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GREAT LAKES WATERS:
RADIATION DOSE COMMITMENTS, POTENTIAL HEALTH
EFFECTS, AND COST-BENEFIT CONSIDERATIONS

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FOREWORD

The Great Lakes constitute a dominant geographical feature in North America and are a major source of water for consumption by the large concentrations of people in the basin. The study herein reported includes an assessment of levels of radioactivity in the waters of the lakes. The report indicates dose levels associated with water ingestion and provides a rational evaluation of cleanup strategies. Emphasis has been placed on the consequences of setting a maximum dose standard for drinking water that is taken from the Great Lakes, and on the feasibility of meeting the standard.

The difficulty of ascertaining the amount of radioactivity in the Great Lakes and the resultant dose levels associated with ingestion of the waters could well be characterized by a historical quotation:

Mathematicians are like Frenchmen: whatever you say to them they translate into their own language, and forthwith it is something entirely different.*

Nonetheless, the contributors to this report have produced a valuable yet concise supplement to the literature treating environmental concerns--in this case, the welfare of residents consuming water in the Great Lakes basin.

July 1977

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*Johann Wolfgang von Goethe (German poet and dramatist) 1749-1832.

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**GREAT LAKES WATERS:
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Abstract

In 1972, a Great Lakes Water Quality Agreement was signed by the United States and Canadian Governments. It was stipulated that the operation and effectiveness of the agreement were to be reviewed comprehensively in 1977. Aspects of the agreement concern nondegradation of Great Lakes waters and maintenance of levels of radioactivity or other potential pollutants at levels considered as low as practicable. A refined radioactivity objective of one millirem is proposed in the Water Quality Agreement. The implications of adoption of this objective are not known fully. The Division of Environmental Impact Studies was commissioned by ERDA's Division of Technology Overview to summarize the information available on the current levels of radioactivity in Great Lakes waters, compute radiation-dose commitment (integrated dose over 50 years after consumption of 2.2 liters of water for one year), and to comment on the feasibility and cost-benefit considerations associated with the refined one-millirem objective. Current levels of radioactivity in the waters of Lakes Michigan, Ontario, Erie, and Huron result in dose commitments in excess of 1 mrem for whole body and 6 mrem for bone. Future projections of isotope concentrations in Great Lakes water indicate similar dose commitments for drinking water in the year 2050. Reduction of the levels of radioactivity in Great Lakes waters is not feasible, but cost-benefit considerations support removal of ^{226}Ra and ^{90}Sr through interceptive technology before water consumption. Adoption of the one-millirem objective is not propitious.

1. INTRODUCTION

The purpose of this short-term project is to provide the Division of Technology Overview (DTO), Energy Research and Development Administration (ERDA), with a summary of radioisotope concentrations and resulting total equivalent dose-50 years (TED_{50}) that would result from consuming 2.2 liters of untreated lake water per day. Radioisotope concentrations in lake water

are summarized from readily available data sources only, and TED₅₀ estimates are calculated. The TED₅₀ estimates are discussed in the context of a refined radioactivity objective for the Great Lakes based on TED₅₀ of one millirem. Future projections of radioisotope concentrations in lake waters are made. Potential health effects, specifically cancers and mutations, that could result from current and projected levels of radioactivity in Great Lakes water are estimated. Action alternatives, such as the feasibility and cost vs. benefit of reducing the levels of radioactivity in lake water, are discussed. Deficiencies in the existing data and monitoring programs, and the insufficient understanding of lake-water dynamics and ecology, are also discussed. Final emphasis is placed on the inadvisability of adoption of a refined radioactivity objective of TED₅₀ equal to one millirem, based on the current state of information regarding radioactivity in Great Lakes water.

2. THE REVISED RADIOACTIVITY OBJECTIVE

Consideration of "dose* objectives" from radioisotopes, contained in Great Lakes water, needs to be improved substantially by: (1) accurate determination of the present radioisotope inventory; (2) quantitative identification of the input sources of radioactivity; and (3) evaluation of any natural processes, viz., concentration of suspended solids, sedimentation, concentration in aquatic organisms, or lake flushing, which may influence water quality.

Implicit in the concept of a water-quality objective is the matter of control, nondegradation, or possibly reduction in pollution levels. Explicit is the fact that only man-made sources of input are subject to interceptive control alternatives "before the fact." These matters must be addressed directly before serious deliberations on water-quality objectives are begun. The large uncertainties and incomplete information about existing radioisotope concentrations in lake waters result in correspondingly uncertain estimates of TED₅₀.

3. BACKGROUND

3.1 DOSE FROM INTERNAL EMITTERS

Relationships between concentration of ingested radioisotopes and biological effects are particularly difficult to predict for internally deposited radionuclides. The difficulties of the conversion from concentration to dose arise from the very complex relationships between incorporation of isotopes into the body, their distribution in various body fluids, organs, or tissues, and their excretion. Dose to cells at risk for carcinogenesis or mutagenesis can only be estimated in most cases.

Doses from the isotopes present in lake waters are estimated through complex equations involving absorption, transfer, retention coefficients for

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*In this document, dose refers to absorbed dose and radiation quality and is expressed in rem.

radioisotopes, and quality factors for the radiations emitted--recommended by ICRP-II, ICRP-20, and ICRP-23. These equations represent the current state-of-the-art. Subcommittees of both ICRP and NCRP take full cognizance of emerging data and new concepts that may influence internal-dose calculations. The staff is advised that new formulations are currently under study, and we emphasize that reevaluations of dose estimate may be necessary when new recommendations become available. Using the formulations described in Appendix A, we provide dose estimates for whole body, bone, and gonads. A Quality Factor (QF) of ten is assumed for all alpha emitters. Whole-body dose is based on total effective energy deposited in a 70-kg standard man. Bone dose is computed based on total effective energy deposited in a bone mass of 7 kg. An additional modifying factor of five is used for all bone-seeking alpha- and beta-emitting nuclides except when ^{226}Ra is the first member of the decay chain. Gonadal dose is based on whole-body dose, but is integrated over 30 years rather than 50 years. The dose commitment to gonads from ^{226}Ra includes a modifying factor of 0.2 to take into account the ratio of mass-absorption coefficients and geometrical factors between bone and soft tissue. Thus, for identical time integrals, the gonadal dose is a factor of 50 lower than the bone dose.¹

Based on the isotope inventories in Great Lakes waters, dose to bone from ^{90}Sr and ^{226}Ra is of greatest significance. Bone is the organ at risk for carcinogenesis and any dose criteria contemplated should be based on dose to bone and gonads. The use of total-body dose from the isotope distribution in lake water is biologically meaningless.

3.2 DEFINITION OF TED_{50}

The revised radioactivity standard for Great Lakes waters is based on TED_{50} . In a sense, this is a commendable gesture because health effects, such as cancer and mutagenesis, depend on "dose" or energy absorbed by the tissues at risk. However, we stress that relationships between dose (sustained at low dose rates) from internal emitters and biological effects remain poorly understood. The Argonne staff believes that relationships between incorporated activity and biological effect are better understood. Perhaps more important to this report, the definition and usage of TED_{50} is not totally consistent among the various documents reviewed. This inconsistency may contribute to differences of opinion regarding the biological significance of the levels of radioactivity currently present in, or projected for, the Great Lakes.

One issue is the relationship between TED_{50} and annual dose rate. Quoting from the Federal Register, 42(65) 5 April 1977, "This water quality level is expressed in terms of the total equivalent dose to ICRP reference man integrated over 50 years: (TED_{50}). It is proposed that water quality outside of any source control area, as defined herein, shall not result in a TED_{50} greater than one mrem to the whole body from daily ingestion of 2.2 liters of lake water for one year. Therefore, even for lifetime (50 years) ingestion, the annual dose rate will not exceed 1 mrem/year." The Argonne staff considers the last sentence regarding the annual dose rate to be misleading. We believe the sentence should state that, for continuous ingestion for 50 years, the dose received in any given year will not exceed 1 mrem.

Quoting from the Public Notice from the Department of State concerning the report of the International Working Group on Radioactive Objective for Great Lakes Water Quality, TED_{50} is defined as follows: "For the purpose of this report, the total equivalent dose to a particular organ, tissue, or whole body is the accumulated dose equivalent over 50 years resulting from the daily ingestion of 2.2 liters of lake water for one year." The expression is $TED_{50} = \sum_i D_{50i} Q_i N_i$ rem; where D_{50i} equals total absorbed dose integrated over a period of 50 years after intake of radionuclide "i", Q_i equals quality factor, and N_i equals the product of all other modifying factors. Note that annual dose rate is not mentioned.

Quoting from Appendix D of the draft of the Fifth Annual Report of the Radioactivity Subcommittee of the Water Quality Board's Implementation Committee (p. 37, para. 3, line 4), "The annual radiological dose (TED_{50}) to the bone and an individual drinking the water at that point ...." Note that TED_{50} is related to annual radiological dose, whereas there is no reference to annual dose in the State Department definition. The Argonne staff objects to quoting annual dose rates. Over a period of 365 days, the dose rate may change considerably. To illustrate this point, Table 1 shows the dose rate per day from ^{90}Sr and ^{226}Ra to bone and whole body for continuous intake and for several instances of one-year intake after several time intervals.

Table 1. Dose Rates to Bone and Whole Body after Ingestion of ^{90}Sr and ^{226}Ra for One Year and for 50 Years (at one picocurie per liter)

Retention Function	Isotope	Interval (years)							
		One-year Residency				50-year Residency			
		1	10	30	50	1	10	30	50
<u>Dose Rates to Whole Body ($\mu\text{rem/day}$)^a</u>									
Exponential ^b	⁹⁰ Sr	0.19	0.13	0.052	0.021	0.19	1.56	3.2	3.87
Power	⁹⁰ Sr	0.28	0.09	0.04	0.02	0.28	1.4	2.7	3.3
Exponential ^c	²²⁶ Ra	16.9	1.34	0.0048	0.001	16.9	64.7	68.8	68.9
Power	²²⁶ Ra	0.98	0.15	0.09	0.06	0.98	3.0	5.1	6.6
<u>Dose Rates to Bone ($\mu\text{rem/day}$)^a</u>									
Exponential ^d	⁹⁰ Sr	2.9	2.0	0.9	0.4	2.9	24	51	64
Power	⁹⁰ Sr	2.8	0.9	0.4	0.2	2.8	14	27	33
Exponential ^e	²²⁶ Ra	25	22	16	12	25	239	617	892
Power	²²⁶ Ra	9.8	1.5	0.8	0.6	9.8	30	51	66

^aDose rates are calculated from the accumulated body burden at the termination of the interval specified.

^bEffective half-life based on biological retention and physical decay = 5700 days.

^cEffective half-life based on biological retention and physical decay = 900 days.

^dEffective half-life based on biological retention and physical decay = 6400 days.

^eEffective half-life based on biological retention and physical decay = 1.6×10^4 days.

Because the water-quality objective is defined in terms of TED_{50} , and TED_{50} estimates can be translated to health effects to predict environmental impacts, the staff emphasizes that total dose commitment after any one-year consumption of water provides an unrealistic estimate of dose and of effect. Integration of dose for 30-50 years of water consumption provides a more realistic assessment of potential health effects.

If the Senior Review Group or any other deliberative body uses a water-quality objective defined in terms of TED_{50} , unambiguous definition is required. The Argonne staff fails to understand how the Radioactivity Subcommittee could press strongly for adoption of the one-millirem objective when: (1) confusion over definition and computation of TED_{50} is apparent; (2) most of the monitoring data evaluated by them is recognized as wholly inadequate for dose computations; (3) TED_{50} s that result from drinking lake water in 1977 were not known accurately by them due to incomplete identification of isotope inventories at the time the one-millirem objective was enunciated; and (4) TED_{50} , based on one-year water consumption, is useful only for comparative purposes and underestimates potential health effects by a large margin. Based on these considerations, we feel that adoption of the one-millirem objective is premature.

3.3 DOSE-RESPONSE RELATIONSHIPS AND HEALTH EFFECTS

Exposure standards or objectives for ionizing radiations, or any other environmental toxicant, focus on protection of the public health with the intention of minimizing adverse health effects. All human activities involve some element of risk, and standards for environmentally significant toxicants should be set based on consideration of potential adverse health effects viewed in the perspective of potential benefits to the society. Because definitive data derived from epidemiologic studies of human populations, or from animal studies, are lacking with respect to the acceptance or rejection of a linear relationship between adverse health effects and radiation dose, the linear relationship is used conventionally for risk estimation for both somatic and genetic effects of ionizing radiations (BEIR Report). This action has been considered prudent by various committees charged with recommendations for risk estimation. Definitive data do not exist to confirm or reject the hypothesis of linearity at doses below 50-100 rem received at low dose rates and over long periods of time.

4. RADIONUCLIDES IN GREAT LAKES WATER

4.1 RADIOISOTOPE CONCENTRATIONS

The concentrations of radionuclides (pCi/l) in the waters of the Great Lakes are summarized in Tables 2-4. Tables 2 and 3 were compiled by Dr. D. N. Edgington, and Table 4 by the staff of the Division of Environmental Impact Studies. Table 4 compiles the summarized results from Tables 2 and 3 and the essentially unpublished monitoring data that have been provided to the Radioactivity Subcommittee of the Great Lakes Water Quality Board. The accuracy and precision of the data and detection limits given in these reports are unknown and have not been evaluated by the Radioactivity Subcommittee. The values in parenthesis for ^{226}Ra for Lake Huron (Table 4)

Table 2. Concentrations of Radionuclides in the Great Lakes^a (pCi/l)

Date	Nuclide	Lake				
		Superior	Michigan	Huron	Erie	Ontario
1950s	²²⁶ Ra ²¹⁰ Pb		0.02 ^b 0.03			0.04 ^c
1972	⁴⁰ K ³ H		1.4 285			
1973	⁹⁰ Sr ¹³⁷ Cs ²³⁹ Pu ²³⁸ Pu	0.53 0.07 0.0006	0.81 0.05 0.0007 0.00003	0.96 0.04 ^d 0.0006	1.1 0.02 0.0002	1.3 0.03 0.0002
1976	⁹⁰ Sr ¹³⁷ Cs ²³⁹ Pu		0.89 0.05 0.0006	0.88 0.031 0.0005	1.0 0.013 0.0002	1.2 0.02 0.0003

^aUnless otherwise indicated, values for these nuclides may be found in articles in RER Division Annual Reports 1972-1975, ANL-7960 Part III, ANL-8060 Part 3, ANL 75-3 Part III, and ANL 75-60 Part III.

^bA. Stehney and H. Lucas, Proc. Int. Conf. Peaceful Uses At. Energy Geneva 11:49-54, 1956.

^cJ. Hursh, Nucleonics 7:45-59, 1950.

^dFrom P. J. Barry, IAEA Symp. on Environ. Behavior of Radionuclides Released in the Nuclear Industry, Paper No. SM-172-43, pp. 499-506, 1973.

Table 3. ⁹⁰Sr in Great Lakes Water (pCi/l)

Date	Lake				
	Superior	Michigan	Huron	Erie	Ontario
1973 (observed)	0.53	0.81	0.96	1.1	1.3
(calculated)	0.55	0.89	0.90	0.89	1.2
1976 (observed)	—	0.89	0.88	1.02	1.2
(calculated)	0.55	0.88	0.88	0.83	1.1
2050 (calculated)	0.58	0.73	0.67	0.62	0.61

Table 4. Current Levels of Radioactivity in the Great Lakes^a (pCi/l)

Nuclide	Lake					NPC ^d
	Michigan ^b	Superior	Huron ^c	Erie	Ontario	
⁴⁰ K	(1.4)					
³ H	200-900 (285)	350	1700-30,300	300-400	700-2530	3 × 10 ⁷
⁹⁰ Sr	0.7-0.9 (0.81)		0.61-1.07	0.9-1.15	0.87-1.0	3 × 10 ²
¹³⁷ Cs	77 (0.05)	0.071-0.079	0.024-0.090	< 0.07	0.06-0.12	2 × 10 ⁴
^{239,240} Pu	(0.0007)					5 × 10 ³
²³⁸ U	0.1					4 × 10 ⁴
²³⁴ U	0.1					3 × 10 ⁴
²²⁶ Ra	< 0.1 (0.02)		0.1-3.0 (5-11)	0.03 ^e	< 1.0-5.5	3 × 10 ¹
²¹⁰ Pb	(0.03)					1 × 10 ²
²¹⁰ Po						7 × 10 ²
¹²⁵ Sb		0.024-0.050	0.035-0.107			1 × 10 ⁵
¹⁴⁴ Ce	31	0.052-0.073	0.030-0.379	< 0.2		1 × 10 ⁴
¹⁰⁶ Ru				< 0.3		1 × 10 ⁴
⁴⁵ Zr				< 0.1	< 0.3	6 × 10 ⁴
¹³⁴ Cs				< 0.05	< 0.08	9 × 10 ³
Gross alpha			< 1 (8-40)		< 1-45	
Gross beta	2-70		2-37 (13-35)	5-15	1-47	

^aUnless otherwise indicated, these data are from the Great Lakes Water Quality Fourth Annual Report, Appendix D, Annual Report of the Radioactivity Subcommittee, June 1976. The accuracy and the precision of data and detection limits given in this report have not been evaluated by the Radioactivity Subcommittee.

^bParenthetical values are from Table 2.

^cParenthetical values represent probable point-source releases.

^dFrom Appendix B of 10 CFR 20. Values (uncontrolled area-soluble) multiplied by 10⁹ to convert to pCi/l.

^eJ. B. Hursh, "The Radium Content of Public Water Supplies," Rep. UR-257, University of Rochester, Rochester, NY, 1953.

represent values in Serpent Harbour on the North Channel of Lake Huron, which drains the Elliot Lake uranium mining area of northern Ontario; these values were not included in dose calculations because we believe they are not a representative concentration for Lake Huron. Data in Table 4 include the results from numerous radioactivity monitoring programs that regularly check inshore surface waters and biota in the vicinity of operating nuclear facilities that have aqueous discharges to the Great Lakes. The open waters of the lakes are studied irregularly, usually in connection with baseline or special studies, with no immediate plans to follow up on a regular basis. The extent to which these isotope concentrations are representative of open-lake waters (outside source-control areas) or may be representative of water withdrawn for treatment and consumption is unknown.

Included for comparative purposes in Table 4 are maximum permissible concentrations of the radionuclides in water as given in Appendix B of 10 CFR 20 for uncontrolled areas.

As shown in Table 4, the radioisotope inventory in all the lakes is incomplete, and large uncertainties are associated with the values reported.

4.2 NATURAL AND MAN-MADE SOURCES OF RADIOACTIVITY

The principal sources of radioactivity in Great Lakes waters are fallout from nuclear weapons, the influx of natural radioactivity from ground or surface waters, medical or industrially used radioisotopes, and releases from the electrical-power industry and other industries utilizing nuclear or fossil fuels. Quantitative identification of source-term contributions is beyond the scope of the current project, but the Argonne staff considers the industrial and nuclear-power contributions small relative to other sources of radioactivity.

It is necessary to distinguish between those nuclides that are part of the natural radioactive-decay chains and those that have been introduced into the lakes since the onset of nuclear testing 30 years ago. The inventory for the latter radionuclides in the Great Lakes is relatively well known as a result of the world-wide monitoring of atmospheric fallout by the Health and Safety Laboratory, New York City. Studies of ^{239}Pu , ^{137}Cs , and ^{90}Sr in Lake Michigan have established that their concentrations are homogeneous throughout the water column following the winter mixing period. Therefore, the numbers reported for these nuclides in Table 2 are for samples collected in the late spring. Samples from all the lakes in the spring of 1976 indicate that the concentrations of these three nuclides have not changed significantly during the three-year period.

Earlier work by Barry² has shown that the concentration of ^{137}Cs was far higher during the period (late 1960s) when the rates of input from fallout were far greater than now. The data suggest that the residence time of ^{137}Cs in the water column was about one year. Because the present inputs are about 1% of the total inventory in the lake, and the concentrations of ^{137}Cs (and ^{239}Pu or ^{90}Sr) are constant from year to year, it appears that these three nuclides are in equilibrium within the system. Therefore, it might be expected that for ^{90}Sr , where 96% of the total inventory is still in the water column, the decrease in concentration will be largely controlled by the residence time of water in each of the Great Lakes and by radioactive decay. On the other hand, greater than 95% of the total inventory of ^{137}Cs and ^{239}Pu is contained in the sediments. If one assumes that their concentrations in the water column are controlled by a distribution coefficient between sediment and water ($K = \text{pCi/g sediment per pCi/ml water} \approx 2 \times 10^5$), the decrease in concentration in the water will be controlled largely by the rate of burial of old sediment with new. The sedimentation rates in the Great Lakes vary between $> 1.0 \text{ mm/yr}$ to $< 1.0 \text{ cm/yr}$ and the situation is complicated by redistribution of sediments by bioturbation and remobilization by in-lake physical processes. It is therefore difficult to predict accurately the time span over which the concentrations of ^{137}Cs or ^{239}Pu will decrease in the water column.

In contrast to the wealth of detail available concerning the behavior of fallout radionuclides in the lakes, little is known concerning the behavior of the natural radionuclides; in fact, it is not clear that even the concentrations of the more important nuclides are known to even one order of magnitude. This becomes clear when an attempt is made to compare the results of analyses for ^{226}Ra , where values between 0.02 and 5.5 pCi/l have been reported (Table 4). Up to now it has been assumed that ^{226}Ra in the lakes is in equilibrium with its parent ^{238}U and, therefore, has a concentration in sediments of 1-2 pCi/g. Edgington and Callender³ showed that ^{226}Ra is concentrated up to 50 pCi/g in manganese nodules in Green Bay paralleling a similar enrichment in barium. The sources of the excess ^{226}Ra and barium were not established. There is no doubt that the inventory of ^{226}Ra (and uranium) in the Great Lakes has to be increasing due to the extensive use of phosphate fertilizers and the burning of coal in the watersheds surrounding the lakes. What effects these additional loadings have had or might have on the concentration of ^{226}Ra in the water is unknown, but must be evaluated. It is known that there are inputs of ^{226}Ra to northern Lake Huron from uranium-processing mills in Ontario.

4.3 ISOTOPE-INVENTORY PROJECTIONS TO 2050

The concentration of ^{90}Sr as a function of time in each lake may be calculated using the simple time-concentration model of Lerman,⁴ provided that the inputs and the residence time for water in each lake are known. Whereas Lerman considered inputs from the watershed, it can be shown that a simpler model taking account only of direct atmospheric deposition over the lake predicts the observed concentrations in the water column from 1954 up to the present (Table 3). If it can be assumed that the present annual rate of input of ^{90}Sr fallout (which is $\sim 1\%$ of the total up to 1976) continues well into the next century, then the concentrations of ^{90}Sr in each of the lakes in 2050 will not be greatly different from those in 1976 (Table 3). We assume conservatively that by 2020 the concentration of ^{90}Sr in Lakes Erie and Ontario should be very similar to those in Lakes Superior and Michigan, i.e. $\sim 0.2\text{--}0.3$ pCi/l, and by 2050 $\sim 0.1\text{--}0.2$ pCi/l.

Concentrations of ^{137}Cs and ^{239}Pu will decrease slowly as their concentrations in the water column appear to be controlled by equilibration with the sediments. Although we know the sedimentation rates quite accurately, the disappearance of activity in the sediments is not as fast as would be expected because of physical and biological mixing processes.

In the water column the concentration of ^{226}Ra , which contributes a major fraction of the dose commitment to man, is poorly known at best. Little or nothing is known about the effects on the concentration of ^{226}Ra in the water column due to fertilizer runoff or inputs associated with flyash from the burning of coal. On the basis of the ^{90}Sr data, indicating that this nuclide is retained in the water column (96% of the known input from fallout to Lake Michigan is still in the water column), there is a small probability that increased agricultural usage of fertilizer in the watersheds and burning of coal may increase the concentration of ^{226}Ra in the water column. Based on these considerations TED₅₀ values would not be expected to decrease in the near future.

The concentration of ^{210}Pb and ^{210}Po is far higher than that expected to be in equilibrium with the ^{226}Ra in the sediments, at least partly because of their formation from ^{222}Rn in the atmosphere.

The operation of nuclear facilities such as reactors, fuel processing plants, uranium mining and milling facilities, hospitals, and industrial and research facilities involves the regular release of small quantities of radionuclides and the small but finite probability that major releases of radioactivity may enter the environment as a result of an accident. The qualitative as well as quantitative nature of some of these releases constitutes an area of some uncertainty at the present time. This is reflected in the paucity of input source-term data for some radionuclides and lack of specific data for certain lakes such as Lake Superior.

Real effects of present actions are not clear nor can effects be predicted of major releases in drinking-water supplies and those segments of aquatic food chains directly affecting man for a time equal to the lifetime of each radionuclide. This information is needed and will be important to develop site-by-site contingency plans for water treatment, and the long-term usage of water from the Great Lakes.

As radionuclides affecting water quality enter principally from the atmosphere or from points on shore, their transport, chemical, physical, or biological transformations, and disposal throughout the lakes system are dependent on water movements (largely turbulent) on a wide range of horizontal and vertical scales and on fluxes and transformations associated with living organisms and with interfaces (air/water, sediment/water). Without adequate understanding of the relevant patterns, pathways, and mechanisms, proper assessment of predictive modeling of past or proposed management strategies will not be achieved nor will optimum results be realized from expenditures for water-quality improvements, and protection will not be obtained.

Man-made radionuclides are widely used in medical and biological research, diagnosis, and therapy, and in such uses where they have essentially replaced ^{226}Ra , particularly in teletherapy units. Solid sources are used in teletherapy units and present little or no waste-disposal problem because they are enclosed in sealed containers and housed in permanent installations.

Internal use of radionuclides in therapy is restricted primarily to iodine, phosphorus, technetium, potassium, chromium, gold, carbon, and a few other short-lived isotopes. In radiotherapeutic applications, the above-mentioned radionuclides are generally utilized at millicurie levels. With lower dose levels (5-10 mCi) 10% to 15% may be excreted by the patient in the first day and smaller amounts thereafter. With higher levels (~ 100 mCi), as in the case with ^{131}I , as much as 60% to 70% may be excreted in the first day. The wastes in the form of urinary or fecal excretion are generally discharged into the hospital sewer.

In diagnostic studies, small amounts of radionuclides (10-100 μCi) are used and add little to hospital-effluent radioactivity levels.

Disposal from research laboratories is difficult to follow except for control of the amount of a specific radionuclide. However, the problem is that of the radioactive waste discarded down sewer drains every day by laboratory workers; some molecules (e.g. those that can be easily incorporated into DNA, the stable genetic material of genes and chromosomes) are far more likely to inflict biological damage than others of equal radioactivity, provided they find their way into food chains.

Because of the diversity of operations, general statements cannot be made on the volumes of waste and activity levels to be expected from industrial and medical facilities. Furthermore, gross-activity levels are quite meaningless because they provide no information on the specific nuclides present. In most cases, monitoring information on the composition of waste discharges is limited, with only the more hazardous nuclides identified. Although these conditions are generally of some concern, within the context of the water-quality objective the concern is minor.

4.4 ^{137}Cs CESIUM CONCENTRATION IN FISH

The only nuclide in lake waters that is known to concentrate significantly in fish (and, therefore, is important for dose to humans) is ^{137}Cs . The concentration factor for this nuclide in 1971 was ~ 5000 (wet-weight basis). There is some evidence to suggest that this value of the concentration factor for "food-chain fish", such as steelhead, may be decreasing slightly even though no decrease in the water concentration has been detected.

Current inventories of radionuclides in Lake Michigan are:

Plutonium	$\sim 120 \text{ Ci}$	(98% in sediment)
^{137}Cs	$\sim 6 \text{ kCi}$	(95% in sediment)
^{90}Sr	$\sim 4 \text{ kCi}$	(4% in sediment)

The current inventory of ^{226}Ra in lake water is $\sim 100 \text{ Ci}$ with $\sim 1 \text{ kCi}$ in the upper one centimeter of sediment. Similarly, the inventory of ^{210}Pb and ^{210}Po is $\sim 150 \text{ Ci}$ in lake water and 27.5 kCi in the upper one centimeter of sediment.

The availability of these nuclides in the sediments is governed by equilibrium between the water and sediment, and even if all sources could be turned off, it is likely that the water would continue to contain radioactivity due to reequilibration.

4.5 RECOMMENDATIONS FOR MONITORING LAKE WATERS

We strongly recommend that a monitoring program be developed and implemented by the responsible agencies associated with the Great Lakes Study. We also suggest that a standard protocol of radionuclide sampling and analysis be developed. Furthermore, these procedures must be adhered to by the various agencies or laboratories participating in surveillance programs. All monitoring data must be compiled in a uniform format and be easily accessible to all interested parties.

5. TED₅₀ CALCULATIONS

Data from various sources, presented in Table 4, have been used to compile isotope inventories and concentrations for the Great Lakes waters. As emphasized above, the accuracy of these measurements is unknown. We must reemphasize the incompleteness of the isotope-concentration data used for dose calculations so that dose estimates may be viewed in proper perspective.

Specific comments on definition of TED₅₀ are in Section 3.2, and methods by which TED₅₀ values were computed are in Appendix A. Retention of ⁹⁰Sr and ²²⁶Ra in bone, and the resulting dose commitments, follow complex functions. Exponentials and the power function have been used to describe retention for isotopes of alkaline-earth elements. The consensus is that the power function provides the best fit to existing data (Norris et al.⁵ and ICRP-20). The power function was used to compute dose commitments to whole body, bone, and gonads for these isotopes. Considerably higher dose commitments would be inferred from the use of exponential retention functions.

5.1 TED₅₀ ESTIMATES BASED ON CONSUMPTION OF 2.2 LITERS OF LAKE WATER FOR ONE YEAR

The TED₅₀ values for whole body, bone, and gonads after ingestion of water from the Great Lakes are summarized in Table 5. No estimates for Lake Superior are provided due to insufficient data on concentration of ²²⁶Ra. TED₅₀ values for whole body, bone, and gonads (higher range) are in excess of the one-millirem objective for all other Great Lakes. Because confidence limits of the concentration data used for dose calculations are unknown to the Argonne staff, the significance of excess above 1 mrem cannot be estimated. Dose ranges, based on the isotope-concentration ranges given in Table 4, are presented for all lakes except Michigan. Isotope-concentration data given in Table 2 were used to compute TED₅₀ for Lake Michigan water.

Based on available data, the TED₅₀ estimates for Lakes Michigan, Erie, and Huron (lower-range) approach the one-millirem objective for whole-body dose, but the higher-range values exceed the objective by an appreciable margin. The upper range of TED₅₀s for whole-body doses for Lake Ontario exceeds the objective by factors of ~ 3-15. Little significance can be attached to the lower-range estimate of Lake Ontario because the concentration of ²²⁶Ra reported was < 0.1 pCi/l; thus, 0.1 pCi/l was used in dose computation.*

Doses to bone are of the order of ten times higher than whole-body doses. The public notice published by the State Department states that "the total dose equivalent to a single organ or tissue shall be in proportion to the dose limit recommended by ICRP for that tissue." ICRP-15 considers gonads and red marrow as the critical organs for whole-body exposure, and provides limits of 5 rem/yr for those occupationally exposed, and 0.5 rem for

* The Argonne staff has no explanation for the range of ²²⁶Ra values reported in Table 4. Orders-of-magnitude differences between lakes would not be expected. Use of a standardized protocol for analytical methods is needed.

Table 5. Summary of Dose Commitments that Result from Ingestion of 2.2 Liters of Lake Water for One Year

Lake	Dose ^a (mrem)		
	Whole Body ^b	Bone ^b	Gonads ^c
Michigan	1.18	11.5	0.25
Huron	1.14-11.5 ^d	9.7-83	0.17-1.64
Ontario	3.5 ^e -14.7	34 ^e -143	0.54 ^e -2.2
Erie	1.2-1.5	11.7-14.7	0.19-0.24
Superior	Insufficient data for meaningful dose estimation		

^aDoses computed according to ICRP-II and ICRP-20.

^bTotal effective dose integrated over 50 years (TED₅₀) is presented for whole body and bone. Isotope concentrations reported for tributaries were not considered in TED₅₀ calculations. The relationship between doses to bone and to whole body vary for the different lakes due to differences in concentrations of bone-seeking isotopes.

^cDose estimates for ovary and testes (gonads) are integrated over 30 years.

^dDose ranges are presented where a range of isotope concentrations was reported for Great Lakes waters.

^eRepresents only an estimate because the ²²⁶Ra concentration was represented as < 1.0 pCi/l.

members of the general public. The dose limits for bone are a factor of six higher. Based on this proportion, the TED₅₀ for bone would be 6 mrem when the whole-body objective is a TED₅₀ of 1 mrem. Thus, an objective of 6 mrem for bone is exceeded in all the lakes.

5.2 CONCLUSION

5.2.1 TED₅₀ Estimates Exceed One-Millirem Objective

The conclusion from these dose estimates is that TED₅₀ values for whole body and bone are currently in excess of the one-millirem objective. The upper-range values for gonads are also in excess of 1 mrem for all lakes except Michigan. Thus, adoption of the one-millirem objective, which likely would soon become a standard, would require extensive "cleanup" of Great Lakes waters. Overall cleanup of Great Lakes waters is technically and economically impractical. However, as pointed out in Section 7, interceptive measures such as water treatment before ingestion are feasible.

Selection of the TED_{50} criterion is questionable because only one-year ingestion is specified. Because excess risk of carcinogenesis or mutagenesis is the endpoint of concern, doses that result from ingestion of water for some period longer than one year, e.g. 30-50 years, should be considered for a conservative estimate of health effects.

5.2.2 Comparison of TED_{50} Estimates after One Year and 50 Years of Water Ingestion

Doses to whole body and bone that result from 50 years of water ingestion are compared with doses after one-year ingestion in Tables 6-9. Total accumulated doses to gonads are compared for a 30-year reproductive period following water ingestion for either one year or 30 years. Whole-body and bone doses, integrated over 50 years after ingesting water from the Great Lakes for 50 years, exceed the TED_{50} estimates by factors of ~ 40 -50. Gonadal doses after 30 years of water consumption exceed the dose following one-year consumption by factors of ~ 20 -25. If the dose after 50 years of water consumption is accepted as more relevant to prediction of deleterious effects of radiation, note should be taken of upper-limit estimates of 41-498 mrem for whole body, 2800-4900 mrem for bone, and 6-50 mrem for gonads after consumption of water from Lakes Huron, Ontario, and Erie for 50 years. Values for Lake Michigan water consumption are appreciably lower due to the ^{226}Ra concentration reported (Tables 2 and 4).

Table 6. TED_{50} from Radionuclides Measured in Lake Michigan Waters

Nuclide	Concentration (pCi/l)	Dose ^a (mrem)					
		Whole Body ^b		Bone ^b		Gonads ^c	
		1 Year	50 Years	1 Year	50 Years	1 Year	30 Years
^3H	285	0.03	1.44			0.03	0.8
^{40}K	1.4	0.03	1.51			0.03	0.9
^{90}Sr	0.88	1.06 ^d 1.24	35.5 ^e 41.9	10.67 ^d 20.4 ^e	355 ^d 693 ^e	0.18 ^d 0.2	3.3 ^d 3.75
^{137}Cs	0.05	< 0.01	0.09			< 0.01	0.05
^{210}Pb	0.03	0.01	0.58	0.35	14.4	0.01	0.32
^{210}Po	0.03	< 0.01	0.1	< 0.01	0.44	< 0.01	0.06
^{226}Ra	0.02	0.05 ^d 0.51	1.62 ^d 23.3	0.48 ^d 6.5 ^f 4.08	16.2 ^d 184 ^f 133 ^g	< 0.01 ^d 0.1	0.15 ^d 2.66
^{234}U	0.1	< 0.01	< 0.01	< 0.01	0.03	< 0.01	< 0.01
^{238}U	0.1	< 0.01	< 0.01	< 0.01	0.03	< 0.01	< 0.01
$^{239,240}\text{Pu}$	0.0007	< 0.01	< 0.01	< 0.01	0.01	< 0.01	< 0.01
Total dose commitment ^h		1.18	40.8	11.5	386	0.25	5.58

^aRate of water consumption is assumed to be 2.2 liters per day.

^bIntegrated over 50 years for a one-year ingestion and for continuous ingestion for 50 years.

^cDetermined from the soft-tissue dose integrated over 30 years for a one-year ingestion and for continuous ingestion for 30 years.

^dDose from radioactive alkaline-earth elements computed using a retention model based on the power function.

^eEffective half-life = 17.5 years for exponential retention function.

^fEffective half-life = 44 years for exponential retention function.

^gEffective half-life = 17.1 years for exponential retention function.

^hDoses < 0.01 mrem are not included. The power-function retention model was used to compute the contribution from ^{90}Sr and ^{226}Ra .

Table 7. TED₅₀ from Radionuclides Measured in Lake Huron Waters

Nuclide	Concentration (pCi/l)	Dose ^a (mrem)					
		Whole Body ^b		Bone ^b		Gonads ^c	
		1 Year	50 Years	1 Year	50 Years	1 Year	30 Years
³ H	1,700-30,300	0.17-3.07	8.6-154				
⁴⁰ K							
⁹⁰ Sr	0.61-1.07	0.73-1.2 ^d 0.86-1.51	25-40 ^d 29-51	7.3-12.0 ^d 14.1-24.7 ^e	246-404 ^d 463-759 ^e	0.12-0.2 ^d 0.14-0.25	2.3-4.0 ^d 2.6-4.6
¹³⁷ Cs	0.024-0.09	0.01	0.04-0.16				
²¹⁰ Pb							
²¹⁰ Po							
²²⁶ Ra	0.1-3.0	0.24-7.2 ^d 2.5-75.3	8.1-263 ^d 117-3,501	2.4-71 ^d 32.4-971 ^f 20.2-605 ^g	80.9-2,427 ^d 921-27,000 ^f 664-20,000 ^g	0.05-1.44 ^d 0.5-15	1.62-49 ^d 23-700
²³⁴ U							
²³⁸ U							
^{239,240} Pu							
Total dose, ^h commitment		1.14-11.5	42-437	9.7-83	327-2,831	0.17-1.64	3.9-53

^aRate of water consumption is assumed to be 2.2 liters per day.^bIntegrated over 50 years for a one-year ingestion and for continuous ingestion for 50 years.^cDetermined from the soft-tissue dose integrated over 30 years for a one-year ingestion and for continuous ingestion for 30 years.^dDose from radioactive alkaline-earth elements computed using a retention model based on the power function.^eEffective half-life = 17.5 years for exponential retention function.^fEffective half-life = 44 years for exponential retention function.^gEffective half-life = 17.1 years for exponential retention function.^hDoses < 0.01 mrem are not included. The power-function retention model was used to compute the contribution from ⁹⁰Sr and ²²⁶Ra.Table 8. TED₅₀ from Radionuclides Measured in Lake Ontario Waters

Nuclide	Concentration (pCi/l)	Dose ^a (mrem)					
		Whole Body ^b		Bone ^b		Gonads ^c	
		1 Year	50 Years	1 Year	50 Years	1 Year	30 Years
³ H	700-2,530	0.07-0.26	3.55-12.8				
⁴⁰ K							
⁹⁰ Sr	0.87-1.0	1.0-1.2 ^d 1.23-1.41	35-40.4 ^d 41.3-47.5	10.4-12.0 ^d 20.1-23.1 ^e	351-404 ^d 660-759 ^e	0.17-0.2 ^d 0.2-0.23	3.2-3.7 ^d 3.7-4.3
¹³⁷ Cs	0.06-0.12	< 0.001	0.1-0.21			< 0.001	0.12
²¹⁰ Pb							
²¹⁰ Po							
²²⁶ Ra	< 1.0 ^f -5.5	< 2.4-13.2 ^d < 25.1-138	< 81-445 ^d < 1,167-6,219	< 23.8-131 ^d < 323.5-1,779 ^g < 201.6-1,109 ^h	< 609-4,450 ^d < 9,210-50,600 ^g < 6,640-36,500 ^h	< 0.37-2.0 ^d < 5.0-27.5	< 7.6-42 ^d < 133-732
²³⁴ U							
²³⁸ U							
^{239,240} Pu							
Total dose, ^h commitment		3.5-14.7	120-498	34-143	1,160-4,854	0.54-2.2	10.8-45.7

^aRate of water consumption is assumed to be 2.2 liters per day.^bIntegrated over 50 years for a one-year ingestion and for continuous ingestion for 50 years.^cDetermined from the soft-tissue dose integrated over 30 years for a one-year ingestion and for continuous ingestion for 30 years.^dDose from radioactive alkaline-earth elements computed using a retention model based on the power function.^eEffective half-life = 17.5 years for exponential retention function.^fWhen a concentration less than a specified value was reported, that concentration was used to compute dose; thus, it represents a maximum estimate of the lower dose range.^gEffective half-life = 44 years for exponential retention function.^hEffective half-life = 17.1 years for exponential retention function.ⁱDoses < 0.01 mrem are not included. The power-function retention model was used to compute the contribution from ⁹⁰Sr and ²²⁶Ra.

Table 9. TED₅₀ from Radionuclides Measured in Lake Erie Waters

Nuclide	Concentration (pCi/l)	Dose ^a (mrem)					
		Whole Body ^b		Bone ^b		Gonads ^c	
		1 Year	50 Years	1 Year	50 Years	1 Year	50 Years
³ H	300-400	0.03-0.06	1.52-2.03				
⁹⁰ Kr							
⁹⁰ Sr	0.9-1.15	1.1-1.6 ^d 1.30-1.6	36.5-46.5 ^d 44-55	11.0-14.0 ^d 21.5-26.6 ^e	361-465 ^d 706-873 ^e	0.18-0.23 ^d 0.21-0.27	3.3-4.3 ^d 3.8-4.9
¹³⁷ Cs	< 0.07 ^f	< 0.01	< 0.12				
²¹⁰ Pb							
²¹⁰ Po							
²²⁶ Ra	0.03	0.07 ^d 0.75	2.43 ^d 35.0	0.71 ^d 7.08 6.1 ^h	24.3 ^d 276 ^g 199 ^h	0.01 ^d 0.15	0.23 ^d 4.0
²³⁴ U							
²³⁸ U							
^{239,240} Pu							
Total dose commitment		1.2-1.5	40.6-51.1	11.7-16.7	388-499	0.19-0.24	3.5-4.5

^aRate of water consumption is assumed to be 2.2 liters per day.

^bIntegrated over 50 years for a one-year ingestion and for continuous ingestion for 50 years.

^cDetermined from the soft-tissue dose integrated over 10 years for a one-year ingestion and for continuous ingestion for 10 years.

^dDose from radioactive alkaline-earth elements computed using a retention model based on the power function.

^eEffective half-life = 17.5 years for exponential retention function.

^fWhen a concentration less than a specified value was reported, that concentration was used to compute dose; thus, it represents a maximum estimate of the lower dose range.

^gEffective half-life = 44 years for exponential retention function.

^hEffective half-life = 17.3 years for exponential retention function.

ⁱDoses < 0.01 mrem are not included. The power-function retention model was used to compute the contribution from ⁹⁰Sr and ²²⁶Ra.

5.3 DOSE PERSPECTIVES

The dose commitments cited above are high in comparison with the one-millirem objective. Background radiation of ~ 120 mrem/yr produces a total dose of 6000 mrem in 50 years. The average population dose limit of 130 mrem/yr for genetic and somatic tissues (MCRP-43) produces a total dose of 6500 mrem in 50 years. Table 10 summarizes the TED₅₀ estimates for maximum permissible concentrations (MPC) of several isotopes of interest. Note especially that the MPC (10 CFR 20) for any single isotope in Table 10 produces a TED₅₀ greatly in excess of the 1-5 mrem range considered in the water-quality objective.

It is unclear to the Argonne staff why the revised water-quality objective is one millirem. Does some basis exist for selection of a TED₅₀ of 1 rather than 0.1, 10, or 100? If MPC levels are considered "safe", or acceptable levels of risk, why were lower values selected?

The dose commitments presented in Table 10 were made by the same procedures used in estimation of doses from lake waters, so whatever the uncertainties about absolute dose calculation, the comparisons between doses from MPC levels and the one-millirem objective are valid. Our methods of dose calculation are documented fully (Table 11 and App. A) to facilitate comparison

Table 10. TED₅₀ from Maximum Permissible Concentrations (MPC) of Radioisotopes in Water

Nuclide	MPC (pCi/l)	Dose ^a (mrem)					
		Whole Body ^b		Bone ^b		Gonads ^c	
		1 Year	50 Years	1 Year	50 Years	1 Year	30 Years
³ H	30,000,000	3,000	153,000			3,000	91,200
⁹⁰ Sr	300	360 ^d 420	12,100 ^d 14,250	3,600 ^d 6,930 ^e	120,000 ^d 230,000 ^e	60 ^d 70	1,110 ^d 1,275 ^e
¹³⁷ Cs	20,000	700	34,600			700	20,600
²¹⁰ Pb	100	43	1,730	1,160	48,000	43	1,080
²¹⁰ Po	700	48.3	2,408	210	10,000	48.3	1,442
²²⁶ Ra	300	71 ^d 753	2,427 ^d 35,000	714 ^d 9,690 ^f 6,048 ^g	24,300 ^d 276,000 ^f 200,000 ^g	11 ^d 150	228 ^d 4,000
²³⁴ U	30,000	8.10 ^d 12.45	330 ^d 630	81.0 ^d 201	3,300 ^d 9,900	12.45 ^d	360 ^d
²³⁸ U	40,000	9.88 ^d 14.56	400 ^d 720	98.8 ^d 228	4,000 ^d 11,200	14.56 ^d	440 ^d
^{239,240} Pu	5,000	350 ^d 75	8,900 ^d 2,000	3,500 ^d 3,150	89,000 ^d 82,000	47.5 ^d 3,937	750 ^d 117,000

^aRate of water consumption is assumed to be 2.2 liters per day.

^bIntegrated over 50 years for a one-year ingestion and for continuous ingestion for 50 years.

^cDetermined from the soft-tissue dose integrated over 30 years for a one-year ingestion and for continuous ingestion for 30 years.

^dDose from radioactive alkaline-earth elements computed using a retention model based on the power function.

^eEffective half-life = 17.5 years for exponential retention function.

^fEffective half-life = 44 years for exponential retention function.

^gEffective half-life = 17.1 years for exponential retention function.

Table 11. TED₅₀ from a Radionuclide Concentration of One Picocurie per Liter

Nuclide	Dose ^a (mrem)					
	Whole Body ^b		Bone ^b		Gonads ^c	
	1 Year	50 Years	1 Year	50 Years	1 Year	30 Years
³ H	0.0001	0.0051			0.0001	0.00304
⁴⁰ K	0.02	1.08			0.02	0.69
⁹⁰ Sr	1.22 ^d 1.4	40.4 ^d 47.5	12.0 ^d 23.1 ^e	404 ^d 759 ^e	0.20 ^d 0.23	3.7 ^d 4.26
¹³⁷ Cs	0.035	1.73			0.035	1.03
²¹⁰ Pb	0.43	19.3	11.8	480	0.43	10.8
²²⁶ Ra	2.4 ^d 25.1	80.9 ^d 1,167	23.8 ^d 232.5 ^f 201.6 ^g	809 ^d 9,210 ^f 6,640 ^g	0.37 ^d 5.0	7.6 ^d 133
²³⁴ U	0.000270 ^d 0.000415	0.011 ^d 0.021	0.0027 ^d 0.0067	0.11 ^d 0.33	0.000415 ^d	0.012 ^d
²³⁸ U	0.000247 ^d 0.000364	0.010 ^d 0.018	0.0021 ^d 0.0057	0.10 ^d 0.28	0.000364 ^d	0.011 ^d
^{239,240} Pu	0.07 ^d 0.015	1.78 ^d 0.4	0.7 ^d 0.63	17.8 ^d 16.4	0.0095 ^d	0.15 ^d

^aRate of water consumption is assumed to be 2.2 liters per day.

^bIntegrated over 50 years for a one-year ingestion and for continuous ingestion for 50 years.

^cDetermined from the soft-tissue dose integrated over 30 years for a one-year ingestion and for continuous ingestion for 30 years.

^dDose from radioactive alkaline-earth elements computed using a retention model based on the power function.

^eEffective half-life = 17.5 years for exponential retention function.

^fEffective half-life = 44 years for exponential retention function.

^gEffective half-life = 17.1 years for exponential retention function.

of dose estimates provided here with dose estimates of others. Unfortunately, we have been unable so far to locate a complete description of procedures used to calculate TED₅₀ by the Radioactivity Subcommittee of the Great Lakes Water Quality Board.

6. COST-BENEFIT CONSIDERATIONS

6.1 INTRODUCTION

6.1.1 Scope of Considerations

This section describes the Argonne staff's preliminary considerations of three issues:

- (1) health effects resulting from ingestion of the radionuclides reported to be in the Great Lakes,
- (2) the cost of reducing the concentrations of radionuclides from Great Lakes water that is ingested, and
- (3) the annual dollar amount American citizens might be willing to pay to reduce the health effects resulting from ingestion of the radionuclides believed to be in the Great Lakes.

The staff also wishes to call attention to the following pertinent issues not treated in this report:

- (1) the number of people who are ingesting water from the Great Lakes,
- (2) the likelihood that others will ingest water from the Great Lakes,
- (3) the value of reducing concentrations only on the chance that water for ingestion might be drawn from the Great Lakes,
- (4) the possibility that the money spent on reducing the radionuclide concentration in water drawn from the Great Lakes could be more wisely spent elsewhere,
- (5) the possibility that radionuclides will adversely affect biota other than man or accumulate in the food chain, and
- (6) the possibility of eliminating or retarding other beneficial uses of the Great Lakes due to unnecessarily stringent water-quality objectives and standards.

6.1.2 Organization

The following sections describe the staff's considerations of the three issues cited in Section 6.1.1. Section 6.2 reviews the principal health effects to be expected from ingesting the concentrations of radionuclides reported to be in the water of the Great Lakes. Section 6.3 describes ways to assign a dollar value to the reduction of these health effects. Section 7

estimates the feasibility and dollar cost of reducing the concentration of radionuclides in drinking water drawn from the Great Lakes.

6.2 HEALTH EFFECTS

6.2.1 Introduction

Our discussion concerns the possible harm from ingestion of the radionuclides (Sec. 6.2.2), the probability of harm (risk) per dose (Sec. 6.2.3), and the expected dose per isotope concentration (Sec. 6.2.4).

6.2.2 The Possible Harms

The radionuclides that we believe to be the principal sources of activity in the Great Lakes are ^3H , ^{226}Ra , and ^{90}Sr . After ingestion, ^3H is the source of a smaller dose than either ^{226}Ra or ^{90}Sr (Table 11). The radiological consequences of current and projected ^3H concentrations in the Great Lakes are considered insignificant.⁶ Most ^{226}Ra and ^{90}Sr retained by the body is incorporated in bone. Because of this, we believe that the principal somatic harm that could result from ingestion is bone cancer and leukemia.

A different possible harm is that done to our descendants because of our exposure to ionizing radiation. Although no clinical evidence of such a result is known for humans, laboratory studies of drosophila and mice have led to the belief that ionizing radiation can induce chromosomal and genetic change in humans that will result in some damaged descendants. Because the relationship between radiation-induced mutations and resulting damage is unknown, we and others assume that the nature of the damage to offspring is independent of the mutagenic agent. These results have great variety and range from polydactyl effects to Huntington's chorea. These genetic effects may be expressed many generations after the ancestor's exposure.

6.2.3 The Probability of Harm per Dose

Although there is widespread agreement that cancers may be induced by ionizing radiation, there are no data from which one can unambiguously infer the probability of the incidence of various cancers induced in man from small doses delivered at low rates. Thus we assume, as do others, that risk depends only on the dose and not on the dose rate. Note that the measure of dose, the unit of which is the rem, is itself not entirely free of ambiguity because it is the product of the energy deposited per mass of tissue and a factor called the "relative biological effectiveness", which is sometimes determined after the fact. Our subject is beset by these uncertainties because the low probability of harm can be isolated from other factors only by studying much larger populations for much longer times than has heretofore been possible. In the following paragraphs we summarize the findings of the BEIR Report (Chapter 7) on the relation of ionizing radiation to the risk of leukemia, bone cancer, and genetic damage.

Leukemia

The principal sources of data for adults come from the study of spondylitis patients, females clinically sterilized with X rays, and atomic-bomb casualties. The conclusion is that the data are compatible with a direct proportion between dose and risk. For persons over the age of ten the dose is followed by a latent period of several years and then an elevated risk of leukemia of $1-2 \times 10^{-6}$ per year per rem. For children under ten years of age the elevated risk is in the range $1.6-3.4 \times 10^{-6}$ per year per rem, whereas for exposure *in utero* the elevated risk is 27×10^{-6} per year per rem.

Bone Cancer

The principal sources of data come from studies of persons who have ingested ^{226}Ra and of spondylitis patients. There is some evidence that by assuming a direct proportion between dose and risk one overestimates the real risk from small doses. We prefer to err on the side of safety. Assuming a direct proportion, the data on persons over 20 years of age gives an elevated risk of $0.10-0.55 \times 10^{-6}$ per year per rem, and for persons under 20 years the elevated risk is 0.96×10^{-6} per year per rem.

Genetic Damage

Because the foregoing estimates of risk are imperfectly known, the staff is in a fundamentally more speculative area when it tries to estimate the damage to future generations from the present generation's exposure to ionizing radiation. Any such estimate must be based on experimental work with drosophila and mice together with educated guesses on the relation of this work to humans. We have adopted the BEIR Report estimate of $60-1500 \times 10^{-6}$ damaged descendants per ancestor-rem, of which $12-200 \times 10^{-6}$ occur in the first generation.

6.2.4 Dose per Concentration

In Section 4, our knowledge of the concentrations of various radionuclides in each of the Great Lakes is reviewed. Table 4 displays these concentrations (pCi/l), and Tables 5-9 present our estimates of the resulting dose (urem) to the whole body, bone, and gonads. Two features of Tables 5-9 must be explained. First, the whole-body dose is less than the bone dose. This is because dose depends on where in the body energy is deposited. Because the radionuclides in question are incorporated in bone, and because the whole body is larger than the skeleton, the skeletal or bone dose is higher. Second, the total dose at the end of 50 years, after ingesting lake water for one year, is more than one-fiftieth of the total dose received from ingesting lake water for 50 years, i.e. 50 times the TED_{50} for one-year ingestion is higher than the TED_{50} for 50-year ingestion. This is because the dose is prorated over a long period of time after isotope ingestion. Thus, the risk from drinking Great Lakes water depends on the number of years that a person has already drunk water. Such an analysis would show that the risk per year increases over the adult lifetime of each person.

We overestimate the annual somatic risk by assuming that it has the same value for each year of adulthood, and that this value is the same as the annual risk after 50 years of ingestion at the rate of 2.2 liters/day. The 50-year bone dose per pCi/l from ^{226}Ra is 0.81 rem. The 20- and 10-year doses per are pCi/l are 0.208 rem and 0.074 rem, respectively. We also overestimate the bone dose to a child *in utero* by using the one-year bone dose to an adult, which is 0.002 rem per pCi/l. To estimate the genetic and chromosomal damage, we assume that the dose to the gonads resulting from 30 years of ingestion is relevant. This is estimated to be 0.0076 rem per pCi/l.

6.3 THE VALUE OF RISK REDUCTION

6.3.1 Introduction

Although the previous section describes the risk of deleterious health effects, this information is not sufficient for a discussion of the costs and benefits of the proposed standard. After all, virtually everyone occasionally submits himself to greater risk by driving fast on slippery roads, and some people smoke cigarettes willingly. A pair of fundamental questions are, "Do people wish to reduce their risk?" and, if so, "How much would these people willingly pay to reduce their risk?" These are not the only fundamental questions because the risk also falls on children and the unborn. Because it is widely held that parents are responsible for the protection of their children, the appropriate question is, "How much would a parent pay to reduce a risk to his or her child?" Although parents accept the responsibility for their children's protection, no flat statement can be made for parents' attitudes toward their more remote descendants. Thus, we must ask, "What is the present generation's responsibility to future generations?"

We assume that people would like to reduce the risk to themselves caused by ingestion of radionuclides in water because we are unaware of any pleasure concomitant with ingestion such as occurs with cigarettes or alcohol. We also assume that people would willingly pay something to reduce their risk. "How much" is discussed in Section 6.3.3. We further assume that parents would pay at least as much to protect their children as to protect themselves.

The question, "What is the present generation's responsibility to future generations?" is difficult to answer. The staff does not believe most people would feel obliged to reduce the concentrations of radionuclides in the Great Lakes below those found at the time of their own birth. If this generation accepts a responsibility for the effects of what it has added to the Great Lakes, it must ask how it might discharge this responsibility. As an alternative to simply cleaning up, we might establish a fund out of which our genetically defective descendants would be compensated. If the size of the fund is sufficient to compensate all of our damaged descendants who could hold this generation liable (between 60 and 1500×10^{-6} per ancestor-rem), we must decide on the size of the compensation. Because some genetic defects can incapacitate a person, we might overestimate the compensation by giving each damaged descendant a lifetime stipend equivalent to the median lifetime earnings of this generation.

6.3.2 Relation of Risk to Earnings

Unfortunately, there are few transactions in which the purchase of safety can be distinguished by analysis from other components of the exchange (e.g. choice of car type or home). However, there is some evidence that as occupational risk increases, so do wages. We believe this broad trend is not accidental and that it provides a rough indication of the incremental pay that will induce some persons to bear an incremental risk or, conversely, it provides a rough indication of the lower limit of the incremental wage most people would willingly forgo to avoid a risky job. In the following paragraphs, we summarize the results of three different efforts to study the relation of job risk to wages.

Thaler and Rosen⁷ studied the relation between wages and mortality by occupation for a population of 900 adult-male heads of households from data collected from the 1967 Current Population Survey. The mean age was 41.8 years, the mean number of years of schooling was 10.11, and the mean weekly wage was \$132.65 [\$132.65 (1967 dollars) = \$224.45 (1976 dollars)]. Each of the members of the sample worked in one of 37 occupations, the members of which had a mortality rate that was significantly higher than the population at large. After controlling for other variables, Thaler and Rosen concluded that the incremental annual wage per incremental annual risk of death was:

$$\frac{\$200 \pm 60 \text{ (1967 dollars)}/\text{yr}}{10^{-3} \text{ risk of death}/\text{yr}}$$

which is:

$$\frac{\$338.41 \pm 101.52 \text{ (1976 dollars)}/\text{yr}}{10^{-3} \text{ risk of death}/\text{yr}}$$

Because the sample members had less-than-average earnings we find it plausible that the money they would forgo for risk reduction is less than the population at large would forgo. In other words, we find it plausible that the general population would spend more for risk reduction than would the sample because the general population is wealthier than the sample.

Smith^{8,9} studied the relation between wages and injuries sustained on the job for hourly workers in three-digit manufacturing industries. He found that a risk reduction from 16×10^{-5} death/yr to 8×10^{-5} death/yr resulted in a decrease of 1.5% in annual wage. Because the average wage in his sample was \$8000 (1973 dollars), one can infer that the incremental annual wage per incremental annual risk for his sample was:

$$\frac{\$120 \text{ (1973 dollars)}/\text{yr}}{8 \times 10^{-5} \text{ risk of death}/\text{yr}}$$

which is:

$$\frac{\$153.00 \text{ (1976 dollars)}/\text{yr}}{8 \times 10^{-5} \text{ risk of death}/\text{yr}}$$

Viscuzzi¹⁰ also studied the relation between wages and injuries resulting in death that were sustained on the job for a sample of 496 blue-collar workers whose average annual income in 1969 was \$6809.90 (1969 dollars). The range of risk for these industries is between zero and 10^{-4} death/yr. He fitted his data in several ways. After controlling for other variables, including whether or not the worker judged his job to be dangerous, Viscuzzi found a range from $\$0.48 \times 10^6$ to $\$168 \times 10^6$ (1969 dollars) or:

$$\frac{\$16.65 \pm 9.25 \text{ (1976 dollars)}/\text{yr}}{10^{-5} \text{ risk of death/yr}}$$

The results (rounded to three significant figures) of these three studies of the labor market are:

Study	$\frac{\$ \text{ (1976 dollars)}/\text{yr}}{10^{-6} \text{ risk of death/yr}}$
Thaler and Rosen ⁷	0.34
Smith ^{8,9}	1.91
Viscuzzi ¹⁰	1.67

Each of these figures was reached by studying samples with less-than-average earnings. Thus, we expect the general population to be willing to spend more than these amounts to reduce their own risk.

6.3.3 The Value of Reducing the Concentration of ²²⁶Radium in Drinking Water

In this section, we roughly estimate the amount of money the population would be willing to spend to reduce the concentration of ²²⁶Ra in its drinking water. Let us suppose the typical adult would pay the same annual tax to reduce the risk of harm from ²²⁶Ra as he would to reduce the risk of death. The annual tax is the product of three factors. The first factor is the risk-avoidance premium discussed in Section 6.3.2, which we believe is greater than $\$1.9 \times 10^6/\text{yr}$ per risk of death/yr. The second factor is the risk of leukemia and bone cancer per year per dose, which was discussed in Section 6.2.3. The third factor is the dose per concentration, which was discussed in Section 6.2.4. The product of the three factors is:

$$\frac{\$1.9 \times 10^6/\text{yr}}{\text{risk/yr}} \times \left[(1.1 - 2.6) \times 10^{-6} \frac{\text{risk/yr}}{\text{rem}} \right] \times \left[0.810 \frac{\text{rem}}{\text{pCi/l}} \right]$$

which is:

$$\frac{\$1.69 \text{ to } 4.00/\text{yr}}{\text{pCi/l}}$$

We also suppose that adults will pay as much to protect each of their children as to protect themselves. If half the adults have two children under

20 years of age, the amount spent to protect the child from bone cancer would be:

$$\frac{\$2 \times 0.5 \times 1.9 \times 10^6/\text{yr}}{\text{risk/yr}} \times \left[0.96 \times 10^{-6} \frac{\text{risk/yr}}{\text{rem}} \right] \times \left[0.208 \frac{\text{rem}}{\text{pCi/l}} \right]$$

If one-quarter of the adults have two children under 10 years of age, the amount spent to protect the child from leukemia would be:

$$\frac{\$2 \times 0.25 \times 1.9 \times 10^6/\text{yr}}{\text{risk/yr}} \times \left[(1.6 - 3.4) \times 10^{-6} \frac{\text{risk/yr}}{\text{rem}} \right] \times \left[0.074 \frac{\text{rem}}{\text{pCi/l}} \right]$$

If one-fiftieth of the adults have a child *in utero*, the amount spent for protection from leukemia would be:

$$\frac{\$0.02 \times 1.9 \times 10^6/\text{yr}}{\text{risk/yr}} \times \left[27 \times 10^{-6} \frac{\text{risk/yr}}{\text{rem}} \right] \times \left[0.002 \frac{\text{rem}}{\text{pCi/l}} \right]$$

The total to protect the average adult and his or her children is in the range of:

$$\frac{\$2.18 \text{ to } 4.62/\text{yr}}{\text{pCi/l}}$$

We stress that this can only be regarded as a rough estimate of what the tax-paying population might pay per capita for reduction in the concentration of ^{226}Ra in its drinking water. The roughness of the estimate stems from at least three sources. First, the risk premium has been inferred from the labor market and may be an underestimate. Second, the risk per dose is imperfectly known and may be an overestimate. Third, the dose per concentration is an overestimate because we have basically assumed that the cumulative dose is delivered at the beginning of the relevant period.

We might also consider the relation between the concentration of ^{226}Ra in our water and the number of damaged descendants. In Section 6.2.3, we quoted the range $60\text{--}1500 \times 10^{-6}$ damaged descendants per ancestor-rem and in Section 6.2.4 we found that the 30-year dose per concentration to gonads from ^{226}Ra was 0.0076 rem per pCi/l. This yields $0.46\text{--}11.4 \times 10^{-6}$ damaged descendants per pCi/l to this generation. The fundamental question to which we do not know the answer is, "How much ^{226}Ra did the present generation put in the Great Lakes?"

7. STRATEGIES ON CLEANUP

The staff has not identified a means of cleaning up the lakes by removing radioactive elements. The focus of the effort has been shifted to strategies to remove man-made inputs of radioactive elements to the lakes, and how water may be intercepted for that purpose before consumption.

Even the reduction of man-made inputs to the lakes has a tenuous relationship to the one-millirem TED_{50} objective as it is now stated. This relationship breaks down because: (1) the relative contribution of radioactive sources is not known quantitatively; (2) most of the ^{226}Ra in the

lakes is naturally occurring; and (3) although input sources may be treated to reduce radioactivity, it is not likely that changes in radioactivity levels will be detectable.

Two different approaches are examined in light of attaining a one-millirem objective. Whatever nondegradation is worth, the alternatives are to: (1) eliminate input to the lakes and (2) remove radioactivity from water after it is withdrawn from a lake and before ingestion by man.

7.1 INPUT SOURCES AND CLEANUP EMPHASIS

The principal dose contributors to man from lake radioactivity are: (1) naturally occurring ^{226}Ra and (2) ^{90}Sr fallout. A few other sources, releases from which may be reduced, have been identified. They are: (1) medical and research facilities, (2) nuclear power plants, (3) runoff from phosphate fertilizer, (4) uranium milling in the Elliot Lake (Ontario) area near Lake Huron, and (5) coal-combustion products. (Examination of coal combustion has indicated uranium in slag and combustion products on the order of 0 to 20 ppm.¹¹ Theoretically, leaching from slag heaps could represent a pathway to Great Lakes water.) Nuclear power plants contribute to radioactivity through discharge of ^3H , ^{137}Cs , and ^{90}Sr isotopes, but they are not contributors to the ^{226}Ra activity. Accordingly, control of these facilities would not be a part of a "cleanup" strategy that would reduce TED_{50} significantly.

However, other sources (e.g. ^3H , ^{134}Cs , and ^{90}Sr) are important if the radioactivity objective is shifted from "in-lake" values to an "as-low-as-practicable" objective for radionuclides other than ^{226}Ra . As far as ^{226}Ra is concerned, only the runoff due to uranium mining and the use of phosphate fertilizer are considered further.

7.2 ESTIMATE OF ^{226}Ra RADIUM INPUT FROM FERTILIZER AND PHOSPHATE PRODUCTS

Probable but unmeasured sources of ^{226}Ra to the lakes are the fertilizers used within the basin, which then run into the lakes. Harvested cropland areas in the Great Lakes basin represent about 6% of the U.S. cropland area.^{12,13} However, there is evidence from trace-element studies that many of these elements are retained in the soil and do not contaminate groundwater or surface water.¹⁴ An estimate of phosphorus loss, which may or may not be commensurate with ^{226}Ra loss, is 0.4 lb/acre.^{15,16} If ^{226}Ra is preferentially adsorbed by soil, then the proportional loss of ^{226}Ra in fertilizer would be less than the proportional loss of phosphate.

Another estimate puts annual phosphorus eroded from farms at 1800 million pounds.¹⁷ This might indicate an annual loss of 48 million pounds to the Great Lakes assuming application rates are constant across producing agricultural croplands. Estimates of ^{226}Ra content from normal superphosphate (Table 12) show a level of 2.3×10^7 pCi/ton of fertilizer from the wet process (assuming 50% P_2O_5 content).¹⁸ If the ^{226}Ra content in fertilizer is multiplied by the loss to the basin, about 552 mCi are distributed annually to the lakes. Again, this assumes that ^{226}Ra is lost at the same rate as phosphorus.

Table 12. Estimated ^{226}Ra , Uranium, and Thorium Activity in Phosphate Fertilizer Products and By-Products Based on 1973 Wet-Process Production Data^a (pCi/ton)

Material	^{226}Ra	Uranium			Thorium			
Normal superphosphate	2.3×10^7	b	b	b	b	b	b	b
Triple superphosphate	1.9×10^7	5.1×10^7	2.5×10^6	5.1×10^7	1.1×10^6	8.1×10^5	4.3×10^7	3.5×10^5
Ammonium phosphates	5.2×10^6	5.7×10^7	2.8×10^6	5.7×10^7	1.4×10^6	7.2×10^5	5.9×10^7	3.6×10^5
Phosphoric acid ^c	5×10^5	b	b	b	b	b	b	b
Gypsum	3×10^7	5.5×10^6	2.9×10^5	5.5×10^6	9.7×10^5	1.3×10^6	1.2×10^7	2.4×10^5
Animal feed	5×10^6	b	b	b	b	b	b	b

^a Derived from "Radioactivity Distribution in Phosphate Products, By-Products, Effluents, and Wastes," USEPA, Office of Radiation Programs, ORP/CSD-75-3, August 1975.

^b Data not available.

^c Assuming 50% P_2O_5 content.

Taking the annual fertilizer-loss estimate of 0.4 lb/acre, ^{226}Ra loss (proportional to phosphorus) can be estimated in another manner. Making the conservative assumption that all fertilizer is superphosphate, the basin has 20.7 million acres of productive cropland that would then lose 5.2 million pounds annually. At 2.3×10^7 pCi/ton, only 60 mCi/yr are lost.

A third estimate indicates that phosphorus input to Lake Michigan from erosion is 3.33 million pounds or, because Lake Michigan represents 43% of the drainage basin, 7.74 million pounds for all the Great Lakes.¹⁹ Using the same ^{226}Ra content as above, 90 mCi/yr are lost. Regardless of how much phosphorus as fertilizer ultimately reaches the Great Lakes, much depends on whether the ^{226}Ra is eroded at the same rate, on the fertilizer application rates in the basin, and on the kinds of fertilizers applied.

Assuming the Great Lakes now contain 500 curies of ^{226}Ra in the water (and assuming, for illustration, that all the ^{226}Ra from fertilizer mixes with the water column rather than settles to the bottom) then the highest estimate yields an annual added ^{226}Ra contribution to the lakes from fertilizer of 0.1%.* With these assumptions it would take about 1000 years for the ^{226}Ra content from fertilizer in the lakes to double, assuming no dilution from lake flushing. If it were assumed that inshore areas received ten times the input of ^{226}Ra as the rest of the lakes, it would take about one million years for shore areas to reach the maximum permissible concentration from the addition of fertilizer alone.

* Based on 0.02 pCi/l, which is the average for Lake Michigan.

Higher existing levels of ^{226}Ra in the Great Lakes other than Lake Michigan do not increase the relative significance of fertilizer as a source. With these higher levels, the time span is shorter before the maximum permissible concentration is reached. Thus, the importance of fertilizer depends on how the objective is expressed rather than on the slow rate of change of radioactivity levels.

7.3 REMOVAL OF ^{226}Ra RADIUM FROM PHOSPHATE AND URANIUM PRODUCTS

^{226}Ra from production of fertilizer occurs primarily in the phosphogypsum (about 80%) rather than in the P_2O_5 acid anhydride used to make phosphate fertilizer.¹⁸ Experience in the uranium-mining and -milling industry, water-treatment technology, and phosphate-product industry, indicates a common set of technologies to remove elements such as ^{226}Ra . In the uranium and phosphate industries ^{226}Ra is usually associated with the waste products, or tailings, as a result of the solubilization of the wanted products. ^{226}Ra normally is not dissolved and, depending on the acid and acid concentration used, at least 90% of it remains with the waste products. When it is with waste products, it can be removed from the soluble waste, as it can be removed from water, by treatment with slaked lime.²⁰ Lime neutralization is about 90% effective with heavily contaminated waste products or with relatively pure drinking water. Its chief advantage is low cost. Slaked-lime treatment is used in community drinking systems as well, and thereby can be used both as a process of treating waste effluents before they reach the aquatic environment and as a method to reduce the amount of ^{226}Ra in drinking water. Other chemicals used in acidic removal of radioactivity from waste products are: ammonia, sodium hydroxide, and barium chloride.

The cost of treatment of phosphate and uranium waste products can be partially offset by increasing the recovery rate of wanted products such as uranium. High prices for uranium increase the incentive to improve removal of radionuclides. More expensive treatment includes ion-exchange technology, which has been successfully employed to increase uranium recovery in uranium production and more recently on a pilot basis as a by-product in phosphate production.²⁰⁻²² Removal rates are in excess of 96%.

In those areas where phosphate and uranium are mined, removal of ^{226}Ra from mining wastes may be a cost-effective approach to nondegradation of water quality. The incremental cost to the companies involved may be minimal in new operations, owing to the increasing opportunity to amortize the cost by improved uranium recovery. Water-quality benefits may be derived, but they will be realized only where mining takes place.

However, for the Great Lakes the only direct application would involve uranium-mining activity in the Elliot Lake area, effluents from which may impact Lake Huron: "Canada's major uranium mining operation is situated north of Lake Huron in the Elliot Lake area which drains to the North Channel from Serpent River. This river system is contaminated with ^{226}Ra from tailings pond seepage."²³ The other examples where adsorption in ion-exchange technology may be cost-effective are in the continual management of the closed nuclear-fuel-reprocessing plant in West Valley, New York, and the uranium-refining and UF_6 -production plant near Port Hope, Ontario.²³ In both

cases further study of cleanup technology may be fruitful, but neither case involves a health consequence of the magnitude to affect a Great Lakes radioactivity objective for the whole basin.

The cleanup of effluents from the man-made sources discussed above does not appear to be an effective scheme to reduce the concentration of radio-nuclides in the Great Lakes. This conclusion is based primarily on the relatively insignificant contribution by fertilizer to existing naturally occurring levels of ^{226}Ra . Specific cases, e.g. Lake Elliot, of effluent-discharge treatment may indeed develop as cost-effective cleanup strategies.

7.4 REMOVAL OF ^{226}Ra RADIUM FROM DRINKING WATER

Although the general objective of this study is the examination of the effects of ingestion of "raw" lake water (as a basis for establishing a non-degradation objective), it is also possible to consider the proposed one-millirem objective from the standpoint of treating existing lake-pollutant levels so as to achieve health objectives commensurate with the one-millirem objective. From the viewpoint of cost-effectiveness, the best case for water treatment in terms of health can be made with respect to treating drinking water. Under the assumption that between 90% and 99.9% of all ^{226}Ra can be removed from drinking water, the benefits and costs of such treatment are examined. Removal rates depend on the technology examined; the benefits and costs depend on removal rates and technology costs. The most total benefits are realized by ^{226}Ra removal because the isotope is a major dose contributor to man from ingestion of raw water.

Because ^{90}Sr and ^{226}Ra are chemically similar, ^{90}Sr may be removed along with ^{226}Ra , but it is not clear how much ^{90}Sr will be removed. The removal of ^{226}Ra appears to be the most effective and least costly way to reduce the dose commitment to man, based on its significance as a dose contributor and on known removal processes.

To the extent that the one-millirem objective makes sense on cost-benefit criteria, it will succeed or fail on the ^{226}Ra -removal example. ^{226}Ra removal from Great Lakes drinking water represents the limiting case for a total Great Lakes radioactivity objective for two reasons. The cost of treating water is proportional to the volume of water treated; and by limiting the case to public water systems, the ^{226}Ra reduction is proportional to the reduction in radiation received by man.

7.5 ^{226}Ra RADIUM REMOVAL - 1970 AND 2030

For the purpose of this assessment, it is assumed that slaked-lime or ion-exchange-resin treatment are not now part of any significant portion of community water systems using lakes waters. Such treatment is usually considered a tertiary water-treatment strategy. These two generic treatment strategies are evaluated in terms of two levels of efficiency: 90% ^{226}Ra removal for slaked lime and 99.9% removal for ion-exchange processes. For illustrative purposes, all the benefits from the one-millirem objective are assumed to derive from ion-exchange drinking-water treatment. The calculated TED_{50} received by all people drinking community water each year for 50 years

would range from 0.2 mrem to 445 mrem, depending on treatment efficiency and source of water. This range reflects the best and worst situations that existing evidence may indicate, as well as the uncertainties that are associated with extrapolation from few data.

It may be argued that further treatment of drinking water results in a benefit far in excess of benefits realized by ^{226}Ra removal alone. For the purpose of setting a Great Lakes radioactivity objective, this argument is not addressed. This assessment addresses only the costs and benefits of water treatment for reduction of radioactivity levels. Treatment of water with salt to remove hardness is not considered. The controversial link between softened water and cardiovascular disease is presumed to be real enough as to preclude this option.²⁴ The benefits derived from further treatment are considered only in terms of ^{226}Ra removal rather than reduced hardness in water or the removal of other trace elements.

The view is taken that the benefits of cleanup would increase in the future owing to an increasing population served by the Great Lakes for drinking water. It is assumed that 25 million people are now being served by the Great Lakes for municipal and rural water consumption. This figure would increase to 82 million people by the year 2000 when 90% of the people of the Great Lakes Basin will consume Great Lakes water.^{25,26}

7.6 THE COSTS OF DRINKING-WATER TREATMENT

In examining the incremental cost of ^{226}Ra removal to meet the radioactivity-level standard for drinking water of 5 pCi/l, EPA estimated the annual cost per person.²⁷ The presumed technology was zeolite ion exchange and the assumed efficiency was 97%. To decrease the initial concentration in water from 1 pCi/l to about 0.1 pCi/l, EPA reported the annual cost per person to be \$5.48 in 1975. EPA's primary interest was the cost to small community-water systems, and the economies of scale of construction were not considered in its estimate. Nevertheless, comparison with similar treatment estimates indicate that capital costs are less than 20% of the annual operation cost of ion-exchange processes.^{28,29} A more serious problem is the assumption that only part of the raw water passing through the system had to be treated: "Usually only a fraction of the supply water need be treated since the mixing of treated and untreated water is acceptable abatement procedure."³⁰ Using the EPA formula for estimating how much water must be treated to reduce the level from 1 pCi/l to 0.001 pCi/l, it can be shown that all the municipal water from the Great Lakes must be treated.

Several studies on ion exchange indicate the cost of treatment. Based on studies of several ion-exchange-process alternatives for 12 communities, a range of costs of 29 cents to \$1.25 per thousand gallons of water treated was reported.²⁸ All of the communities examined had extremely poor, brackish, municipal-water supplies. The primary purpose was to remove the hardness and sodium from the water. A slightly lower capital cost was estimated for ion-exchange equipment for removal of ^{226}Ra for plants with a capacity of five to ten million gallons per day.³¹ The annual cost estimated by Bresler and Miller ranges from 30 to 40 cents per year per 1000 gallons treated at a 90% on-stream rate.³²

At 30 cents per 1000 gallons for a treatment of three million gallons per day, the cost to a small community would be about \$330,000 per year. If such a community consumed 100 gallons per person per day then treatment would cost no less than \$11.00 per person per year. The less expensive approach using lime-soda ash would indicate the lower bound on treatment cost. The city of Owosso, Michigan, spent about \$76,000 in 1976 on lime for its 2.63-million-gallon-per-day plant.³³ At a consumption of 100 gallons per day, lime costs were about \$3.00 per person excluding associated labor costs and depreciation.

The total costs of treatment are estimated using a variety of assumptions as given in Table 13.

Total annual costs range from \$160 million per year to over \$2.19 billion per year depending on assumptions. Obviously, a large proportion of the treated water would never actually be ingested.

Another approach to water treatment is to have all households invest in home water conditioners. This would cost in the range of \$1.4 billion dollars per year at current water-softener prices.

Total cost to remove 90% of the ²²⁶Ra prior to ingestion would be expected to be no less than about \$160 million each year. However, this cost could be about \$2.2 billion per year to demonstrate progress toward the one-millirem objective for water that would actually, or potentially, be consumed on a daily basis by residents of the Great Lakes basin.

Table 13. Total Treatment Costs per Year for the Great Lakes
(millions of dollars)

Water Treatment	Process			
	Lime-Soda Ash		Ion Exchange	
	At 10¢ per 1000 gallons	At 20¢ per 1000 gallons	At 30¢ per 1000 gallons	At 60¢ per 1000 gallons
1. Treatment of all U.S. municipal water supplied at current capacity of 7409 Mgd	270.4	540.8	811.2	1622.4
2. Treatment of all U.S. municipal consumption (1970) of 4356 Mgd	159.0	318.0	477.0	954.0
3. Treatment of all Canadian municipal and rural water use (1965) of 887 Mgd	32.3	64.6	96.9	193.8
4. Treatment of all Canadian municipal and rural water use (2030) of 2592 Mgd	94.6	189.2	283.8	567.6
5. Total in the year 2030 (lines 1 plus 4)	365.0	730.0	1095.0	2190.0

The cost of treatment depends on the efficiency of the treatment processes, which are assumed to range from 90% to 99.9%. At this time, the staff could not determine the precise efficiency of any given ion-exchange technology. Consequently, it must be assumed, at this time, that it may cost \$2.2 billion per year to achieve 90% efficiency. This is a most conservative assumption, given that the scale of operation has no precedent and would require indirect costs of large magnitude. Also, efficiency would be difficult to maintain over hundreds of individual operations.

The ^{90}Sr concentration appears similar in all the Great Lakes and ranges from 0.6 to 1.0 pCi/l. This concentration produces a TED_{50} for whole body of 0.7 to 1.2 mrem, and the corresponding dose to bone is 7-12 mrem. Thus, even if ^{226}Ra were removed completely, further reduction of ^{90}Sr would be required to achieve a one-millirem objective for whole body or a six-millirem objective for bone. The only feasible way to achieve a one-millirem objective is by an ion-exchange process, which could remove up to 90% of both ^{226}Ra and ^{90}Sr .

Estimates of the value and cost of removing ^{90}Sr from drinking water such as those reported above for ^{226}Ra , should be performed. It is assumed that the value of ^{90}Sr removal will be of the same order of magnitude as that of ^{226}Ra removal:

$$\frac{\$2.18 \text{ to } 4.62/\text{yr}}{\text{pCi/l}}$$

The staff believes that the relatively inexpensive (slaked-lime) ^{226}Ra -removal process will remove some of the ^{90}Sr and that the relatively expensive (ion-exchange) ^{226}Ra -removal process may remove as much as 99% of the ^{90}Sr . If removal of both ^{226}Ra and ^{90}Sr by the slaked-lime process achieves a high degree of efficiency (J% or more), then there may be some value to examining thoroughly the costs and benefits of treating drinking water. Further consideration would include a closer examination of all the elements of health risk, technological efficiency, and possible side effects associated with treatment processes.

8. COST-BENEFIT SUMMARY

This section summarizes the considerations in Sections 6 and 7. The most cost-effective method of reducing the dose resulting from ingestion of drinking water drawn from the Great Lakes is to reduce the concentration of radionuclides in the water after it is withdrawn and before ingestion. The most important sources of harm appear to be ^{226}Ra and ^{90}Sr . A rough estimate of the cost and efficiency of removing ^{226}Ra has been made. The degree to which the processes of ^{226}Ra removal will also remove ^{90}Sr has not been estimated. A rough estimate of the annual amount of money Americans might be willing to spend to reduce the concentration of ^{226}Ra in their drinking water in order to reduce the projected somatic harm to themselves and their children has been made also.

The result of these estimates is that a 90% reduction in the concentration of ^{226}Ra could be achieved for $\$160 \times 10^6/\text{yr}$ to $\$2200 \times 10^6/\text{yr}$ and that this range is not very dependent on the population (25×10^6 in 1970 or

80×10^6 in 2030) to be served. It also appears possible to reduce the concentration of ^{226}Ra by 99.9%, but this cost may be at the high end of the range because of the vast scale of operations that would be necessary relative to existing experience. Assuming that two-thirds of the population are adults, this implies an annual range of costs per adult between \$9.60 and \$132.00 for a 90% reduction in ^{226}Ra for the 1970 population of 25×10^6 persons and an annual range per adult of \$3.00 to \$41.25 for a population of 80×10^6 persons. This 90% reduction in ^{226}Ra concentration would result in a change of 0.018 pCi/l in drinking water drawn from Lake Michigan and changes of 0.9 to 2.7 pCi/l, 0.9 to 4.95 pCi/l, and 0.027 pCi/l in drinking water drawn from Lakes Huron, Ontario, and Erie, respectively. The reader should bear in mind that the staff does not know the concentration of ^{226}Ra in Lake Superior. Because the range of \$1.69 to \$4.00/yr per pCi/l was designated as a rough estimate of what the persons affected would spend to reduce the concentration of ^{226}Ra in their drinking water, one could infer that a ^{226}Ra -reduction goal for Lake Michigan would not win widespread appreciation. However, such a program for removal is not ruled out on cost-benefit grounds for Lakes Huron and Ontario. It is emphasized that the staff's considerations are only approximate in nature, should be further refined, and have been restricted to the three sets of issues described in Section 6.1.1. Consideration should also be given to the issues not covered by this report, which were also cited in Section 6.1.1, before drawing conclusions based on cost-benefit considerations. Also, the reader should bear the following questions in mind: "What are the concentrations of radionuclides in Lake Superior?" "What is the likelihood that the Great Lakes will be used to supply untreated drinking water?" "How does the radiation that would be received from this water compare to other natural and man-made sources?"

To place these questions in context for further consideration, Table 14 summarizes average radiation dose to man from a variety of sources. Additional radiation received from ingestion of 2.2 liters of raw water would add 2-4 mrem/yr from ^{226}Ra and 1.2 mrem/yr from ^{90}Sr . This additional ^{226}Ra is based on the assumption of an average concentration in lakes waters of 1 pCi/l.

Further considerations are the standards that already exist specifically for ^{226}Ra . These standards are expressed in activity levels (pCi/l) rather than dose effects. Standards are shown in Table 15, and these can be compared with the "in-lake" levels given in Table 4.

It appears that acceptance of the one-millirem objective implies a need for actions to reduce current levels of ^{226}Ra activity in the lakes. Before such actions are undertaken, the foregoing fundamental questions should be examined. A better baseline radioactivity condition must be defined. A better view is required of natural vs. man-made source contributions to the lakes.

9. RECOMMENDATIONS FOR FURTHER ASSESSMENTS BY THE DIVISION OF TECHNOLOGY OVERVIEW (DTO)

The DTO request for a short completion time placed clear limits on the completeness with which aspects of the study could be treated. Moreover, as the study progressed, we identified questions and problem areas beyond the original scope of the study. This section deals briefly with these matters.

Table 14. Sources of Whole-Body Doses

Radiation	BEIR ^a (mrem/yr)	Pochin ^b (mrem/yr)	UNSCEAR ^c (mrad/yr)
Natural			
Cosmic	44		
Radionuclides in body	18		
External gamma	40		
Subtotal	102	100	89
Man-made			
Medical and dental	73	35	
Fallout	4	6	160 ^d
Nonnuclear occupational exposure	0.8	0.3	
Nuclear power (1970)	0.003	6	0.009
Nuclear power (2000)	< 1		
Other miscellaneous sources	2	0.3	
Approximate total	182	148	e

^a National Academy of Sciences, "The Effects on Populations of Exposure to Low Levels of Ionizing Radiation," p. 38, Washington, DC, 1972.

^b E. E. Pochin, "Estimated Population Exposure," pp. 20 & 31, for Organization for Economic Cooperation and Development, Paris, 1975.

^c United Nations Scientific Committee, "Ionizing Radiation: Levels and Effects," Vol. I: Levels, Table 76, p. 109 and paragraph 406, p. 73, United Nations, New York, 1972.

^d Due to all weapons tests conducted prior to 1971. Subject to a decay function representing about two years of natural exposure.

^e Not applicable.

Topics not specifically considered were:

- (1) detailed demography and computation of population-dose commitments (man-rem) for persons drinking water from each lake,
- (2) quantitative identification of specific source terms (i.e. input sources of radioactivity),
- (3) the acceptance of existing levels of radioactivity as a reasonable new base for nondegradation, and
- (4) costs for monitoring programs in support of a one-millirem objective.

Table 15. ^{226}Ra Radium Activity Levels (pCi/l)

<i>Water Systems and Lakes</i>	<i>Level</i>	<i>Source</i>
Range of values in public supply systems	0.1-60	EPA, 40 FR 34325, 14 August 1975
Recommended standard for drinking water by interstate carriers	3.0	U.S. Public Health Service, "Drinking Water Standards," Publ. 956, 1962
Maximum contaminant levels in community water systems	5.0	EPA, 41 FR 28404, 9 July 1976
NPDES-permit limitation for soluble disposal by release into sanitary sewage systems	30.0	10 CFR 20, Appendix B, Table II
Lake Michigan	0.02	Table 2
Lake Ontario	< 1.0-5.5	Table 4
Lake Erie	0.03	Table 4
Lake Huron	0.1-3.0	Table 4

Further consideration should be given to:

- (1) development of specific protocols for isotope-monitoring programs,
- (2) determination of the accuracy of the reported isotope concentrations in Great Lakes waters,
- (3) prediction of the impact of reducing input from controllable sources to radioactivity concentrations in Great Lakes waters,
- (4) more detailed cost-benefit analysis concerning costs of ^{226}Ra and ^{90}Sr removal, and
- (5) epidemiological studies of populations that are known to ingest levels of ^{226}Ra in excess of those reported for the Great Lakes.

10. FUTURE RESEARCH NEEDS

It is clear from the data presented in this report that we know far more regarding the behavior of the fission products and transuranic elements in the Great Lakes than we do about the radionuclides in the natural uranium and thorium series. It is imperative that a major effort be made to describe the sources, transport, and fate of ^{226}Ra in the Great Lakes. Considering that reported values for ^{226}Ra in lake waters vary between 0.02 and 5 pCi/l, and

that even at the lowest concentration the TED_{50} will be within a factor of two of the proposed standard of one millirem, top priority should be placed on experiments to determine the concentration of ^{226}Ra in lake waters and to assess the temporal and spatial variations.

Although it is well known that ^{210}Pb (another daughter in the ^{238}U decay series) is concentrated in surficial sediments, due to natural fallout following decay of ^{222}Ra in the atmosphere, few observations have been made of the distribution of ^{226}Ra in the sediments. It is well known that ^{226}Ra can enter the lakes as a result of mining and milling operations, runoff from agricultural fields after the application of phosphate fertilizers, and wet and dry deposition of airborne particulates released in the burning of coal. Because little is known regarding the biogeochemistry of ^{226}Ra , studies should be initiated to determine its pathways in the aquatic environment and its availability to organisms through the water itself and the food chain. Such studies should include the determination of the chemical form of ^{226}Ra and other natural radionuclides and the relative importance of biotic and abiotic processes, such as uptake by phytoplankton and inorganic particulate matter in the water column, in their transport and fate within the system. A full knowledge of such processes is essential if we are to be able to predict the resulting concentrations of these radionuclides in drinking waters.

11. SUMMARY

Current measurements indicate that naturally occurring ^{226}Ra and ^{90}Sr from fallout are the greatest contributors to radiation dose for persons drinking water from the Great Lakes. Levels of radioactivity in Great Lakes waters are neither well documented nor known accurately. Therefore, a large margin of uncertainty must also be attached to the total equivalent dose-50 years (TED_{50}) estimates provided. The radioisotope data for Lake Superior were insufficient for meaningful TED_{50} calculations. TED_{50} estimates for water from the other lakes are in excess of the one-millirem objective for whole body or other critical organs. TED_{50} estimates for gonads also exceed 1 mrem for Lakes Huron, Ontario, and Erie if the upper range of reported isotope concentrations is used for dose calculations. Doses to man that result from ingestion of water from the Great Lakes are not expected to decline by 2050. Because consumption for only one year is considered, the TED_{50} criterion underestimates the potential health effects that may result from ingestion of lake water. Reduction of controllable isotope releases from the sources would not be expected to reduce TED_{50} appreciably. TED_{50} could be influenced by treating water after its removal from the lakes. Cost-benefit considerations support the policy of ^{226}Ra and ^{90}Sr removal, and the concomitant reduction of dose to man, through treatment of water before ingestion. Although this procedure is considered cost-effective in an overall sense, large expenditures would be necessary to achieve the one-millirem objective after ingestion of waters from Lakes Huron, Ontario, and Erie. Such treatment of water from Lake Michigan could reduce TED_{50} to less than 1 mrem for whole body and gonads and to less than 6 mrem for bone. Adoption of the revised water-quality objective of one millirem for Great Lakes water is premature and should be delayed until the levels of radioactivity in Great Lakes waters are established more accurately. Further assessment of the efficiency of ^{226}Ra and ^{90}Sr removal and a more comprehensive cost-benefit analysis are necessary.

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APPENDIX A

The generalized equation that determines the activity of a radioisotope in an organ of reference after ingestion is:

$$q(t) = pCfH(t)$$

where:

t = time after ingestion,

p = amount of food or liquid consumed per day,

C = concentration of radioisotopes in substance consumed,

$f = f_1 f_2$,

f_1 = fraction of isotope that enters bloodstream from the intestinal system,

f_2 = fraction of isotope that passes from blood to organ of interest (this constant depends on the retention function used), and

$H(t)$ = fraction of isotope remaining in organ of interest (this includes both the biological retention function and physical-decay properties).

EXPONENTIAL RETENTION FUNCTION

During an interval of continuous ingestion, the activity present in the organ of interest is determined by:

$$\frac{dq}{dt} = pCf_w - \lambda q$$

where:

$f_w = f_1 f_2'$,

f_1 = fraction of radioactivity that reaches the bloodstream from the intestinal tract,

f_2' = fraction of f_1 activity that reaches the organ of interest, and

λ = effective decay constant that determines residency time in organ of interest;

and:

$$\lambda = 0.693 \left[\frac{T_b + T_p}{T_b T_p} \right]$$

where:

T_b = biological half-life for specific organ, and

T_p = physical half-life.

Thus the activity in the organ after time t is:

$$q(t) = \frac{PCf_w}{\lambda} \left[1 - e^{-\lambda t} \right]$$

The integrated organ burden (activity $\times$ time) over the interval of consumption is:

$$Q(t) = \frac{PCf_w}{\lambda} \left[t + \frac{1}{\lambda} (e^{-\lambda t} - 1) \right]$$

For the time interval following termination of injection the activity in the organ of interest is:

$$q(t) = q(t_0) e^{-\lambda t}$$

where:

$q(t_0)$ = activity at the time injection terminates.

The integrated organ burden in activity-days is:

$$Q(t-t_0) = \frac{q(t_0)}{\lambda} \left[1 - e^{-\lambda(t-t_0)} \right]$$

POWER RETENTION FUNCTION

Frequently, the complicated metabolic pathways yield retention characteristics that are best described by several independent first-order exponential terms. It has been demonstrated by Norris et al.* that such a sum of exponentials can be represented over a large interval by a single power function. This function has the form:

$$R(t) = At^{-n}$$

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*Reference 5 of the main text.

where:

$R(t)$ = fraction of activity remaining after time t , and

A = fraction of isotope remaining after the first day following injection.

The instantaneous rate of excretion is then:

$$\frac{dR}{dt} = -\frac{n}{t} R \quad \text{for: } t \geq 1$$

Including the physical-decay process, the activity following a single injection is:

$$\begin{aligned} \frac{dq}{dt} &= -\frac{n}{t} q - \lambda_p q & \left(\lambda_p = \frac{0.693}{T_p} \right) \\ q(t) &= q_0 t^{-n} e^{-\lambda(1-t)} & \text{for: } t \geq 1 \\ q_0 &= pCfA \end{aligned}$$

where:

$$f = f_1 f_2,$$

f_1 = fraction of injected isotope that transfers from intestinal tract into bloodstream and is eventually excreted according to $R(t)$, and

f_2 = fraction of isotope retained according to $R(t)$ that is associated with the organ of interest.

The body burden resulting from daily ingestion over m days and integrated over a total interval of M days is:

$$Q(t) = (M-m) \sum_{t=1}^{M-m} q(t) + \sum_{t=1}^m (m+1-t) q(M-m+t)$$

When the physical half-life is much greater than the biological retention time, the integrated burden for instantaneous uptake can be solved directly:

$$Q(t) = q_0 \frac{t^{1-n}}{1-n}$$

It should be emphasized that the exponential retention function contains an effective decay constant associated only with a specific organ. Thus, f refers to the fraction transferred directly to that organ. The power function refers to the retention in the whole body. The fraction f refers only to that remaining whole-body portion located at or associated with the organ of interest. For example, 30% of the ingested ^{226}Ra is transferred from the GI

tract to blood and subsequently excreted according to a power function. Almost all of the retained ^{226}Ra (99%) is associated with exchange processes in the bone.

A more detailed retention function for alkaline-earth elements has been proposed in ICRP-20. We have compared the bone-dose commitment as computed from this model for ^{226}Ra with the simple power function and obtained results that compare within 15%.

DOSE CALCULATION

The following relationship is used to convert the time-integrated burden $Q(t)$ to dose (from ICRP-II):

$$\text{Dose (mrem)} = \frac{\text{EFRBE} \times 51.2}{g} Q$$

where:

EFRBE = Effective energy (MeV) of all the decay products adjusted according to appropriate quality factors,

Q = organ burden in $\mu\text{Ci-days}$, and

g = mass of organ in grams.

Table A lists the parameters used in our dose calculations.

Table A. Parameters Used for Dose Calculations

Nuclide	Organ	f_1	f_2'	f_2	f	T_b (days)	T_p (days)	EFRBE	Mass (g)
^3H	WB ^a	1.0	1.0		1.0	12	4,500	0.01	70
^{137}Cs	WB	1.0	1.0		1.0	70	11,000	0.59	70
^{90}Sr	WB	0.3	1.0		0.3	13,000	10,000	1.1	70
^{90}Sr	Bone	0.3	0.3		0.09	18,000	10,000	5.5	7
^{90}Sr	Bone	0.3		0.99	0.3	$0.95t^{-0.25}$	10,000	5.5	7
^{226}Ra	WB	0.3	1.0		0.3	900	590,000	110	70
^{226}Ra	Bone	0.3	0.13		0.04	16,400	590,000	110	7
^{226}Ra	Bone	0.3		0.99	0.3	$0.54t^{-0.52}$	590,000	110	7
^{234}U	Bone	0.0001	0.11		1.1×10^{-5}	300	9.1×10^7	240	7
^{234}U	Bone	0.0001		0.85	8.5×10^{-5}	$0.72t^{-0.8}$	9.1×10^7	240	7
^{238}U	Bone	0.0001	0.11		1.1×10^{-5}	300	1.6×10^{12}	220	7
^{238}U	Bone	0.0001		0.85	8.5×10^{-5}	$0.72t^{-0.8}$	1.6×10^{12}	220	7
^{239}Pu	Bone	3×10^{-5}	0.8		2.4×10^{-5}	73,000	8.9×10^5	270	7
^{239}Pu	Bone	3×10^{-5}		0.9	2.7×10^{-5}	$0.99t^{-0.01}$	8.9×10^5	270	7

^aWhole body.