Regional

The regional population growth within 80 km (50 miles) of a plant was estimated from population growth projections for reactor sites given in environmental reports submitted to the AEC by electric power companies. These indicated a regional population doubling time of approximately 40 years. This value was also considered to be applicable to fuel reprocessing plants and other nuclear facilities.

B. United States

The population projection used for the United States is shown in figure B.1. This growth curve approximates the 1970 Series C projections of the Bureau of the Census with the added variation that the population will level off at 400 million. A straight line approximation to the curve was used to simplify calculations.

C. World

The world population growth was estimated from median values from a United Nations projection. The 1970 world population was estimated as 3.56×10^9 with an annual growth rate of 1.9 percent.

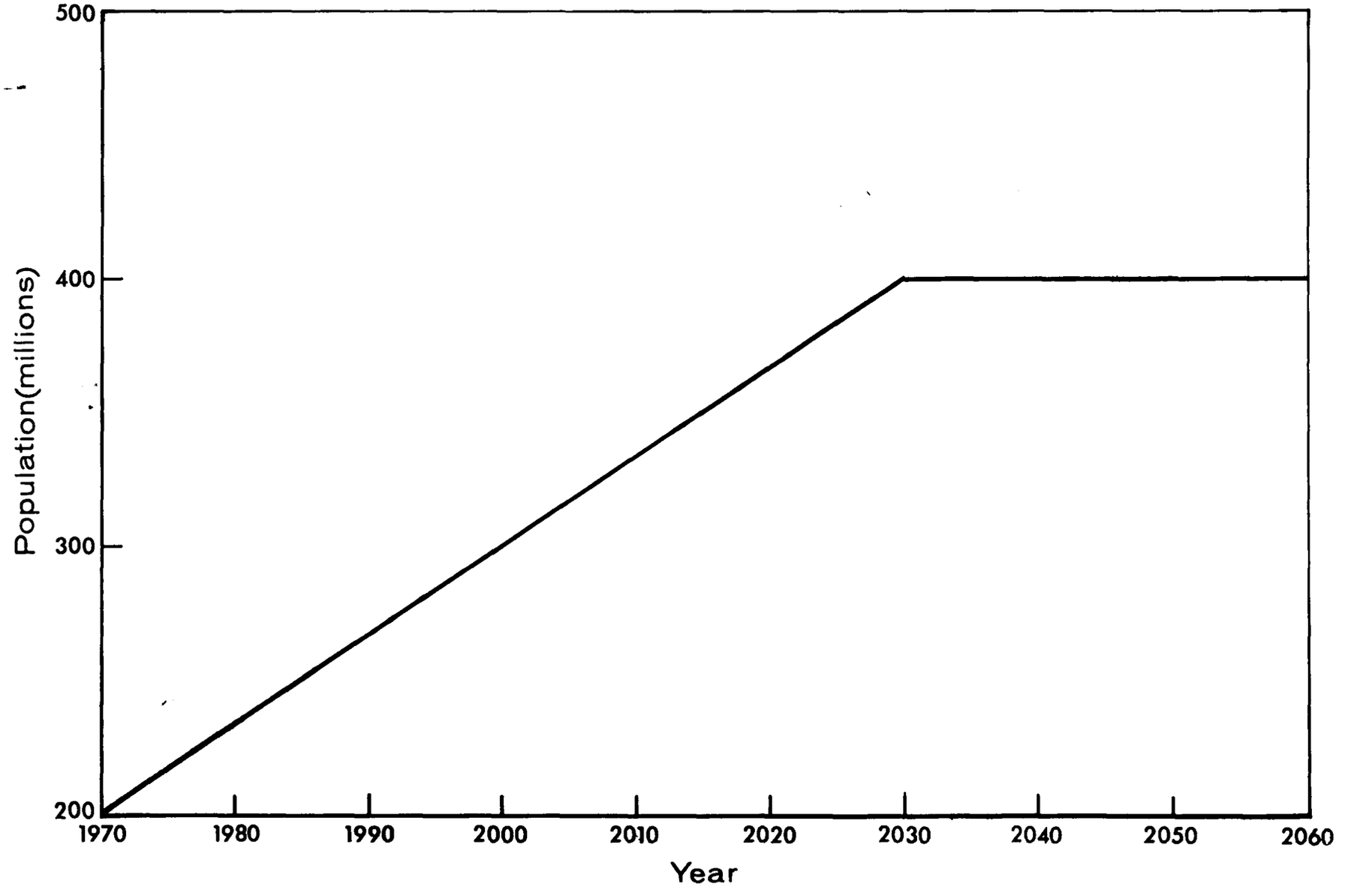

Figure B.1 United States population projection

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APPENDIX C

ENVIRONMENTAL TRANSPORT

I. INTRODUCTION

Radioactive materials released to the environment become dispersed in the surrounding media (air, water, etc.) and ultimately may produce health effects in man. A factor necessary to assess the impact of a given radionuclide release on populations is the transport factor which converts quantity released to concentration of the radionuclide in a specific medium at a given location and time following its release. This appendix discusses this factor, as applied in the calculations carried out in this study.

For reasons discussed below, different environmental transport models were used for: (a) local populations (defined as those within 80 km, or approximately 50 miles, of the point of release); (b) regional populations (including portions of the eastern United States and Canada); and (c) the world population. In developing these models, only the environmental pathways of principal importance from the viewpoint of human uptake and potential health impact have been considered.

II. TRANSPORT TO LOCAL POPULATIONS

Members of the local population around a source of radionuclides discharged to the environment are exposed to higher concentrations of these radioactive materials than is the average individual in the U.S. In general, these higher concentrations arise because environmental transport is at an early stage and ultimate dilution has not yet occurred. The doses delivered to local populations during this "first pass" of an effluent will usually constitute a substantial fraction of the entire environmental dose commitment that accrues to these local

populations. For this reason, these local populations are considered a special case for which the initial contribution to environmental dose commitment is calculated separately.

For the nuclides considered (with the exception of tritium) airborne releases constitute the most important release mode. For tritium, since doses resulting from waterborne releases are on the average comparable to those from airborne releases, all tritium was assumed to be discharged to the atmosphere. Annual meteorological conditions for a variety of representative facilities were analyzed. These data indicated an average value for (χ/Q) at a distance of 3 km of 5×10^{-8} $\mu Ci/cm^3$ per $\mu Ci/s$ for a representative facility. This value was used for all airborne-release calculations in this study.

The effect of local population distribution on the average dose to an individual within 80 km of a facility can be calculated theoretically by assuming a "typical" population distribution, or it can be determined directly from actual or projected populations around real plant sites. For this study an analysis of the results of calculations of doses due to gaseous effluents for real and projected populations at 50 reactor sites was used. These results yielded an average value of 0.028 rem/person within 80 km per rem/person at 3 km. This ratio is sufficiently insensitive to variations for specific radionuclides to be representative of all long-lived nuclides in airborne releases that were addressed by this study. Factors for individual long-lived radionuclides at specific facilities may vary by as much as a factor of 5 from the average given above.

The population within 80 km of a nuclear facility site was taken as the average of population values for the above-mentioned 50 reactor sites, obtained primarily from environmental reports. The average population around a site was found to be 1.5×10^6 people in 1980. Population density around individual plants can vary from this by a factor of 3. The average doubling time of these populations is about 40 years. For purposes of calculating age specific factors, 2.5 percent of the population is taken to be under 1 year of age, 45 percent between 1 and 20 years, and the remainder over 20 years of age.

III. TRANSPORT TO REGIONAL POPULATIONS

The transport of radionuclides in the environment is dependent upon both their physical and chemical states. It is assumed that each of the radionuclides considered in this report is released as a gaseous effluent. These effluents spread from the local region to major parts or all of the eastern United States and Canada and, in some cases, are then transported over the entire globe. The pathway leading to doses to these population differs for each radionuclide considered. Iodine-129 and the actinides are assumed to produce population exposures only through buildup in soils in the United States. Tritium is assumed to expose only the population in the eastern United States initially, and ultimately the entire population of the northern hemisphere. Krypton is assumed to expose populations in the eastern United States and Canada initially, and ultimately the world population.

A. Tritium

It is assumed that tritium is released as a gaseous effluent and that a portion of the amount released enters the hydrological cycle through deposition by rainout over the eastern United States (1.5×10^6 mi^2), where it is diluted by the average annual rainfall (40 inches) over this area and then works its way through soil into river systems and finally into the oceans. With some further dilution by uncontaminated water this rainout becomes the water concentration to which the population of the eastern United States (80% of the total U.S. population) is initially exposed. The balance of the amount released is assumed to rainout directly into the oceans where it is augmented by tritium from river outfalls and gives rise to population exposures in the northern hemisphere via the hydrological cycle.

The annual water concentration of tritium in the eastern United States is taken to be the yearly input to the environment diluted by the average annual rainfall over the eastern United States, with an additional dilution factor of one-half applied to take into account dilution of tritium by uncontaminated rainfall and water from deep artesian wells, as well as that portion of tritium effluents that does not fall out over the eastern United States but passes directly out over the eastern coast to the Atlantic Ocean.

B. Krypton-85

Part of the population of the eastern United States and Canada is exposed to air concentrations of krypton-85 as it passes from the points of release to the Atlantic Ocean on its first pass around the world in

general meteorological patterns of flow. The dose from this exposure pathway was derived from the results of a study recently performed at the National Oceanic and Atmospheric Administration. That study estimated that for a plant located in Morris, Illinois, releasing one curie of krypton-85 per year, the population-weighted concentration on its first pass over the eastern United States and Canada to the Atlantic Ocean is 2.5×10^{-16} man-Ci/cm^3. For purposes of this study, this value was considered sufficiently representative for all release points.

C. Iodine-129

As a first approximation, all iodine-129 releases are assumed to deposit uniformly over the eastern United States and to assume a uniform equilibrium distribution with stable iodine in the soil to a depth of 20 cm. This mixture is assumed to give rise to the specific activity of iodine-129 in the diet to which all persons in this part of the country will be exposed. The movement of iodine-129 in the biosphere is not well documented at the present time. For the purpose of this analysis no further dilution or reconcentration in the environment was assumed beyond this equilibrium mixing in the first 20 cm of soil. Thus, these estimates of population exposure to iodine-129 are subject to considerable uncertainty. However, because of its long half-life (17 million years), even if a substantial fraction of iodine-129 migrates into environmental sinks, the total impact of environmental iodine-129 may be considerably larger than that calculated here for 100 years only.

D. Actinides

The actinides are assumed to build up in the eastern United States in a manner similar to that postulated for iodine-129, but with the only exposure pathway taken to be inhalation of resuspended material. The fraction of actinides released that deposits on the soil was taken to be 0.5 for this study; the balance was assumed to rainout over the oceans, where it remains unavailable for human uptake. The assumption that the actinides are uniformly distributed over the eastern United States is made for simplicity in calculating exposures. It would have been equally simple to assume a uniform distribution of population and a non-uniform dispersion of actinides. The essential question for evaluating the acceptability of either of these assumptions is in what direction does the actual population density depart from the average value for the eastern United States at locations where the actinides are most likely to be initially deposited. A review of population densities in the vicinities of three existing fuel reprocessing plants indicates that average population densities are generally higher, by up to an order of magnitude, in the vicinities of such plants. However, in view of the possibility of releases from a wide variety of facilities, and operations such as transportation and waste disposal which may occur in sparsely populated regions, as well as of migration of the actinides to yield a more uniform dispersal, it was judged acceptable to make the less conservative assumption of uniform deposition for this analysis.

IV. TRANSPORT TO WORLD POPULATIONS

Releases of krypton-85 and tritium are dispersed on a global scale and result in exposures of the entire world's population. Doses from releases of iodine-129 and the actinides were assumed to be restricted to the United States population.

A. Krypton-85

This effluent attains close to a uniform distribution in the world's atmosphere in less than a year following its release. The worldwide concentration of krypton-85 can be estimated by diluting a release into the world's atmosphere (5.14×10^{21} g; sea level air density = 0.00129 g/cm^3). For the purposes of this study, the small correction required for non-uniform distribution during the first year following release has been ignored, except for the previously calculated first pass doses delivered to local and regional populations.

B. Tritium

The worldwide dose due to tritium releases is estimated by diluting the amount released into the circulating waters of the northern hemisphere (7×10^{18} liters) and assuming that the northern hemisphere's population (80 percent of the world's population) is exposed to the resulting concentration.

APPENDIX D

CONVERSION FACTORS

FOR

RADIOLOGICAL DOSE AND HEALTH EFFECTS

I. INTRODUCTION

Two factors required to assess the impact of radionuclide releases to the environment on population-s are: (1) a medium concentration-to-dose conversion factor, and (2) a factor for converting population dose to an expected number of a specific adverse health effect. These factors are discussed below for tritium, krypton-85, iodine-129, and selected actinides.

II. MEDIA CONCENTRATION-TO-DOSE CONVERSION FACTORS

Dose estimates are sensitive to assumptions made concerning the mode of exposure, the amount of radioactivity inhaled or ingested daily, the fraction of activity retained in the organ of interest, and the residence time of the activity in various parts of the body. Other necessary elements entering into dose computations are the physicial considerations of organ mass and radionuclide distribution within the organ. In the present state of the art, the complexities of the radionuclide distribution within organs are nearly always circumvented by assuming a uniform distribution. Information concerning other inputs is based mainly on empirical evidence, gathered largely from fallout studies and medical investigations. In order to reduce the number of variables to be considered in dose calculations, the International Commission on Radiological Protection (ICRP) has postulated a "standard man"; i.e., a model system having standardized biological parameters based on either average values or best estimates as listed in the scientific literature. The standard man is a hypothetical adult

industrial worker and it is not clear to what extent parameters so defined are applicable to an environmentally exposed population.

For particular radionuclides, the sensitivity of certain age groups may be the limiting factor. For example, in the case of iodine-131 exposures, the Federal Radiation Council has defined children as the most sensitive population group; therefore, the biological parameters used in the media-to-dose conversion factors for this radionuclide are not based on standard man. Rather, models appropriate for children's thyroid glands and thyroid metabolism have been used. For the other radionuclides considered here, little is known concerning differences between adults and children. Such differences are seldom considered in the literature. Thus, the conversion factors listed in the subsequent sections, while adequate, must be considered only as first order approximations and not as definitive estimates of doses from environmentally distributed radionuclides.

Media concentration-to-dose conversion factors used in this report are listed in Table D.1 and discussed below for the radionuclides considered in this report--krypton-85, tritium, iodine-129, and certain of the actinides.

Table D.1

Summary of air concentration-to-dose conversion factors

Radionuclide	Critical Organ	Conversion Factor[a] (rem/yr)/(pCi/m^3 air)
^{85}Kr----------	Whole body	1.5×10^{-8}
	Gonads (female)	1.5×10^{-8}
	Gonads (male)	2.0×10^{-8}
	Lung	3.0×10^{-8}
	Skin	50.0×10^{-8}
^{3}H----------	Whole body	1.7×10^{-6}
^{129}I----------	Infant thyroid	15
	Adult thyroid	4.6
^{239}Pu----------	Lung	12

[a]These factors are for continuous exposure to concentrations expressed in pCi/m^3 of air.

A. Krypton-85

About 99 percent of the decay energy of krypton-85, a noble gas, is dissipated by beta particles which have no potential for deep penetration in tissue.

Kirk has recently reviewed the literature on krypton-85 dose and established relationships between the krypton-85 concentration in air and the resultant doses to various organs. A review of these results shows which radiations are important. For the whole body, dose and risk estimates can be based on a consideration of external photon exposures, i.e., gamma rays and bremsstrahlung. For genetic risk calculations, the

gonadal dose, in the case of males, is from exposure from external photons; while for females, the whole body dose estimate can be used. Dose estimates for the lung are based on internal beta dose plus the total body gamma-ray dose. Skin dose is based on the dose delivered by external beta radiation after making an allowance for the shielding provided by clothing and the nonviable epithelium (a 75 percent reduction of dose).

B. Tritium

Dose estimates from tritium exposure are usually based on the assumption that the isotope is contained in body water. Chronic exposure to environmental tritium, however, has been shown to result in the incorporation of tritium into organic molecules from which tritium is lost at a slower rate than from body water. If it is assumed that, under equilibrium conditions, all body hydrogen (7.0 kg in standard man) is uniformly labelled, a sustained concentration of 1 pCi/liter body water would lead to a body burden of 63 pCi, as opposed to 43 pCi if, as in the ICRP model, distribution in body water alone is considered. Evans found that tritium was not, in fact, quite uniformly distributed through deer tissues. Assuming his observed factors to be applicable to man, he calculated a body burden of 60 pCi for standard man with sustained concentration of 1 pCi/liter in body water, i.e., a body burden a factor of 1.4 higher than that based on the ICRP model. A factor of 1.5 (63/43), although only marginally different from Evans, was selected as an appropriate value for this analysis.

Since it is apparent that, under chronic exposure conditions, tritium may become incorporated into the genetic material (DNA), it has been suggested that the relative biological effectiveness of tritium in terms of genetic effects may be greater than unity as a result of DNA degradation from transmutation and recoil processes in addition to that due to absorbed energy from ionization processes due to beta emissions. However, from both experimental and theoretical considerations, it has been concluded that it is the absorbed dose to mammalian cell nuclei from incorporated internuclear tritium which determines quantitatively the degree of effect. The assumption made in these calculations is that the appropriate value for the quality factor for tritium dose equivalent estimation is 1.0 as recently adopted by both the National Council on Radiation Protection and Measurements (NCRP) and the ICRP.

A sustained concentration of 1 pCi tritium per liter of body water would thus be equivalent to a specific activity (assuming uniform labelling of all body hydrogen) of 9×10^{-3} pCi tritium/g hydrogen, and would deliver an annual dose to body tissues of approximately 10^{-4} mrem.

The concentration of tritium in body water resulting from exposure to tritium in air is obtained by diluting the daily intake of tritium by inhalation into the 43 liters of body water with a biological half-life of 12 days. This amount of tritium is doubled to account for absorption of tritium through the skin. This leads to an annual dose of 1.7×10^{-3} mrem for an air concentration of 1 pCi tritium/m^3.

C. Iodine-129

Atmospheric releases of iodine-129 may result in its accumulation in the thyroid glands of persons living in the area surrounding the point of release. For radioiodines, the most significant pathway for exposure of man is generally the grass-cow-milk chain, particularly when milk is not diluted with uncontaminated supplies. Direct deposition on foliage is likely to be the most important route of contamination of edible herbage.

Because of the long half-life of iodine-129, plant uptake of this radionuclide from the soil should also be considered. In general, it is assumed that such plant uptake will be proportional to its specific activity (curies of iodine-129 per gram of stable iodine) in the soil. The specific activity in the soil at a specific location will be a function of distance from the point of release and the buildup from continuing releases. At any given time the specific activity in the ecological chain will be somewhat less that the specific activity of the iodine-129 in the air. In many cases the specific activity will be much less because of the large stable iodine reservoir in soils and other parts of the terrestrial pathway.

For a given concentration of iodine-129 in milk, it has been determined that a 6-month-old child would sustain the highest dose when considering the exposure of individuals to this radionuclide via the grass-cow-milk chain. According to Durbin, the average daily intake of whole milk by U.S. children during the first year of life is about 760 ml. Appropriate representative data to define the relationship between

the amount of iodine ingested by a 6-month-old child and its resultant concentration in the thyroid gland are: thyroid weight, 1.8 g; fraction of ingested iodine in thyroid, 0.35, and biological half-life of iodine in thyroid, 23 d. Equivalent data for adults, appropriate to the calculation of average population doses, are: daily milk consumption, 500 ml; thyroid weight, 20 g; fraction ingested reaching critical organ, 0.3; and biological half-life in thyroid, 138 d. Use of these values yields an annual dose to the adult thyroid of 1.9 mrem for an iodine-129 concentration of 1 pCi/liter of milk. The corresponding annual dose to the thyroid of children whose daily consumption of milk during the first year of life contains 1 pCi/liter is 6.3 mrem.

To determine milk concentrations from given ground and air concentrations, use was made of the following factors derived from references by Bryant (1970) and the Federal Radiation Council Report No. 1:

(a) 2.4×10^3 pCi/liter of milk per pCi/m^3 of air; and

(b) 0.28 pCi/liter of milk per pCi/m^2 of ground surface.

For these intermediate conversion factors, it was assumed that the grazing area for a dairy cow is 80 m^2 per day and that airborne radioiodine has a deposition velocity of 0.5 cm/s.

The annual thyroid dose rate corresponding to unit specific activity (1 μCi iodine-129/g total iodine) in the thyroid is 0.44 rem/yr for an adult and 0.24 rem/yr for a 6-month-old child. Adoption of a value of 0.44 rem/yr as the dose delivered to a thyroid containing 1 μCi iodine-129/g total iodine would thus appear to be a conservative

estimate for all cases. This factor was applied in conversions of medium concentrations to thyroid dose rate for the long-term assessment of iodine-129 releases.

D. Plutonium-239 and Other Actinides

The potential health risks from inhalation of a radionuclide depend on whether it is in a soluble or an insoluble form. In this report, it was assumed that all actinides were in an insoluble form. Present experience indicates that this is the case for plutonium effluents to the atmosphere from fuel reprocessing plants.

In this report, dose estimates from inhaled actinides are based on the new ICRP lung model. However, the biological half-life of insoluble actinides in the lung (pulmonary region) was assumed to be 1,000 days. Using this mode, sustained exposure to an air concentration of 1 pCi/m^3 of insoluble plutonium-239 would lead to a dose rate of 12 rem per year in the pulmonary region. It is assumed that the risk to this region is representative of the total risk to the lung.

Media-concentration-to-dose conversion factors for other actinide radionuclides relative to plutonium-239 were computed by taking into account the effective energy absorbed per disintegration and the physical half-life of each radionuclide, as given in ICRP Publication Nos. 2 and 6. These relative conversion factors are listed in table D.2

Table D.2

Air concentration-to-lung dose conversion factors
for actinide radionuclides relative to that for plutonium-239

Radionuclide	Relative conversion factor[a]
^{238}Pu	1
^{239}Pu	1
^{240}Pu	1
^{241}Pu	0.001
^{241}Am	0.25
^{242}Cm	0.17
^{244}Cm	0.33

[a]Plutonium-239 conversion factor = (12 rem/yr)/(1 pCi/m^3).

It is realized that there are a number of complexities involved in the computation of doses resulting from inhalation of radionuclides. For example, the health risk resulting from a given amount of a radionuclide in the pulmonary region of the lung is dependent upon its distribution as well as the total amount present. However, due to limitations of current models, the amount of radionuclide in the organ was assumed to be uniformly distributed. In the case of alpha emitters, such averaging is obviously inappropriate if there are only a few particles present. ICRP Publication No. 6 recognized this and states, "...in the case of the lung, an estimate of the dose equivalent to the critical tissue determined merely by the product of quality factor and mean dose may be greatly in error, but further experimental evidence is needed before a better estimate can be made...."

For the purposes of this report, inhalation was the only route of intake considered for plutonium-239 and other actinides. Because of the assumed insoluble form for the actinides, doses resulting from other pathways were considered to be of relatively minor importance.

III. DOSE-TO-RISK CONVERSION FACTORS

The numerical values of the dose-to-risk conversion factors used in this study were derived primarily from the recent (November, 1972) National Academy of Sciences Committee on Biological Effects of Ionizing Radiation (BEIR) report. It is emphasized that although these numbers may be used as the best available for the purpose of making risk- and cost-benefit analyses, they cannot be used to accurately predict the number of casualties. For a given dose equivalent, the BEIR report estimates a range for the health impact per million exposed persons. For example, the BEIR results from a study of the major sources of cancer mortality data yield an absolute risk[1] estimate of 54-132 deaths annually per 10^6 person-rems for a 27-year followup period. Depending on the details of the risk model used, the BEIR Committee's relative risk[2] estimate is 160-450 deaths per 10^6 person-rems. It is seen that these estimates differ by a factor of 3 to 4, even when applied to sample populations studied on the basis of the same dose rates.

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[1]Absolute risk estimates are based on the reported number of cancer deaths per rad that have been observed in exposed population groups, e.g., Hiroshima, Nagasaki, etc.

[2]Relative risk estimates are based on the percentage increase of the ambient cancer mortality per rem.

The application of the BEIR risk estimates to exposures at lower dose rates and to population groups more heterogeneous than those studied increases the uncertainty in the risk estimates. Considering the limitations of presently available data and the lack of an accepted theory of radiocarcinogenesis, emphasis should be placed on the differences in health impacts projected by this analysis rather than on the absolute numbers. Where the absolute numbers must be used for risk-cost-benefit balancing, it should be remembered that these health effect estimates are likely to be revised as new information becomes available.

A basic assumption used in the derivation of the dose-to-risk conversion factors was the existence of a no-threshold, linear relationship between absorbed dose and biological effects.

Following are discussions concerning the dose-to-risk conversion factors for the radionuclides of interest in this report.

A. Krypton-85

1. Total Body Dose-to-Somatic Risk

The BEIR report calculates the cancer (including leukemia) mortality risk from whole body radiation exposure by two different models. The absolute risk mode predicts about 100 cancer deaths per 10^6 person-rems; the relative risk model predicts between 160 and 450, or an average of about 300 deaths per 10^6. The average value of the absolute and relative risk models is 200, which is close to the estimates of cancer mortality risk listed as "most likely" by the Committee. Since some types of cancer are not always lethal, cancer mortality is not a

measure of the total cancer risk, which the Committee states as being about twice that of the cancer mortality risk.

For krypton-85 whole-body doses, the following conversion factors were used in this report:

a. 200 cancer deaths per population whole body dose of 10^6 person-rems, and

b. 400 total cancer cases per population whole body dose of 10^6 per-rems.

2. Gonadal Dose-to-Genetic Risk

The range of the risk estimates for genetic effects set forth in the BEIR report is so large that such risks are better considered on a relative basis for different exposure situations than in terms of absolute numbers. The range of uncertainty for the "doubling dose" (the dose required to double the natural mutation rate) is 10-fold (from 20 to 200 rad); and because of the additional uncertainties in (1) the fraction of presently observed genetic effects due to background radiation, and (2) the fraction of deleterious mutations eliminated per generation, the overall uncertainty is about a factor of 25. In a population of one million assumed to receive 30 years of exposure prior to reproduction, the total number of live births showing very serious genetic effects such as congenital anomalies, constitutional and degenerative diseases, etc., is estimated at somewhere between 1,800 and 44,000 if the population is exposed continuously at a dose rate of 1 rem per year. This applies to an equilibrium condition, which occurs after continuous exposure of 5 or more generations. As such, there are expected to occur 60 to 1,500 cases per year at a dose rate of one rem

per year, if a 30-year regeneration period is assumed. The risk to the first generation following initial exposure is about a factor of 5 less.

For the purposes of this study, the geometric average of this annual range was used as the value of the dose-to-risk conversion factor, that is, 300 effects per year for a gonadal dose of 10^6 person-rem per year. This conversion factor was considered to be applicable only for persons up to 30 years of age.

In the BEIR report the notion of "genetic death" as a measure of radiation risk is rejected. Risk analysis was in terms of early and delayed effects observed postpartum and not in early abortion, still-births, or reduced fecundity. Many of the postpartum effects, however, lead directly to infant mortality. Because of the seriousness of the genetic effects considered here (e.g, mongolism), the emotional and financial stress would be somewhat similar to death impact.

Less serious genetic effects have also been considered by the BEIR Committee. These have been quantified under the category "unspecified ill health." The Committee states that a continuous exposure of one rem per year would lead to an increase in the number of ill health cases by 3 to 30 percent. These less serious genetic effects were not taken into consideration in this study.

3. Lung Dose-to-Cancer Risk

Due to the insufficient data for the younger age groups, estimates of lung cancer mortality in the BEIR report are only for that fraction of the population of age 10 or more. For the risk estimate made below, it is assumed that the fractional abundance for lung tumors in the

younger group is the same as that in the older one. On an absolute risk basis lung cancer mortality is about 26 deaths per annum per 10^6 persons irradiated continuously at a dose rate of 1 rem per year. This is a minimum value. The BEIR report states that the absolute risk estimates may be too low because observation times for exposed persons are still relatively short compared to the long latent period for lung cancer. Furthermore, lung cancer risks calculated on the basis of relative risk would be larger. For the risk estimates made here, it was assumed that the value of the applicable conversion factor was twice that of the absolute risk value, as was the case with whole-body exposures. As such, the lung dose conversion factor was taken to be 50 lung cancer deaths per population lung dose of 10^6 person-rems.

4. Skin Dose-to-Cancer Risk

The dose to the skin delivered by krypton-85 is a factor of 30 higher than that to other organs. However, there is currently no epidemiological evidence of actual risk from the skin dose levels considered here. This does not rule out the linear dose-effect assumption for skin cancer; but the BEIR Committee found that from the extensive evidence they examined, "numerical estimates of risk at low dose levels would not seem to be warranted." However, rather than defining a zero risk per rad for any radiation insult from krypton-85, an upper limit of risk is proposed.

For the purposes of this analysis the following conversion factor was considered to be indicative of an upper limit of risk from skin exposures: 3 skin cancers per year for an annual skin dose of 10^6

person-rems. This assumes that an individual will accrue 30 years of exposure and that the incidence of skin cancer will be 10 percent of all radiation-produced cancers except leukemia, breast, lung, G.I. tract, and bone cancers. It should be noted that skin cancers are rarely fatal and usually not very debilitating.

B. Tritium

1. Total Body Dose-to-Somatic Risk

The somatic effects from tritium doses are not expected to be unique. They are the same as described above for krypton-85. Hence the same conversion factors are applicable.

2. Gonadal Dose-to-Genetic Risk

Some experiments with bacteria have shown that the location of a tritium atom on a particular DNA base can enhance the mutation rate. However, if it is assumed that tritium labeling is a random phenomenon, the percentage for such locations that are specifically labeled will be extremely small at the exposure levels considered here. Therefore, the gonadal dose-to-genetic risk conversion factor for krypton-85 is assumed to be also appropriate for estimating the genetic risk from tritium exposures.

C. Iodine-129

Radioiodine intakes by humans are concentrated primarily in the thyroid. Because of this and other relevant factors the thyroid is considered to be the critical organ for doses resulting from such intakes. Doses to other organs are orders of magnitude less. Two health effects follow high level exposures of thyroid tissue to ionizing

radiation: benign neoplasms and thyroid cancer. Though the former are a more common radiation effect, only the more serious risk from cancer is considered in this study.

Children are more susceptible than adults to thyroid damage from radiation exposures. Thyroid cancer, however, is not usually a fatal disease for young persons, but mortality from it approaches 25 percent for persons well past middle age.

The BEIR report provides risk estimates only for morbidity (not mortality) and only for persons under 9 years of age. From the Hiroshima data and other studies it would appear that for persons over 20 years old the radiation-induced thyroid cancer incidence is much lower and may approach zero.

Conversion factors used in this study are based on risk estimates described in ICRP Publication No. 8 as well as on the mean values derived from the BEIR Committee's various estimates of incidence per rem of dose. Infants and fetuses, comprising approximately 2.5 percent of the population, are the most sensitive group. For this age group, about 150 thyroid cancers may accrue annually per 10^6 person-rem annual exposure to the thyroid. For the approximately 40 percent of the population that is in the 1-19 year age group it is assumed that the incidence is a factor of about 4 less, and that for the balance of the population, it is a factor of 30 less.

Following are the values used in this report for the factor converting thyroid dose to number of cases of thyroid cancers (morbidity, not mortality):

a. Less than age 1:

150 cases per population thyroid dose of 10^6 person-rems.

b. Ages 1-19:

35 cases per population thyroid dose of 10^6 person-rems.

c. Ages greater than 20:

5 cases per population thyroid dose of 10^6 person-rems.

It is unlikely that the annual number of cases of mortality from these cancers would be much larger than 25 percent of the total number of cases.

D. Plutonium and other Actinides

For the purposes of this study, it was assumed that the conversion factor for lung dose-to-cancer risk for plutonium and other actinides has the same numerical value as that for krypton-85; namely, 50 lung cancer deaths per population lung dose of 10^6 person-rems. It should be recognized that the use of doses calculated on the basis of uniform distribution of actinides in the lung introduces uncertainties in this dose-to-risk conversion factor, since the risk data is based upon such doses being delivered to the basal cells of the bronchial epithelium. Revision of this factor may be necessary when more adequate models become available for dynamic lung clearance and any differences in effects due to non-uniform dose distribution.

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ABSTRACT: The concept of environmental dose commitment is developed and illustrated by application to projected releases of selected radionuclides from the nuclear power industry over the next fifty years. The concept encompasses the total projected radiation dose to populations committed by the irreversible release of long-lived radionuclides to the environment, and forms a basis for estimating the total potential consequences on public health of such environmental releases. Because of the difficulty of making projec-

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