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Considerations on a *De Minimis* Dose and Disposal of Exempt Concentrations of Radioactive Wastes

D. C. Kocher
F. R. O'Donnell

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CONSIDERATIONS ON A *DE MINIMIS* DOSE AND DISPOSAL OF EXEMPT CONCENTRATIONS OF RADIOACTIVE WASTES

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CONSIDERATIONS ON A DE MINIMIS DOSE AND DISPOSAL OF EXEMPT CONCENTRATIONS OF RADIOACTIVE WASTES

D. C. Kocher and F. R. O'Donnell

ABSTRACT

This report considers a generally applicable *de minimis* radiation dose for members of the general public and its application to determining exempt concentrations of radioactive wastes for purposes of disposal. The concept of a *de minimis* dose is reviewed in relation to limits on acceptable dose from all sources of exposure, limits on dose from specific practices, and application of the ALARA principle to reductions of public exposures. On the basis of current recommendations of radiation protection authorities, we propose as a generally applicable *de minimis* dose for members of the general public (1) a principal limit on annual committed effective dose equivalent averaged over a lifetime of 0.01 mSv (1 mrem) and (2) a subsidiary limit on committed effective dose equivalent in any year of 0.05 mSv (5 mrem). We then review existing methodologies for using a *de minimis* dose to derive exempt concentrations of radionuclides in solid wastes for purposes of disposal, and we present a methodology that was developed for application to disposal on the Oak Ridge Reservation. Discussion of these methodologies emphasizes difficulties associated with their application to determining exempt concentrations of uranium-bearing wastes. First, it is illogical to use an annual *de minimis* dose of 0.01 mSv (1 mrem) to determine exempt concentrations of uranium when natural uranium in its undisturbed state leads to annual doses much greater than the *de minimis* value. Second, because of the very long half-lives of uranium isotopes, doses to an inadvertent intruder from the drinking water pathway are likely to be important at any non-arid, near-surface disposal site, but there is considerable uncertainty in predicting dose from this pathway due to the site-specific nature of geohydrologic and geochemical conditions that determine mobilization and transport of uranium in water. Third, over long time periods, the buildup of ^{226}Ra and daughter products from uranium decay can greatly increase potential doses to intruders if the uranium in the disposal facility remains immobile. These issues are discussed with reference to a proposal for an exemption level for uranium in solid wastes of 30 pCi/g. Finally, we briefly discuss existing methods for measuring the uranium content in bulk solid wastes, because such measurements probably will be required in exempting uranium-bearing wastes for disposal.

30 pCi/g = 1.11 Bq/g see p 15

1. INTRODUCTION

Operations at U.S. Department of Energy (DOE) facilities in Oak Ridge, Tennessee, routinely generate large volumes of solid waste materials that may contain relatively small amounts of radioactivity. Procedures for disposal of waste materials must include consideration of the most appropriate means of disposal of such low-activity wastes.

It is widely accepted within the radiation protection community that there are levels of radioactivity so low as to be below regulatory concern (e.g., see ref. 1); i.e., potential risks from radiation exposures that might result from unrestricted use of such low-activity materials generally would be of no concern to members of the public. These so-called "exempt" quantities of radioactive materials then could be handled in all respects as if they were nonradioactive; e.g., solid wastes containing exempt concentrations of radionuclides could be treated as ordinary trash and placed in a sanitary landfill rather than in a disposal facility for low-level radioactive wastes.

DOE Order 5820.2 assigns to the heads of DOE field organizations the responsibility for establishing so-called "threshold" concentrations of low-level radioactive wastes that would require disposal by shallow-land burial or greater confinement disposal.[2] Concentrations of radionuclides below the threshold values then would be acceptable for disposal in a sanitary landfill. The primary advantages of establishing threshold concentrations of radionuclides in solid waste materials include (1) significant reductions in the required capacity of radioactive waste storage and disposal facilities, (2) reductions in costs associated with storage and disposal of radioactive materials, and (3) a potential increase in recycling or public sale of waste materials. A reduction in the volume of wastes that are placed in a radioactive waste-disposal facility also may lead to significantly improved long-term performance of the facility, e.g., by reducing the potential for subsidence of the trench cap or infiltration of water through the trench.

In this report, we generally do not use the term "threshold" adopted by the DOE in describing concentrations of radionuclides in solid wastes that are below regulatory concern,[2] because this term has the connotation that there is a threshold dose below which the risk of radiation-induced health effects (i.e., fatal cancers or genetic defects) is zero. Rather, we usually will use the term "exempt" to describe quantities of radionuclides that are below regulatory concern, because the risk from exposure to exempt materials, while very small, may be nonzero.

The primary purpose of this report is to present a methodology relating radiation dose and exempt concentrations of radionuclides in solid waste materials for disposal on the Oak Ridge Reservation. The

outputs of the methodology are estimates of radiation dose per unit concentration of radionuclides in the waste for individuals who might intrude inadvertently onto the waste disposal site following loss of institutional controls over the facility. We focus on postulated exposure scenarios for an inadvertent intruder, because doses to an intruder are expected to be larger than doses resulting from off-site exposures of members of the general public. The dose-assessment methodology then can be used in two ways: (1) to derive exempt concentrations of radionuclides if a dose that is below regulatory concern for waste disposal has been established or (2) to estimate doses from exposure to exempt concentrations of radioactive wastes if the latter have been determined by some means other than establishment of a radiation dose that is below regulatory concern.

In this report, we apply the methodology relating radiation dose and concentrations of activity in solid wastes to the radionuclides that are anticipated for disposal in the proposed Central Waste Disposal Facility (CWDF) in Oak Ridge.[3] We particularly emphasize application of the methodology to uranium-bearing wastes from the Y-12 Plant in Oak Ridge, because of the need to save currently available space for disposal of low-level radioactive wastes at that site prior to development of the CWDF or an alternative storage or disposal facility.

The remainder of this report is organized as follows. Section 2 discusses such radiation-protection concepts as (1) limits on acceptable dose from all sources of exposure, (2) limits on dose from specific practices, (3) application of the ALARA (As Low As Reasonably Achievable) principle for reducing public exposures, and (4) a generally applicable *de minimis* dose, and the relation of these concepts to the determination of exempt concentrations of radionuclides in solid waste materials. Section 3 then presents a review of current efforts by national and international authorities to establish a generally applicable level of radiation dose that would be below regulatory concern, i.e., a *de minimis* dose, and presents a recommendation for such a dose. The establishment of a generally applicable *de minimis* dose would provide the most defensible means of deriving exempt concentrations of radionuclides in solid waste materials. Section 4 briefly describes the methodologies that have been developed by various investigators for relating doses to inadvertent intruders at a waste-disposal site to exempt concentrations of radionuclides in the wastes. This section also includes an evaluation of the concentration limit that has been proposed for determining exempt concentrations of uranium-bearing wastes at the Y-12 Plant in Oak Ridge. Section 5 presents a summary description of the methodology that we have developed for relating doses and exempt concentrations of radionuclides for disposal on the Oak Ridge Reservation, and presents a comparison of

the results of this methodology with the results obtained from the other methodologies reviewed in Section 4. In this comparison, we again emphasize the estimation of exempt concentrations of uranium in solid wastes. The methodology developed for the Oak Ridge site is presented in detail in Appendix A of this report. Section 6 presents a brief discussion of measurement techniques and associated costs that might be used to determine exempt concentrations of uranium in large volumes of solid waste materials. Finally, Section 7 presents a summary of this work and the conclusions obtained from the analyses.

2. CONCEPTS IN RADIATION PROTECTION

2.1 Introduction

This section presents a discussion of fundamental concepts in radiation protection that is intended to provide an understanding of the significance of exempt concentrations of radionuclides in solid waste materials and the bases for establishing such concentrations. The concepts discussed here include (1) limits on acceptable dose for members of the general public from all sources of exposure, (2) limits on dose from specific practices, (3) application of the ALARA principle for reducing public exposures to specific sites and specific practices and the relationship between ALARA and exempt quantities of radioactivity, and (4) a generally applicable *de minimis* dose. These concepts and their interrelationships are depicted in Fig. 1, which is adapted from ref. 1.

The fundamental goal of radiation protection is limitation of risks from radiation exposure, and this goal normally is accomplished by means of a system of dose limitation. As recommended by the International Commission on Radiological Protection (ICRP), for example, the system of dose limitation has three requirements: (1) justification of the practice, (2) optimization of exposures, and (3) limitation of dose to maximally exposed individuals.[4] Justification means that no practice involving radiation exposures shall be adopted unless it produces a positive net benefit. Optimization of exposures means that all exposures (i.e., the population dose) shall be kept As Low As Reasonably Achievable (ALARA), economic and social factors being taken into account. For releases of radioactivity to the general environment, limitation of dose to individuals means that the dose equivalent to members of the general public from all sources, excluding natural background radiation and deliberate medical practices, shall not exceed specified limits, except under unusual circumstances, regardless of the cost of control measures that would be required to meet the limit.

2.2 Limits on Acceptable Dose from All Sources

For exposures of the general public, dose limits from all sources, excluding natural background and deliberate medical practices, are intended to represent a limit on incremental risk that is "acceptable" to most individuals. This level of risk is estimated on the basis of risks from other activities that are widely accepted by the general public. The top horizontal line in Fig. 1 represents the limit on acceptable dose to any member of the general public from all sources of exposure.

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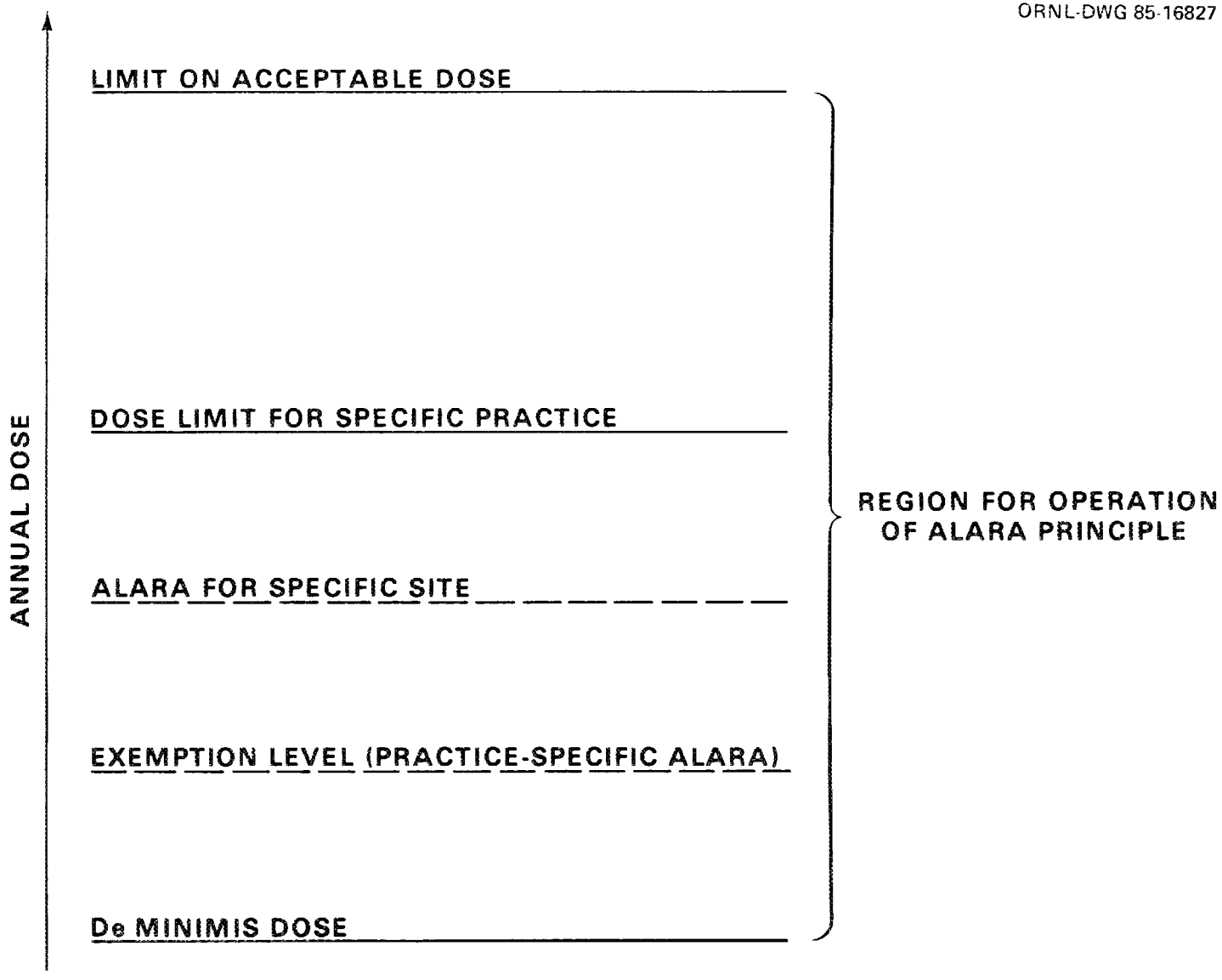

Fig. 1. Relationships between limit on acceptable dose, dose limits for specific practices, doses resulting from application of ALARA principle to specific sites and specific practices, and *de minimis* dose. The vertical scale for dose is arbitrary.

In the U.S., the limit on acceptable dose for members of the general public from all sources of exposure is established in radiation protection standards of the Nuclear Regulatory Commission (NRC), and currently is an annual dose equivalent from uniform whole-body irradiation of 0.5 rem (5 mSv).[5] In proposed revisions of these standards, the limit would become an annual dose of 0.5 rem (5 mSv) which is the sum of the committed effective dose equivalent from internal exposure and the dose equivalent to whole body from external exposure.[6] The committed effective dose equivalent, as defined in ICRP Publication 26, is a weighted sum of committed dose equivalents for several different body organs and tissues.[4]

The DOE establishes dose limits for its operations similar to those of the NRC. Current standards for DOE operations include limits on annual dose equivalent of 0.5 rem (5 mSv) for occasional exposures (i.e., exposures that are temporary in nature and will not continue for more than 5 years) and 0.1 rem (1 mSv) for prolonged exposures.[7] These limits apply to the sum of the committed effective dose equivalent from internal exposure and the effective dose equivalent from external exposure.

The radiation protection standards for DOE operations[7] are similar to current recommendations of the ICRP which include two dose limits:[9] (1) a principal limit on annual committed effective dose equivalent averaged over a lifetime of 1 mSv and (2) a subsidiary limit on committed effective dose equivalent in any year of 5 mSv. A similar two-tiered dose-limitation system, i.e., limits on annual committed effective dose equivalent of 0.1 rem (1 mSv) for continuous exposures and 0.5 rem (5 mSv) for occasional exposures, is being considered by the National Council on Radiation Protection and Measurements (NCRP).[9,10]

A useful point of reference for dose limits for public exposures is provided by levels of natural background radiation. The average annual committed effective dose equivalent from natural background in the U.S., including contributions from radon daughter products during indoor residence which are about half of the total, is about 0.2 rem (2 mSv).[11,12]

2.3 Limits on Dose from Specific Practices

Regulatory authorities in the U.S. have established generally applicable limits on dose from specific practices that often are well below the limit on acceptable dose from all sources. The standards for specific practices essentially represent a judgment by the regulatory authorities that the dose limit is "reasonably achievable," taking into account costs of achieving the limits with available technologies and associated reductions in health risks to the general public. Thus, the

dose limits may be viewed as an application of the ALARA principle to standard setting itself.

The generally applicable dose limit for a specific practice is represented by the second horizontal line in Fig. 1. This dose limit could apply, for example, to all low-level waste disposal facilities, and is a limit not to be exceeded for that practice at any site.

For practices that do not primarily involve naturally occurring radionuclides, many of the current standards of the NRC and the U.S. Environmental Protection Agency (EPA) specify a limit on annual dose equivalent to whole body of 25 mrem (0.25 mSv) - i.e., (1) the EPA's standards for various operations of the uranium fuel cycle,[13] operations of facilities for management of spent fuel, high-level and transuranium wastes,[14] operations of thorium processing facilities,[15] and airborne emissions of radionuclides,[16] and (2) the NRC's standards for near-surface disposal of radioactive wastes.[17] The DOE also has established an interim limit on annual dose equivalent of 25 mrem (0.25 mSv) as guidance for developing new disposal facilities for low-level radioactive wastes.[18]

It is important to note, however, that the dose limit that is judged reasonably achievable may vary from one practice to another. For example, the EPA's remedial action standards for inactive uranium processing sites[15] set limits on radium concentrations in soil, indoor radon levels, and indoor gamma radiation that correspond to annual dose equivalents considerably in excess of 25 mrem (0.25 mSv),[19] whereas the EPA's interim standards for radioactivity in drinking water contain a limit on annual dose equivalent to whole body or any organ from man-made, beta/gamma-emitting radionuclides of 4 mrem (0.04 mSv).[20]

2.4 Application of the ALARA Principle to Public Exposures

The ALARA principle involves a balancing of reductions in population dose with the increased costs required to achieve such reductions,[4] and application of the ALARA principle to specific practices may reduce doses to individuals below established limits for those practices provided it is cost-effective to do so. Use of the ALARA principle for controlling public exposures is specified in the NRC's current and proposed radiation protection standards,[5,6] and also is specified in current standards for the design of nuclear power plants[21] and the operation of near-surface disposal facilities for radioactive wastes.[17] Use of the ALARA principle also is required in the DOE's radiation protection standards for the general public.[7]

The ALARA principle may be applied in two ways: (1) on a site-specific basis for a given practice and (2) on a practice-specific basis irrespective of site. The doses to individuals that would result from these two applications of the ALARA principle are indicated by the two dashed lines in Fig. 1.

Application of the ALARA principle on a site-specific basis for a given practice is illustrated by the NRC's requirement that all near-surface disposal facilities for radioactive wastes shall reduce annual dose equivalents to whole body for off-site members of the general public as far below the limit of 25 mrem (0.25 mSv) as is reasonably achievable.[17] The determination of what is ALARA is to be performed for each facility, so application of the ALARA principle to different sites generally will result in different maximum doses to off-site individuals.

General application of the ALARA principle to specific practices may lead to the determination of quantities of radionuclides that are "exempt" or "below regulatory concern" for that practice, irrespective of site. With regard to low-level waste disposal, for example, exempt concentrations of radionuclides would define materials that could be disposed of as if they were nonradioactive. The determination of exempt quantities of radionuclides for a specific practice essentially represents a judgment by the regulatory authorities that the doses associated with those levels of radioactivity are ALARA for that practice at any site, but the dose associated with exempt quantities of radionuclides may vary from one practice to another. As discussed in Section 3.2.1, several NRC rulemakings specify quantities of radionuclides that are generally exempt from certain licensing requirements.

2.5 *De Minimis* Dose

The concept of a generally applicable *de minimis* dose arises from the consideration that there must be a limit beyond which no further reduction in dose should be attempted using the ALARA principle, either at specific sites for a given practice or for particular practices irrespective of site. The *de minimis* dose is represented by the lowest line in Fig. 1.

As described in Section 3.1, a *de minimis* dose corresponds to a level of risk that most individuals in the general public would regard as "negligible." Thus, such a dose must be set well below established limits on acceptable dose from all sources of exposure and, furthermore, must be set below any established dose limit for specific practices. The *de minimis* dose not only is of no concern to regulatory authorities, as are doses associated with exempt quantities of radionuclides for specific practices, but such a dose also defines a level below which control of

radiation exposures would be deliberately and specifically curtailed;[1] i.e., it is a negligible dose level which is applicable to all practices and all sites. Proposals by regulatory authorities for a generally applicable *de minimis* dose are discussed in Section 3.2.

2.6 Summary of Concepts

The fundamental concepts in radiation protection described above and their interrelationships may be summarized as follows. A dose limit from all sources of exposure, excluding natural background and deliberate medical practices, corresponds to a limit on risk that is generally "acceptable" to members of the public and constitutes a ceiling for application of the ALARA principle; i.e., this dose limit must be met regardless of cost, except under unusual circumstances. Regulatory authorities then may establish lower dose limits for specific practices (e.g., low-level waste disposal), based on a judgment that such doses generally are "reasonably achievable," and this dose limit may vary from one practice to another. Application of the ALARA principle leads to reductions in doses to individuals below the generally applicable dose limits on both a site-specific and a practice-specific basis. The latter application also may lead to the determination of quantities of radionuclides that are "exempt" or "below regulatory concern," based on a judgment by regulatory authorities that doses associated with the exempt levels are ALARA. The generally applicable *de minimis* dose must be well below the limit on acceptable dose from all sources and below any dose limit for specific practices, and constitutes a floor for application of the ALARA principle; i.e., efforts to reduce doses below the *de minimis* level would, in all cases, be deliberately and specifically curtailed.

Thus, the limit on acceptable dose from all sources and the *de minimis* dose in Fig. 1 are generally applicable limits that define boundaries within which the ALARA principle operates. It must be emphasized, however, that while the *de minimis* dose constitutes a lower limit to ALARA, this dose is not the goal of ALARA since the dose that is "as low as reasonably achievable" for a specific practice at a particular site may be above the *de minimis* level.[1] The ALARA principle requires only that reasonable efforts be made to reduce and maintain doses as far below applicable limits as is practicable, taking into account a variety of technological, social, and economic factors, not that doses must be reduced to *de minimis* levels.

From the different concepts depicted in Fig. 1, it is evident that two approaches may be used to determine exempt concentrations of solid waste materials for purposes of disposal. The first approach involves

general application of the ALARA principle to waste disposal, and this optimization procedure based on a cost-benefit analysis could result in doses from disposal of exempt concentrations of radionuclides that are greater than the generally applicable *de minimis* dose. Alternatively, exempt concentrations of radionuclides for solid-waste disposal, or for any other practice, can be determined on the basis of the generally applicable *de minimis* dose itself without the need for a cost-benefit analysis based on application of the ALARA principle, provided a widely accepted value for a *de minimis* dose can be established.

3. RECOMMENDATIONS OF RADIATION PROTECTION AUTHORITIES FOR A *DE MINIMIS* DOSE

This section presents a review of recommendations by radiation protection authorities for a generally applicable *de minimis* dose for members of the general public, i.e., a dose that would be below regulatory concern for any practice at any site. This review also includes recommendations related to establishing exempt concentrations of radionuclides for purposes of waste disposal. Section 3.1 presents the general approach, based on the concept of negligible risk, that normally is used in developing a *de minimis* dose. Section 3.2 then reviews the various recommendations for a *de minimis* dose by radiation protection authorities. Finally, Section 3.3 presents a proposal for a generally applicable *de minimis* dose for members of the general public that we have developed on the basis of this review. We reiterate that establishment of a *de minimis* dose would provide the most defensible basis for deriving exempt concentrations of radionuclides in solid wastes.

3.1 General Approach to Defining a *De Minimis* Dose

The approach normally used in developing a generally applicable *de minimis* dose is based on the concept of negligible risk,[1] as outlined below.

[1] On the basis of risks that are widely accepted by the general public, a "negligible" lifetime risk from radiation exposure is defined. The negligible risk must be well below the limit for acceptable risk on which radiation protection standards are based.

[2] From the negligible lifetime risk from radiation exposure so defined, a lifetime *de minimis* dose is derived using an accepted factor for the risk of fatal cancers and genetic defects per unit dose. In radiation protection, the risk factor is based on a linear no-threshold, dose-response hypothesis, and the value usually is assumed to be in the range 1-2 x 10^{-2} per Sv (1-2 x 10^{-4} per rem).[4]

[3] From the lifetime *de minimis* dose so derived, an annual *de minimis* dose is obtained by assuming exposure over a 70-year lifetime for an average individual; i.e., the lifetime *de minimis* dose is divided into equal annualized increments.

The result of this approach is that a negligible lifetime risk is

expressed in terms of a *de minimis* dose for each year of exposure. Alternatively, one can define an annual negligible risk that forms the basis for an annual *de minimis* dose, instead of first deriving a lifetime *de minimis* dose, but all approaches are based on an assumption for a negligible level of risk.

The approach to defining a generally applicable *de minimis* dose does not involve consideration of particular practices or facility locations. However, use of a *de minimis* dose to derive exempt quantities or concentrations of radionuclides involves consideration of specific practices (e.g., waste disposal) and, perhaps, specific locations (e.g., differences in doses per unit concentration from disposal in humid or arid environments).

3.2 Current Recommendations for a *De Minimis* Dose

This section reviews current recommendations by regulatory authorities in the U.S. and other countries for a *de minimis* dose and for defining exempt concentrations of radionuclides for purposes of waste disposal.

3.2.1 *U.S. Nuclear Regulatory Commission*

The NRC is considering a generally applicable *de minimis* dose in revising its radiation protection standards for the general public.[6] Although a proposal for a *de minimis* dose is not given, the supplementary information in the proposed rule presents a calculation of an annual dose equivalent of 0.1 mrem (0.001 mSv), based on an assumed negligible lifetime risk of 10^{-6}. The proposed rule also contains an annual dose equivalent of 1 mrem (0.01 mSv) as a cutoff level for use in population dose evaluations;[6] i.e., individual doses below the cutoff level would be excluded in applying the ALARA principle to control of exposures of the general public. However, the proposed cutoff level for population dose evaluations is not a *de minimis* dose for maximally exposed individuals.

In developing its standards for near-surface disposal of radioactive wastes,[17] the NRC considered the issue of establishing generally applicable exemption levels for radioactivity in waste materials.[22] However, instead of developing general criteria for defining exempt wastes, the NRC chose to maintain its policy of considering waste streams on a case-by-case basis to determine if they are below regulatory concern. The NRC believed that consideration of a variety of waste streams would facilitate the desirable goal of establishing a generally applicable *de*

minimis dose. In this regard, the NRC's dose-assessment methodology for near-surface disposal of radioactive wastes was used to estimate potential doses from disposal of different low-activity waste streams.[23] In each case, the estimated annual dose equivalent to whole body for an inadvertent intruder into the disposal facility was 0.4 mrem (0.004 mSv) or less.

The NRC recently has issued a policy statement regarding handling of petitions for exempting specific radioactive waste streams from disposal in a licensed low-level waste disposal facility.[24] This guidance provides several decision criteria that will be used by the NRC in judging whether to grant a petition. Some of these criteria are as follows: (1) the maximum expected effective dose equivalent to an individual member of the public does not exceed a few mrem per year for normal operations and anticipated events; (2) the collective doses to the critical population and the general population are small; and (3) the potential radiological consequences of accidents or equipment malfunction involving the wastes and intrusion into disposal sites after loss of normal institutional controls are not significant. The NRC also has indicated that decision criteria for exempting radioactive wastes from disposal in licensed facilities will be implemented in a generic rulemaking,[25] but the policy guidance discussed above will be used on an interim basis until final regulations have been adopted.

Several current NRC rulemakings specify quantities of radionuclides that are exempt from certain licensing requirements: in 10 CFR Part 20, scintillation materials and animal carcasses containing concentrations of ^{3}H and ^{14}C less than 0.05 μCi/g* may be disposed of without regard to their radioactivity;[5] 10 CFR Part 30 lists exempt quantities and concentrations of many radionuclides that are byproduct materials and exempt quantities of radionuclides in such items as resins, self-luminous products, and gas and aerosol detectors;[26] 10 CFR Part 40 describes "unimportant" quantities of source materials;[27] and 10 CFR Part 71 states that packages containing radioactive materials having a specific activity not greater than 0.002 μCi/g are exempt from licensing requirements for packaging and transportation.[28] However, the exempt quantities of radioactivity in the different NRC rulemakings do not appear to be associated with the same dose to members of the general public; i.e., considerations other than dose were important in establishing exempt levels of radioactivity in each case. Thus, these exempt quantities of

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* Throughout this report, quantities of radioactivity are given in units of μCi, rather than in the SI unit of Bq, primarily because quantities of radioactivity generally have been given in the non-SI units in the regulations and other documents cited herein. The conversion between the two units is 1 μCi = 37 kBq.

1 μCi = .037 Bq
30 μCi/g = 1.11 Bq/g

radioactivity probably do not provide an appropriate basis for developing a generally applicable *de minimis* dose.

3.2.2 *U.S. Environmental Protection Agency*

In developing standards for disposal of low-level radioactive wastes,[29] the EPA is considering waste streams with activity levels that could be below regulatory concern. The EPA has performed dose calculations for many low-activity waste streams by assuming a variety of exposure scenarios and geographical locations for a disposal facility. In about half of the calculations, the estimated annual dose equivalent to whole body was less than 1 mrem (0.01 mSv), but annual dose equivalents as large as 70 mrem (0.7 mSv) were obtained. The EPA has not yet indicated a dose that might be considered below regulatory concern for waste disposal or any other purpose.

3.2.3 *U.S. Department of Energy*

In response to the requirement in DOE Order 5820.2 that exempt quantities of low-level radioactive wastes be established for disposal at DOE sites,[2] two DOE-contractor organizations have developed proposals for defining exempt concentrations of radionuclides that could be placed in an on-site sanitary landfill. A draft document prepared for the National Low-Level Waste Management Program[30] contains a recommendation for a threshold limit on annual committed effective dose equivalent to an inadvertent intruder at a disposal site in the range 1-10 mrem (0.01-0.1 mSv). A threshold limit is distinguished from a dose that is *de minimis* or below regulatory concern by the administrative controls that would be placed on wastes with exempt concentrations of radionuclides; i.e., wastes below the threshold limit are those generated by the DOE for disposal in a DOE-operated sanitary landfill on a DOE-controlled site. Thus, a threshold limit dose might reasonably be higher than a generally applicable *de minimis* dose or a dose from waste disposal that would be below regulatory concern at any site. The second DOE-contractor report assumes as a *de minimis* level an annual committed effective dose equivalent of 1 mrem (0.01 mSv).[31]

The derivation of exempt concentrations of radionuclides in the two DOE-contractor reports[30,31] on the basis of the assumed dose limits for an inadvertent intruder is discussed further in Section 4.2.

3.2.4 *National Council on Radiation Protection and Measurements*

The NCRP is considering as a *de minimis* level an annual committed effective dose equivalent of 1 mrem (0.01 mSv), based on an assumed negligible risk of 10^{-7} per year.[9,32] The proposed *de minimis* dose is 1% of the annual limit on acceptable dose from all sources for continuous exposures that is being considered by the NCRP (see Section 2.2). The NCRP also is considering a proposal that population dose assessments should exclude those individuals who receive annual committed effective dose equivalents less than 1 mrem (0.01 mSv). Thus, the proposed *de minimis* dose for maximally exposed individuals is the same as the value for truncating population dose calculations.

3.2.5 *International Commission on Radiological Protection*

The ICRP has issued a set of recommendations for exempting sources of radiation exposure from licensing, registration, or notification requirements.[33] Although these recommendations focus on exempting solid wastes from requirements for disposal as radioactive materials, the recommendations also would apply to any practice.

On the basis of an assumed negligible risk of 10^{-6} per year, the ICRP recommends that an annual committed effective dose equivalent of 0.1 mSv (10 mrem) may be regarded as negligible. However, to take into account the possibility that an individual could be exposed to several exempt sources, the ICRP further recommends that the exemption criterion be reduced to an annual committed effective dose equivalent per source of 0.01 mSv (1 mrem). Since it seems almost certain that the total annual dose equivalent to a single individual from all exempt sources will be less than ten times the contribution from the exempt source giving the highest dose, the dose limit per source will ensure that the recommended negligible dose level will not be exceeded when all exempt sources are taken into account. The ICRP also recommends that an exempt source should result in a collective committed effective dose equivalent over a defined period of operation that does not exceed 1 person-Sv (100 person-rem).

Thus, in summary, the ICRP recommends that sources could be exempted if the collective committed effective dose equivalent is less than 1 person-Sv (100 person-rem) and if this collective dose equivalent is made up of annual individual dose equivalents less than 0.01 mSv (1 mrem).

3.2.6 *Atomic Energy Control Board (Canada)*

The Atomic Energy Control Board of Canada has issued a proposal for exempting the disposal of certain radioactive materials from licensing requirements.[34] The proposal is that exempt concentrations of radionuclides are to be determined from a *de minimis* level of an annual committed effective dose equivalent of 0.05 mSv (5 mrem), which is based on an assumed negligible risk of 10^{-6} per year. A further recommendation is that the potential for exposure of large populations to the *de minimis* dose will be small; i.e., the radiological impacts from disposal of exempt materials will be localized.

3.2.7 *National Radiological Protection Board (U.K.)*

The U.K.'s National Radiological Protection Board (NRPB) has recommended as a *de minimis* level an annual committed effective dose equivalent of 0.05 mSv (5 mrem),[35] which is 1% of the limit on annual dose for members of the general public recommended in ICRP Publication 26.[4] The NRPB also recommends, however, that the *de minimis* dose from any practice be reduced by a factor of 10, i.e., to an annual committed effective dose equivalent of 0.005 mSv (0.5 mrem), when an individual could be exposed to several exempt sources.

3.2.8 *International Atomic Energy Agency*

The International Atomic Energy Agency (IAEA) has convened an advisory group to recommend methods for use in determining exempt levels of radioactive waste for disposal in the terrestrial environment.[36] The advisory group has recommended that the determination of exempt concentrations of radionuclides be derived from a *de minimis* level of an annual committed effective dose equivalent of 0.01 mSv (1 mrem), based on an assumed negligible risk of 10^{-7} per year. The advisory group also points out that the proposed *de minimis* dose essentially applies only to man-made radionuclides, because it is illogical to apply a limit on annual dose equivalent of 0.01 mSv (1 mrem) to naturally occurring radionuclides which, in their undisturbed state, lead to doses much greater than the *de minimis* value.

The IAEA also has convened an advisory group to consider general principles for exempting radioactive materials from basic safety standards.[37] This advisory group also has recommended as a *de minimis* level an annual committed effective dose equivalent of 0.01 mSv (1 mrem)

and noted the problem of establishing exempt concentrations for naturally occurring radionuclides discussed above. A second recommendation is that a practice may be left unregulated if the annual collective dose-equivalent commitment is less than 1 person-Sv (100 person-rem).

3.2.9 *Summary of recommendations*

The recommendations of various radiation protection authorities that could be used as a generally applicable *de minimis* dose for members of the general public are summarized in Table 1. The proposals for an annual dose equivalent are in the range 0.001-0.1 mSv (0.1-10 mrem), with most recommendations focusing on 0.01 or 0.05 mSv (1 or 5 mrem). We emphasize that the three U.S. Government agencies have not yet endorsed either a value for a generally applicable *de minimis* dose or a dose that could be used for establishing exempt concentrations of radionuclides in waste materials for purposes of disposal. The NRC has indicated, however, that waste streams giving annual dose equivalents of a few mrem to individual members of the public could be exempted from licensing requirements for disposal of radioactive wastes.

3.3 Proposal for a *De Minimis* Dose

This section presents some comments on the recommendations for a *de minimis* dose discussed in the previous section, and presents a proposal based on this review.

3.3.1 *Comments on recommendations for a de minimis dose*

The following comments are offered on the recommdations by various regulatory authorities for a *de minimis* dose for members of the general public.

[1] A *de minimis* dose need not be expressed as a limit for each year of exposure, as is customary in all of the proposals reviewed, primarily because the negligible risks on which the *de minimis* dose are based generally are not constant over a lifetime. The alternative of expressing the *de minimis* level as an annual dose averaged over a lifetime is more closely related to the fundamental goal of limiting lifetime risk while also encouraging proper consideration of the age dependence of dose and risk in deriving

Table 1. Recommendations of regulatory authorities for a *de minimis* dose

Authority	Annual dose (mSv)[a]
U.S. Department of Energy[b]	0.01-0.1
Atomic Energy Control Board (Canada)	0.05
National Radiological Protection Board (U.K.)	0.05[c]
U.S. Department of Energy[d]	0.01
National Council on Radiation Protection and Measurements	0.01
International Commission on Radiological Protection	0.01[e]
International Atomic Energy Agency	0.01[f]
U.S. Nuclear Regulatory Commission[g]	0.001 (?)
U.S. Environmental Protection Agency	?

[a]Values are annual committed effective dose equivalents to individuals, and may be converted to units of mrem by multiplying by a factor of 100.

[b]Proposal for waste disposal only at DOE sites for DOE's National Low-Level Radioactive Waste Management Program (ref. 30).

[c]Recommended value is 0.005 mSv when an individual could be exposed to several exempt sources.

[d]Proposal for waste disposal only at DOE sites from Paducah Gaseous Diffusion Plant (ref. 30).

[e]Recommended value of 0.1 mSv for exposure to all exempt sources is reduced by a factor of 10 to take into account the possibility of exposure to several exempt sources. In addition, the collective dose equivalent from an exempt source should be less than 1 person-Sv.

[f]Value is intended for application only to man-made radionuclides. Practice also may be left unregulated if annual collective dose equivalent is less than 1 person-Sv.

[g]Value obtained from example calculation in Supplementary Information for proposed revision of 10 CFR Part 20 (ref. 6). In addition, guidance has been established that waste streams may be exempted from licensing requirements for disposal of radioactive wastes if the annual dose equivalent does not exceed a few mrem (ref. 24).

exempt concentrations of radionuclides in solid wastes.[38] Furthermore, one can specify both a limit on annual dose averaged over a lifetime and a higher limit on dose in any year.

[2] An annual dose equivalent of about 0.2 mSv (20 mrem), which corresponds approximately to the standard deviation of natural background radiation in the U.S.,[39,40] might seem a reasonable choice for a *de minimis* level. However, this dose probably is too high to be acceptable as *de minimis* in the U.S. because of the widespread use of an annual dose equivalent of 0.25 mSv (25 mrem) as a limit for specific practices, including waste disposal.[13-17] A *de minimis* dose, by definition, must be below any dose that is of concern to regulatory authorities; and, indeed, the interim standards for radioactivity in drinking water[20] may preclude an annual *de minimis* dose equivalent as low as 0.05 mSv (5 mrem), unless the EPA chooses to replace the different dose and concentration limits for various radionuclides by a single dose limit, as is being considered in proposed revisions of the standards.[41]

[3] An annual dose equivalent much below 0.01 mSv (1 mrem) may be too low to be reasonable as a *de minimis* level, because of the difficulties that likely would be encountered in measuring associated quantities of radioactivity. Furthermore, an annual *de minimis* dose equivalent approaching 0.001 mSv (0.1 mrem) would be less than 0.1% of the dose from natural background radiation. Such a low *de minimis* dose for each source perhaps could be justified only if exposures of individuals to many exempt sources were likely.

3.3.2 *A proposed de minimis dose*

From the considerations outlined above, we offer the following proposal for a generally applicable *de minimis* dose for members of the general public:

[1] a principal limit on annual committed effective dose equivalent averaged over a lifetime of 0.01 mSv (1 mrem); and

[2] a subsidiary limit on committed effective dose equivalent in any year of 0.05 mSv (5 mrem).

This proposal is based primarily on (1) the recommendation being

considered by the NCRP that the *de minimis* dose be set at 1% of the limit on acceptable dose from all sources of exposure[9,32] and (2) the two-tiered dose-limitation system for continuous and occasional exposures currently recommended by the ICRP and being considered by the NCRP (see Section 2.2). An annual dose equivalent of 0.01 mSv (1 mrem) also has been proposed as a *de minimis* level by Kathren *et al.* on the basis of a cost-benefit analysis.[42]

The proposed *de minimis* dose would apply to man-made but not to naturally occurring radionuclides, as recommended by advisory groups of the IAEA.[36,37] Levels of naturally occurring radionuclides that are generally exempt or below regulatory concern would be determined instead from application of the ALARA principle to specific practices. The annual dose equivalents to members of the public resulting from exempt levels of naturally occurring radionuclides presumably will be well above the *de minimis* level for man-made radionuclides, because naturally occurring radionuclides in their undisturbed state lead to doses much greater than the proposed *de minimis* value.

The proposed *de minimis* dose is about 1% of the dose from natural background radiation and, assuming a risk factor of about 2×10^{-4} per rem (2×10^{-2} per Sv),[4] corresponds to a lifetime risk from continuous exposure of about 10^{-5}. This level of risk generally is regarded as negligible by most individuals in the general public. This is a dummy page.

4. REVIEW OF METHODOLOGIES FOR DERIVING EXEMPT CONCENTRATIONS OF RADIONUCLIDES IN SOLID WASTES

4.1 Introduction

This section presents a review of selected methodologies developed by other investigators for deriving exempt concentrations of radionuclides in solid waste materials. The outputs of these methodologies can be expressed as factors that convert radionuclide concentrations in a disposal facility to annual dose equivalents to inadvertent intruders at the disposal site. The establishment of an annual *de minimis* dose thus would provide a basis for deriving exempt concentrations of radionuclides in solid wastes. The methodologies for estimating annual doses per unit concentration of radionuclides in solid wastes usually focus on exposures of inadvertent intruders, because doses to an intruder generally are higher than those to off-site individuals due to the effects of dilution in the transport of radionuclides from the disposal facility to locations at which off-site exposures occur.

Annual doses to an inadvertent intruder per unit concentration of radionuclides in the disposal facility may be estimated for a number of exposure pathways. The general equation for estimating the annual dose at time t from radionuclide i and exposure pathway p is[43]

$$H_{ip}(t) = C_{ip}(t)U_pD_{ip} \ , \qquad (1)$$

where

H = annual dose equivalent,
C = radionuclide concentration in medium of exposure (air, water, soil, or foodstuffs),
U = usage parameter (annual exposure time or annual intake of contaminated material), and
D = dose conversion factor (annual dose equivalent per unit radionuclide concentration in environment for external exposure or committed dose equivalent per unit radionuclide intake for internal exposure).

We prefer to express H as the annual committed effective dose equivalent,[4] which includes the effective dose equivalent from external exposure, but this practice has not been followed by all other investigators.

The radionuclide concentrations C in eq. (1) are obtained from consideration of environmental transport of radionuclides following disposal, either by such natural processes as leaching by water and infiltration through soil, root uptake by vegetation, or suspension into the atmosphere, or by direct actions of the intruder, e.g., excavation of

the waste materials in a trench. Most models treat the radionuclide concentrations in the different exposure media as constant fractions of the concentrations in the trench; these are the so-called equilibrium models. For example, the ratio of the concentration in vegetation to the concentration in contaminated soil is assumed to be a constant for all isotopes of a given element. However, some models treat the radionuclide concentrations in the exposure media as dynamic variables. In these models, the transfers of activity between environmental compartments are described by fractional transfer rates that are constant with time, and the resulting concentrations in each compartment as a function of time are obtained as solutions of sets of simultaneous first-order linear differential equations. The asymptotic solutions of these equations give the same concentrations as would be obtained from an equilibrium model. The use of the simpler equilibrium models generally is adequate for purposes of deriving exempt concentrations of radionuclides in solid wastes.

The usage parameter U and the dose conversion factor D in eq. (1) are treated as constants by most investigators, since the methodologies generally focus on exposures of adults. Age-dependent usage parameters and dose conversion factors may be needed if exposures of infants and children also are considered in the dose analysis. The dose conversion factors for internal exposures via ingestion or inhalation are the same for all exposure pathways that lead to intakes by the given route. For external exposures, however, the dose conversion factors depend on the assumed distribution of sources in the environment and the location of the exposed individual relative to the source region.

Some methodologies for estimating dose to an inadvertent intruder explicitly include consideration of such factors as the leachability or solubility of the wastes that affect the concentration of radionuclides in the different exposure media relative to the concentration in the disposal facility itself. Other methodologies may calculate dose with the assumption that all activity in the trench is available for transport to the exposure media, so that additional correction factors may be needed in cases where this assumption is not appropriate (e.g., for mobilization of activated metals). An advantage of the second method is that the dose calculations are easily modified if the assumptions for the correction factors are changed.

Finally, various methodologies may differ in the way they treat an assumed time delay between disposal of the wastes and the onset of intruder exposures, e.g., following loss of institutional controls over the facility. Most methodologies calculate annual doses per unit concentration of radionuclides initially in the trench and include in the calculations loss by radioactive decay or other physical removal processes

during an assumed period of institutional controls (e.g., 100 years). Other methodologies calculate annual doses per unit concentration of radionuclides at the time exposure occurs, so additional correction factors are needed to calculate the corresponding concentrations at the time of disposal. The second method has the advantage that it allows for flexibility in describing reductions in concentrations between the time of disposal and the time exposures are assumed to occur.

The remainder of this section presents a review of the selected methodologies that can be used to estimate annual doses to an inadvertent intruder per unit concentration of radionuclides in a disposal facility, and the results from some methodologies are tabulated for selected radionuclides. We particularly emphasize results for the uranium isotopes that occur in solid wastes generated at the Y-12 Plant in Oak Ridge, the most important of which are ^{238}U and ^{234}U. Section 4.2 considers methodologies that have been developed by the DOE and its contractors; Section 4.3 reviews methodologies that have been developed by the NRC and its contractors; and Section 4.4 discusses a methodology that has been developed under the auspices of the IAEA. Finally, Section 4.5 discusses a proposal that has been used by the DOE for determining exempt concentrations of uranium in solid wastes at the Y-12 Plant.

4.2 Methodologies Developed by the DOE

This section reviews a number of methodologies developed by the DOE and its contractors that can be used to estimate annual doses to an inadvertent intruder per unit concentration of radionuclides in solid wastes. We focus primarily on the methodologies developed by EG&G Idaho[30] and by Jokerst,[31] since they were developed specifically to derive exempt concentrations of radionuclides for disposal in a sanitary landfill, but other calculations also are discussed.

4.2.1 *Methodology developed by EG&G Idaho*

The methodology developed by EG&G Idaho for DOE's National Low-Level Waste Management Program[30] considers doses resulting from the following scenarios for exposure at a disposal site:

- exposures of workers during the operational period of the facility;

- exposures of an individual who constructs a home on the site following loss of institutional controls (i.e., the intruder-construction scenario); and

- exposures of an individual who inhabits a homestead on the site following loss of institutional controls (i.e., the intruder-agriculture scenario).

The exposure pathways for the site-worker and intruder-construction scenarios include the following:

- external exposure to radionuclides in soil and suspended in air;

- inhalation of suspended radionuclides;

- ingestion of contaminated soil.

The exposure pathways for the intruder-agriculture scenario include the following:

- external exposure to radionuclides in soil and suspended in air;

- inhalation of suspended radionuclides;

- ingestion of foodstuffs contaminated by atmospheric deposition and root uptake from soil;

- ingestion of animal products (meat and milk) obtained from livestock grazing on contaminated vegetation;

- ingestion of contaminated water;

- ingestion of contaminated soil.

Environmental transport of radionuclides for each of the exposure scenarios and associated pathways is calculated using the DOSTOMAN computer code developed at the Savannah River Laboratory.[44] This code contains a dynamic model that performs calculations of time-dependent inventories of radionuclides in the different environmental compartments of interest. The output of this methodology is given in terms of annual doses per unit concentration of radionuclides in the facility at the time of disposal. Included in the model are factors that account for decay of radionuclides and removal rates from soil by other processes during the

period of institutional controls, which is assumed to be 100 years.

Calculations of annual committed effective dose equivalents per unit concentration of radionuclides in the disposal facility at the time of disposal were performed for an arid site (i.e., Idaho National Engineering Laboratory) and a humid site (i.e., Savannah River Plant). The results of the calculations for selected radionuclides for disposal at the Savannah River Plant are given in Table 2; this site most resembles the Oak Ridge Reservation. For all radionuclides, the limiting exposure scenario (i.e., the scenario giving the highest dose per unit concentration of radionuclides) was either the on-site worker or the intruder-agriculture scenario. Table 2 includes results only for those radionuclides for which the intruder-agriculture scenario was limiting, since occupational exposures are not of concern in this report.

The results in Table 2 include possible contributions from ingestion of contaminated drinking water from a well drilled into an aquifer below the disposal trench, as well as the contributions from all the other pathways that involve activity in the trench itself. The radionuclides that could yield doses from the drinking water pathway presumably include those with low retardation coefficients in transport through the soil column to the aquifer (e.g., ^{3}H, ^{14}C, and ^{99}Tc and those with moderate retardation coefficients but long half-lives (e.g., the long-lived uranium isotopes). However, the importance of the drinking water pathway to the results in Table 2 is not indicated in ref. 30. Such information is potentially important when comparing these results with those obtained by other investigators, because the dose from the drinking water pathway generally will be much more site-specific than the dose from the other pathways.

The calculations for the Savannah River site included ^{238}U but not ^{234}U. The annual dose per unit concentration for ^{238}U is based on the ingestion pathways for the intruder-agriculture scenario. Again, however, the importance of the drinking water pathway to the dose from ^{238}U is not indicated in ref. 30. From the methodology for the Oak Ridge Reservation discussed in Section 5 of this report, we expect that the annual dose from ^{238}U + ^{234}U in secular equilibrium per unit concentration of ^{238}U will be about twice the annual dose from ^{238}U alone.

4.2.2 *Methodology of Jokerst*

Jokerst[31] has presented calculations of annual doses per unit concentration of radionuclides in a disposal facility assuming the following exposure scenarios:

Table 2. Annual doses to an inadvertent intruder per unit concentration of radionuclides in disposal facility obtained from EG&G Idaho methodology[a]

Nuclide	Annual dose[b] (mrem/y per $\mu Ci/m^3$)
H-3	7.8E-7
C-14	3.8
Ni-63	1.7E-4
Sr-90	2.8E-2
Tc-99	1.0E-2
Cs-137	1.8
U-235	1.2
U-238	5.7E-1
Pu-239	5.6E-1
Pu-240	5.7E-1
Pu-242	5.9E-1
Am-241	7.2E-1
Am-243	2.4

[a]From ref. 30; values are for site at Savannah River Plant.

[b]Results are given as annual committed effective dose equivalents per unit concentration at time of disposal, and 100 years of institutional controls is assumed before exposures occur.

- water intrusion and groundwater transport to an access location at the site boundary;

- overflow of the disposal facility and transport to a nearby surface stream;

- an intruder-agriculture scenario.

The intruder-agriculture scenario was found to limit the radionuclide concentrations in most cases. The exposure pathways for this scenario include the following:

- external exposure to radionuclides in soil and suspended in air;

- inhalation of suspended radionuclides;

- ingestion of foodstuffs contaminated by atmospheric deposition and root uptake from soil.

These pathways involve only exposures to activity in the disposal facility itself; i.e., the drinking water pathway is not included in the intruder-agriculture scenario.

The dose calculations for the intruder-agriculture scenario were based on the methodology developed by the NRC[23] in support of its rulemaking on shallow-land disposal of radioactive wastes.[17] The calculations give annual committed effective dose equivalents per unit concentration of radionuclides at the time of disposal, and they include consideration of loss of activity by decay and other physical removal processes during the period of institutional controls, which is assumed to be 20 years.

The results of the calculations for selected radionuclides are given in Table 3. Both ^{238}U and ^{234}U are included in the calculations, and the most important exposure pathways for $^{238}U + ^{234}U$ are those involving airborne activity and external exposure from the soil volume. The annual doses per unit concentration for ^{238}U and ^{234}U in Table 3 are about four orders of magnitude less than the value for ^{238}U in Table 2. The source of such a large discrepancy is not known. We attempted to reproduce the result for ^{238}U in Table 3 using the sources of data given by Jokerst[31] but were not successful. The value we obtained was only about a factor of 25 less than the value from ref. 30 given in Table 2.

Table 3. Annual doses to an inadvertent intruder per unit concentration of radionuclides in disposal facility obtained from methodology of Jokerst[a]

Nuclide	Annual dose[b] (mrem/y per $\mu Ci/m^3$)
H-3	2.6E-5
C-14	1.0E-5
Sr-90	4.2E-6
Tc-99	1.9E-5
Cs-137	9.1E-4
Ra-226	4.2E-3
Th-232	4.8E-3
U-234	4.5E-5
U-235	3.6E-4
U-238	6.7E-5
Pu-238	1.1E-4
Pu-239	1.4E-4

[a]From ref. 31.

[b]Results are given as annual committed effective dose equivalents per unit concentration at time of disposal, and 20 years of institutional controls is assumed before exposures occur.

4.2.3 *Other DOE methodologies*

This section discusses results from other methodologies presented in DOE and DOE-contractor reports that could be used to calculate dose to an inadvertent intruder at a waste disposal facility. We again emphasize results for ^{238}U and ^{234}U.

An analysis of radionuclide migration pathways for the proposed Central Waste Disposal Facility in Oak Ridge[3] considered doses to inadvertent intruders assuming an intruder-agriculture scenario.[45] The following exposure pathways were included in the analysis:

- external exposure to radionuclides in soil;
- inhalation of suspended radionuclides;
- ingestion of vegetables contaminated by root uptake from soil;
- ingestion of contaminated water.

Doses were calculated for whole body and for bone, kidneys, and lungs.

The annual doses to whole body per unit concentration for selected radionuclides are given in Table 4; these results apply to trench disposal only but not to disposal in above-ground tumuli. The dose from the drinking water pathway is an important contributor to the total dose for ^{3}H, ^{99}Tc, and all the uranium isotopes. The dose from the drinking water pathway for all uranium isotopes was determined by the assumed solubility limit for uranium in water of 45 μmol/L.[45] Thus, the dose from each uranium isotope for the drinking water pathway is independent of the total concentration of uranium in the disposal facility, provided the concentration exceeds the solubility limit. The effect of a solubility limit for uranium on the dose from the different uranium isotopes is discussed further in Section 5.3.

The DOE has published a pathways analysis for estimating doses to individuals who would intrude onto contaminated sites that were used for storage and processing of uranium and thorium ores.[46] The only radionuclides of concern to the analysis were ^{238}U, ^{234}U, and their daughter products, and ^{238}U and ^{234}U were assumed to be in secular equilibrium. An intruder-agriculture scenario was assumed with the following exposure pathways:

- external exposure to radionuclides in soil and suspended in air;

Table 4. Annual doses to an inadvertent intruder per unit concentration of radionuclides in disposal trench obtained from methodology of Pin *et al.*[a]

Nuclide	Annual dose[b] (mrem/y per $\mu Ci/m^3$)
H-3	7.4E-5
C-14	4.0E-1
Sr-90	5.4E-3
Tc-99	4.1E-2
Cs-137	1.6E-2
U-234	4.6E-1[c]
U-235	5.6E-2[d]
U-238	7.5E-2[e]

[a]From Tables 3.3 and 6.7-6.11 of ref. 45. Results do not apply to disposal in above-ground tumuli.

[b]Results are given as annual committed dose equivalents to whole body per unit concentration at time of disposal, and 100 years of institutional controls is assumed before exposures occur.

[c]Contribution from vegetable pathway only; annual dose equivalent from drinking water pathway is 34 mrem independent of concentration in disposal facility, provided total concentration of uranium exceeds assumed solubility limit of 45 μmol/L.

[d]Contribution from vegetable, external exposure, and inhalation pathways only; annual dose equivalent from drinking water pathway is 75 mrem independent of concentration in disposal facility, provided total concentration of uranium exceeds assumed solubility limit of 45 μmol/L.

[e]Contribution from vegetable and inhalation pathways only; annual dose equivalent from drinking water pathway is 73 mrem independent of concentration in disposal facility, provided total concentration of uranium exceeds assumed solubility limit of 45 μmol/L.

- inhalation of suspended radionuclides;
- ingestion of foodstuffs contaminated by atmospheric deposition and root uptake from soil;
- ingestion of meat and milk derived from livestock grazing on contaminated vegetation and drinking contaminated water;
- ingestion of contaminated water;
- ingestion of contaminated fish;
- ingestion of contaminated soil.

Annual doses per unit concentration for the different pathways were calculated for whole body and for bone, liver, kidneys, and lungs. The dose from ^{238}U + ^{234}U in secular equilibrium is determined primarily by ingestion of contaminated water and external exposure to contaminated soil. The estimated annual dose to whole body from ^{238}U + ^{234}U, as obtained from Table S.1 of ref. 46, is 2.5 mrem/y per pCi/g of ^{238}U in soil. If we assume a soil density of 1.4 g/cm^3 (ref. 47), then we obtain a result of 1.8 mrem/y per μCi/m^3, which is within a factor of 3 of the result in Table 2 for ^{238}U alone obtained from the EG&G Idaho methodology.[30] As discussed in Section 5 of this report, we expect that ^{238}U and ^{234}U each contribute about half of the dose when the two isotopes are in secular equilibrium, so the results from refs. 30 and 46 appear to be in good agreement.

4.3 Methodologies Developed by the NRC

The NRC developed a comprehensive dose-assessment methodology[22,23,48] in support of its rulemaking on near-surface disposal of radioactive wastes.[17] The dose analysis for an inadvertent intruder was used to establish the concentration limits for disposal of different radionuclides that are contained in the rulemaking (i.e., the limits for Class-A, -B, and -C wastes).[17]

The NRC's dose analysis for an inadvertent intruder considered intruder-construction and intruder-agriculture scenarios. The intruder-agriculture scenario generally was limiting in determining maximum radionuclide concentrations that are generally acceptable for near-surface disposal. The following exposure pathways were considered for this scenario:

- external exposure to radionuclides in soil and suspended in air;

- inhalation of suspended radionuclides;

- ingestion of foodstuffs contaminated by atmospheric deposition and root uptake from soil.

The calculations give annual doses to whole body and several body organs per unit concentration of radionuclides at the time of disposal. The calculations include consideration of loss of activity by decay during the period of institutional controls, which is assumed to be 100 years, and they include consideration of the dispersibility, leachability, stability, and accessibility of the wastes.

The results of the calculations for selected radionuclides are given in Table 5. These results apply to Class-A wastes, which have the least stringent requirements on stability of the waste form[17] and, thus, would most closely resemble the types of wastes that would be placed in a sanitary landfill. We note that the results for the uranium and transuranium isotopes were obtained from ref. 22, rather than the rulemaking itself.[17]

The Pacific Northwest Laboratory also has developed a methodology for the NRC that estimates doses to an inadvertent intruder.[49] An intruder-agriculture scenario is assumed with the following exposure pathways:

- external exposure to radionuclides in soil;

- inhalation of suspended radionuclides;

- ingestion of foodstuffs contaminated by atmospheric deposition and root uptake from soil;

- ingestion of contaminated water.

The user also may construct other exposure pathways by appropriate definition of conditions associated with each pathway. No calculational results were presented in the report describing this methodology.

4.4 Methodology Developed by the IAEA

A methodology for deriving exempt concentrations of radionuclides in solid wastes is being developed by an Advisory Group of the IAEA.[50] The exempt concentrations are based on an annual *de minimis* dose of 0.01 mSv

Table 5. Annual doses to an inadvertent intruder per unit concentration of radionuclides in disposal facility obtained from NRC methodology[a]

Nuclide	Annual dose[b] (mrem/y per $\mu Ci/m^3$)
H-3	1.3E-5
C-14	6.3E-4
Ni-63	1.4E-4
Sr-90	1.3E-2
Tc-99	1.7E-3
Cs-137	5.0E-4
U-235	1.3E-2
U-238	1.0E-2
Pu-238	1.8E-2
Pu-239	4.8E-2
Pu-240	4.8E-2
Pu-242	4.5E-2
Am-241	6.3E-2
Am-243	7.6E-2
Cm-244	9.3E-4

[a]From Tables 1 and 2 of ref. 17, except from Table 4.5 of Main Report of ref. 22 for uranium isotopes and from p. C-135 of Appendix C of ref. 22 for transuranium isotopes.

[b]Results are given as annual committed dose equivalents to whole body per unit concentration at time of disposal, and 100 years of institutional controls is assumed before exposures occur.

(1 mrem) committed effective dose equivalent, as recommended by the same Advisory Group.[36] The following exposure scenarios are considered:

- site equipment operator;
- intruder-construction;
- residential (i.e., intruder-agriculture);
- trench fire;
- groundwater transport;
- incineration.

For the intruder-agriculture scenario, the following exposure pathways are considered:

- external exposure to radionuclides in soil;
- inhalation of suspended radionuclides;
- ingestion of foodstuffs contaminated by atmospheric deposition and root uptake from soil.

The methodology developed by the Advisory Group is used to calculate exempt concentrations for a variety of man-made radionuclides. However, no calculations were performed for uranium isotopes because of the view of the Advisory Group that it is not reasonable to apply an annual *de minimis* dose equivalent of 0.01 mSv (1 mrem) to naturally occurring radionuclides (see Section 3.2.8).

4.5 DOE Proposal for Defining Exempt Concentrations of Uranium in Solid Wastes at the Y-12 Plant

The DOE has developed a proposal for exempting concentrations of uranium in solid wastes within DOE exclusion areas, including the Y-12 Plant in Oak Ridge.[51] The proposal is that concentrations of depleted, natural, or enriched uranium below 30 pCi/g may be placed in sanitary landfills within DOE exclusion areas without restrictions based upon radioactivity.

The recommendation of 30 pCi/g as a limit for defining exempt concentrations of uranium in solid wastes was not based explicitly on selection of an annual *de minimis* dose and calculation of the corresponding concentrations using a pathways analysis for shallow-land disposal. Rather, the exempt concentrations for uranium were based on the EPA's cleanup standards for inactive uranium processing sites,[15] which specify a limit on average ^{226}Ra concentrations in surface soils of 5 pCi/g, and an analysis by the NRC which concluded that the annual dose from 30 pCi/g of uranium in soil was comparable to the dose from 5 pCi/g of radium.[52]

In essence, the EPA's standards for cleanup of inactive uranium processing sites are based on the principle that is it unreasonable to require cleanup of naturally occurring radionuclides to levels that are below natural background for areas where the uranium ore was mined; i.e., requiring cleanup of radium only to levels approaching natural background is deemed ALARA. This is a reasonable approach, because requiring more stringent cleanup standards would not significantly reduce the risk from exposure to naturally occurring radionuclides. However, it may be questionable whether cleanup standards developed for sites in the western U.S. are appropriate for other locations where concentrations of naturally occurring radionuclides may be considerably lower.

It is important to recognize that the doses associated with the EPA's cleanup standards for inactive uranium processing sites, and thus with a limit on exempt concentrations of uranium for waste disposal of 30 pCi/g, may be relatively large compared with doses associated with exempt concentrations of other radionuclides for other activities as defined by the NRC[5,26-28] or with an annual *de minimis* dose equivalent of 1 mrem (0.01 mSv). For example, as noted by the EPA,[19] soils with radium concentrations of 5 pCi/g to a depth of several feet can produce annual external dose equivalents above ground of about 80 mrem (0.8 mSv).[53] The annual doses per unit concentration of ^{238}U given in Sections 4.2.1 and 4.2.3 of this report[30,45,46] yield estimated annual dose equivalents for a ^{238}U concentration of 30 pCi/g of about 20-80 mrem (0.2-0.8 mSv), assuming a waste density of 1.4 g/cm^3 (ref. 47). The NRC also has estimated the following annual dose equivalents from 30 pCi/g of ^{238}U in secular equilibrium with all its daughter products: about 100 mrem (1 mSv) to whole body from external exposure; about 65 mrem (0.65 mSv) to the lungs from inhalation of resuspended material; and about 400 mrem (4 mSv) to bone from ingestion of contaminated vegetation, beef, and milk. These results are discussed further in Section 5.3.

5. METHODOLOGY FOR DERIVING EXEMPT CONCENTRATIONS OF RADIONUCLIDES FOR DISPOSAL ON THE OAK RIDGE RESERVATION

This section presents a summary description of a methodology that has been developed for estimating annual committed effective dose equivalents to an inadvertent intruder per unit concentration of radionuclides for disposal on the Oak Ridge Reservation. The methodology is similar in many respects to those discussed in Section 4. The various methodologies differ primarily in (1) the exposure pathways included in the dose analysis for an inadvertent intruder, (2) the parameter values used in the models for each pathway, and (3) the use in some cases of dose equivalents to whole body instead of the effective dose equivalent.

Following a description of the methodology for the Oak Ridge Reservation in Section 5.1, Section 5.2 presents a comparison of annual doses per unit concentration of radionuclides obtained from the different methodologies reviewed in this report. Finally, Section 5.3 focuses on the annual doses per unit concentration obtained from the different methodologies for the uranium isotopes of importance to solid wastes generated at the Y-12 Plant in Oak Ridge.

5.1 Description of Methodology for the Oak Ridge Reservation

The methodology for estimating annual committed effective dose equivalents to an inadvertent intruder per unit concentration of radionuclides that has been developed for the Oak Ridge Reservation is described in detail in Appendix A of this report. The methodology assumes an intruder-agriculture scenario with the following exposure pathways:

- ingestion of contaminated drinking water;
- ingestion of vegetables grown in contaminated soil;
- ingestion of contaminated soil from the vegetable garden;
- ingestion of milk and meat from dairy and beef cattle that drink contaminated water;
- external exposure to contaminated soil; and
- inhalation of suspended activity from contaminated soil.

The assumptions used in calculating annual doses to an inadvertent intruder per unit concentration of radionuclides for each exposure pathway are described as follows.

[1] Drinking Water - An individual obtains all drinking water from a contaminated source on the disposal site, and the daily intake of water is 1 L.

[2] Vegetables - An individual obtains all vegetables from a garden on the disposal site, and the annual intake of vegetables is 90 kg. Contaminated soil from the disposal facility is mixed with native soil in the vegetable garden with a dilution factor of 0.2, and radionuclides are transferred to the vegetables via root uptake.

[3] Soil Ingestion - An individual consumes contaminated soil from the vegetable garden in conjunction with vegetable intakes, and the daily intake of soil is 0.1 g.

[4] Milk and Meat - An individual obtains all milk and meat from dairy and beef cattle that obtain all their drinking water from the same contaminated source on the disposal site that supplies drinking water for the individual. The annual intakes by the individual are 110 L of milk and 90 kg of beef, and the daily intakes of water by the dairy and beef cattle are 60 L and 50 L, respectively.

[5] External Exposure - An individual receives external exposures while working in the contaminated vegetable garden and while residing in a house that is constructed immediately on top of the disposal facility. The dose in both cases takes into account the shielding provided by the soil in which the radionuclides are mixed. The individual annually spends 100 h in the vegetable garden and 4380 h in the house, and the shielding factor during indoor residence is 0.7 for all photon-emitting radionuclides.

[6] Inhalation - An individual also receives inhalation exposures while working in the vegetable garden and residing in the house on the disposal facility. The airborne concentrations of radionuclides are described using a mass-loading approach, and the concentration of suspended soil is 10^{-6} kg/m^3 while working in the vegetable garden and 10^{-7} kg/m^3 while indoors. A dose-reduction factor during indoor residence of 0.24 is applied to all radionuclides except ^{3}H and ^{14}C.

Thus, for two of the assumed exposure pathways, the annual dose depends on the concentrations of radionuclides in water at the disposal site. For the remaining four exposure pathways, the annual dose depends on the concentrations of radionuclides in the disposal facility itself.

The calculations of annual dose per unit concentration of radionuclides in water or in the disposal facility are based on the assumption that exposures occur with a probability of unity at any time after loss of institutional controls over the facility. However, the calculations do not include explicit assumptions regarding reductions in concentrations due to radioactive decay during the period of institutional controls. A correction factor for decay is easily applied if, for example, one assumes that institutional controls are maintained for 100 years. In this case, the decay correction would be important for such radionuclides as ^{3}H, ^{90}Sr, and ^{137}Cs.

Because of the assumption of unit probability for the occurrence of intruder exposures at any time after loss of institutional controls, we have attempted to choose reasonable average values, rather than maximum possible values, for model parameters that describe transport of radionuclides through terrestrial foodchains and the intake rates and exposure times for an inadvertent intruder. We believe it is unreasonable to assume not only that an intruder will be exposed with unit probability at any time according to each of the assumed pathways, but also that an intruder will experience extreme intake rates and exposure times. This approach agrees with the recommendation of the ICRP that dose limits should apply to average individuals in the critical group of maximally exposed individuals.[4] Nonetheless, the calculations probably provide conservative overestimates of actual risks that will be experienced by individuals in the general public after loss of institutional controls over the facility.

The annual committed effective dose equivalents per unit concentration of radionuclides in water or in the disposal facility itself at the time intruder exposures occur, as obtained from the methodology for the Oak Ridge Reservation, are given in Tables 6 and 7, respectively. Results for additional radionuclides are given in Tables A-20 and A-21 of Appendix A.

The results for the drinking water pathways in Table 6 and in Table A-20 of Appendix A can be related to the annual dose per unit concentration of radionuclides in the disposal facility itself by means of models for leaching of wastes into water and transport to a source of water for an intruder. For disposal in trenches, which would most closely resemble disposal in a sanitary landfill, transport usually is assumed to occur via downward infiltration through soil to an underlying aquifer into which a well is drilled. For disposal in above-ground tumuli,[3] transport

Table 6. Annual doses to an inadvertent intruder per unit concentration of radionuclides in water obtained from methodology for Oak Ridge Reservation[a]

Nuclide	Annual dose[b] (mrem/y per μCi/L)
H-3	3.2E1
C-14	9.3E2
Ni-63	2.1E2
Sr-90	5.7E4
Tc-99	6.4E2
Cs-137	2.6E4
Ra-226	3.2E6
Th-232	1.8E6
U-233	1.1E5
U-234	1.1E5
U-235	9.8E4
U-238	9.9E4
Pu-238	1.4E6
Pu-239	1.6E6
Pu-242	1.5E6
Am-241	1.6E6
Am-243	1.6E6
Cm-244	8.5E5

[a]From Table A-20 of Appendix A for drinking water and milk and meat ingestion pathways.

[b]Results are given as annual committed effective dose equivalents per unit concentration in water at the time intrusion occurs; corrections for radioactive decay during period of institutional controls are not included.

Table 7. Annual doses to an inadvertent intruder per unit concentration of radionuclides in disposal facility obtained from methodology for Oak Ridge Reservation[a]

Nuclide	Annual dose[b] (mrem/y per $\mu Ci/m^3$)
H-3	3.9E-3
C-14	1.1E-3
Ni-63	2.4E-4
Sr-90	2.9E-1
Tc-99	9.4E-2
Cs-137	8.2E-1
Ra-226	3.7
Th-232	4.5
U-233	1.2E-2
U-234	1.2E-2
U-235	1.4E-1
U-238	3.5E-2
Pu-238	6.1E-2
Pu-239	6.9E-2
Pu-242	6.6E-2
Am-241	7.0E-2
Am-243	2.3E-1
Cm-244	3.4E-2

[a]From Table A-21 of Appendix A for vegetable, soil ingestion, external exposure, and inhalation pathways.

[b]Results are given as annual committed effective dose equivalents per unit concentration in disposal facility at the time intrusion occurs; corrections for radioactive decay during period of institutional controls are not included.

usually is assumed to occur via over-ground runoff to a nearby surface stream which serves as the source of water for an intruder.[45] The models for converting radionuclide concentrations in the disposal facility to concentrations in water may be highly complex and are quite site-specific. Results for disposal in trenches on the Oak Ridge Reservation are discussed in Sections 5.3.2-5.3.4 below.

5.2 Comparison with Results of Other Methodologies

Table 8 presents a comparison of the results from Table 7 for the vegetable, soil ingestion, external exposure, and inhalation pathways for disposal on the Oak Ridge Reservation with the results from the various methodologies reviewed in Section 4. Not shown in Table 8 is the result of Gilbert *et al.*[46] for $^{238}U + ^{234}U$ in secular equilibrium discussed in Section 4.2.3 of 1.8 mrem/y per $\mu Ci/m^3$ of ^{238}U. This result is discussed further in Section 5.3.1. The annual doses per unit concentration of radionuclides in the disposal facility apply in each case to the concentrations at the time intrusion occurs, not to the concentrations at the time of disposal. Thus, the correction factors used in some of the methodologies that account for radioactive decay during an assumed time period for institutional controls are not included in obtaining the results given in Table 8.

In comparing the results from the different methodologies, we would emphasize again that the results of EG&G Idaho[30] and Gilbert *et al.*[46] include ingestion of contaminated drinking water as well as the pathways involving activity in the disposal facility itself. The importance of the drinking water pathway to the total dose for each radionuclide is not indicated in the results reported by EG&G Idaho, but Table 5.6 of Gilbert *et al.* indicates that ingestion of contaminated drinking water contributes about 50% of the total dose from $^{238}U + ^{234}U$ in secular equilibrium. The calculations of Pin *et al.*[45] for the Central Waste Disposal Facility (CWDF) in Oak Ridge also include the drinking water pathway, and this pathway is the most important contributor to the dose for ^{3}H given in Table 8. The drinking water pathway also is the most important for ^{234}U, ^{235}U, and ^{238}U in the calculations of Pin *et al.* However, the dose from this pathway was estimated on the basis of an assumed limit on solubility of uranium in water, so the dose from each isotope is not proportional to its activity concentration in the disposal facility provided the solubility limit is exceeded. The dose from the drinking water pathway is not included in the results of Pin *et al.* for the uranium isotopes in Table 8. The other results in Table 8 include only pathways involving activity in the disposal facility itself. The methodology for the Oak

Table 8. Comparison of annual doses to an inadvertent intruder per unit concentration of radionuclides in disposal facility obtained by different investigators[a,b]

	Annual dose (mrem/y per $\mu Ci/m^3$)				
Nuclide	EG&G Idaho[c]	Jokerst[d]	Pin *et al.*[e]	NRC[f]	This report[g]
H-3	2.2E-4[h]	8.0E-5[h]	2.1E-2[i]	3.7E-3[i]	3.9E-3[h]
C-14	3.8	1.0E-5	4.0E-1	6.3E-4	1.1E-3
Ni-63	3.4E-4			2.8E-4	2.4E-4
Sr-90	3.2E-1	6.8E-6	6.1E-2	1.5E-1	2.9E-1
Tc-99	1.0E-2	1.9E-5	4.1E-2	1.7E-3	9.4E-2
Cs-137	1.8E1	1.4E-3	1.6E-1	5.0E-3	8.2E-1
Ra-226		4.2E-3			3.7
Th-232		4.8E-3			4.5
U-233					1.2E-2
U-234		4.5E-5	4.6E-1[j]		1.2E-2
U-235	1.2	3.6E-4	5.6E-2[j]		1.4E-1
U-238	5.7E-1	6.7E-5	7.5E-2[j]		3.5E-2
Pu-238		1.3E-4		4.0E-2	6.1E-2
Pu-239	5.6E-1	1.4E-4		4.8E-2	6.9E-2
Pu-240	5.7E-1			4.8E-2	
Pu-242	5.9E-1			4.5E-2	6.6E-2
Am-241	8.5E-1			7.4E-2	7.0E-2
Am-243	2.4			7.6E-2	2.3E-1
Cm-244				4.3E-2	3.4E-2

See next page for footnotes.

Footnotes for Table 8

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[a]Results apply to concentrations at the time intrusion occurs; corrections for radioactive decay during period of institutional controls that have been applied by some investigators are not included.

[b]An additional result of Gilbert *et al.*[46] is that for $^{238}U + ^{234}U$ in secular equilibrium, the dose equivalent to whole body is 1.8 mrem/y per $\mu Ci/m^3$ of ^{238}U.

[c]Obtained from results in Table 2.

[d]Obtained from results in Table 3.

[e]Obtained from results in Table 4.

[f]Obtained from results in Table 5.

[g]Results from Table 7 for vegetable, soil ingestion, external exposure and inhalation pathways; see Sections 5.3.2 and 5.3.4 for discussion of results from drinking water pathway for uranium isotopes.

[h]Results are given as annual committed effective dose equivalents.

[i]Results are given as annual committed dose equivalents to whole body.

[j]Results for vegetable, external exposure, and inhalation pathways only; see Table 4 for results from drinking water pathway.

Ridge Reservation presented in this report also includes consideration of the drinking water pathway, and the calculations for this pathway for the uranium isotopes also are based on an assumed solubility limit in water. The effect of a solubility limit on the dose for the uranium isotopes is discussed further in Sections 5.3.2 and 5.3.4.

In Table 8, we find reasonable agreement in many cases between the results of the methodology for the Oak Ridge Reservation presented in this report and the results of the NRC.[17,22] The results of EG&G Idaho[30] and this report also compare favorably in many cases. As discussed in Section 4.2.2, there appears to be a problem with the results of Jokerst[31] which tend to be quite different from all other calculations for most radionuclides. The large discrepancies with the results of the NRC are particularly puzzling, because Jokerst's work purportedly was based to a large extent on the models developed by the NRC.

The differences among the various methodologies other than Jokerst's reflect ifferences in the exposure scenarios, environmental parameters, and dosimetric data assumed by the different investigators. For example, the dose from ^{137}Cs is determined primarily by the external exposure pathway, and the calculations are sensitive to the assumed distribution of activity in soil. The transport of ^{14}C through terrestrial foodchains is quite uncertain, so it is not surprising that different investigators obtain significantly different results for this radionuclide. For ^{90}Sr and ^{99}Tc, the dose is most sensitive to the assumed value for the plant-to-soil concentration factor describing root uptake by vegetation, and there are wide ranges of reported values for this parameter. Finally, EG&G Idaho[30] and the methodology for the Oak Ridge Reservation presented in this report calculate effective dose equivalents, whereas Pin *et al.*[45] and the NRC[17,22] calculated doses to whole body. The differences between these two dosimetric quantities can be significant for alpha-emitting radionuclides that do not irradiate the body uniformly. On the whole, however, most of the methodologies appear to give reasonably comparable results when the various differences are taken into account.

5.3 Comparison of Results for Uranium Isotopes

This section presents a comparison of annual dose equivalents per unit concentrations of ^{238}U in the disposal facility, as obtained by EG&G Idaho,[30] Gilbert *et al.*,[46] and the methodology for the Oak Ridge Reservation presented in this report. We also briefly discuss the calculation of annual dose equivalents per unit concentration for ^{234}U and ^{235}U in the presence of ^{238}U. The discussion particularly emphasizes the difficulties in the calculations that arise from (1) the assumption of a

solubility limit for uranium in water and (2) the possibility of long-term buildup of daughter products of ^{234}U in the disposal facility which could lead to significant increases in doses to intruders at far future times.

5.3.1 *Results of EG&G Idaho and Gilbert et al. for U-238*

As discussed in Sections 4.2.1 and 4.2.3, the calculations of EG&G Idaho[30] and Gilbert *et al.*[46] include the drinking water pathway as well as pathways involving exposure of an inadvertent intruder to ^{238}U in the disposal facility itself. However, only Gilbert *et al.* indicate explicitly the importance of the drinking water pathway to the total dose per unit concentration, and the contribution from this pathway is about 50%. We also recall that Gilbert *et al.* estimate the dose from ^{238}U + ^{234}U per unit concentration of ^{238}U by assuming that the two isotopes are in secular equilibrium.

Each of these methodologies uses a similar approach for obtaining concentrations of ^{238}U or ^{238}U + ^{234}U in an aquifer below the disposal facility. Each uranium isotope is assumed to be leached by infiltrating water at a rate proportional to its concentration in the solid wastes, and the resulting solute concentration propagating downward through soil is inversely proportional to the retardation factor for uranium in the soil/water system. Thus, the resulting concentration of each isotope in drinking water is proportional to its concentration in the disposal facility, irrespective of the concentration of any other uranium isotopes, and the constant of proportionality is essentially equal to the reciprocal of the product of the dilution factor for transport of water from the disposal facility to the location at which water is consumed (i.e., a well below the disposal facility) and the retardation factor for uranium. It is reasonable to assume that the travel time of ^{238}U to an aquifer is sufficiently short that radioactive decay is not important in reducing the solute concentration.

For the leaching and water-transport scenario described above, the annual dose equivalents per unit concentration for the drinking water pathway and for the pathways involving activity in the disposal facility itself are additive, and the results of EG&G Idaho and Gilbert *et al.* in Table 8 give the total annual dose equivalent from all pathways for any concentration of ^{238}U in the facility. We repeat these results below:

Annual dose equivalent per unit concentration of ^{238}U –
EG&G Idaho: 0.57 mrem/y per $\mu Ci/m^3$ (dose from ^{238}U only);
Gilbert *et al.*: 1.8 mrem/y per $\mu Ci/m^3$ (dose from ^{238}U + ^{234}U).

The dose in each case is the annual committed effective dose equivalent.

Again, the EG&G Idaho result applies only to ^{238}U, whereas the result of Gilbert *et al.* gives the annual dose from $^{238}U + ^{234}U$ in secular equilibrium per unit concentration of ^{238}U. In the calculations of Gilbert *et al.*, the dose from $^{238}U + ^{234}U$ arises almost entirely from internal exposures. Therefore, since the dose commitment per unit intake via inhalation or ingestion is nearly the same for these two isotopes (see Tables A-2 and A-3 in Appendix A), each isotope contributes about half of the dose from $^{238}U + ^{234}U$ in secular equilibrium. Thus, for ^{238}U alone, the results of EG&G Idaho and Gilbert *et al.* agree well within a factor of two.

5.3.2 *Results of Oak Ridge methodology for U-238*

The calculations for the drinking water pathway described in Section 5.3.1 do not consider the possibility that the concentration of uranium in the leachate may be limited by the solubility of uranium in water, in which case the mass concentration of uranium in solution will be a constant for any concentration in solid wastes in the disposal facility above the solubility limit. The resulting concentration in drinking water and the annual dose from this pathway then would not be proportional to the concentration of uranium in the disposal facility. Solubility limits for uranium and other elements were used by Pin *et al.*[45] in the dose analysis for the CWDF in Oak Ridge, and the same model was adopted in the methodology for the Oak Ridge Reservation presented in this report.

Pin *et al.*[45] assumed a solubility limit for uranium of 45 μmol/L. For ^{238}U, which has a specific activity of 3.37×10^{-7} Ci/g, the resulting limit on leachate concentration in the disposal facility is 3.6×10^{-3} μCi/L, or 3.6 pCi/cm^3 (3.6 μCi/m^3). Further, for leachate concentrations near the solubility limit, the equilibrium distribution coefficient (K_d) for uranium in the soil/water system was assumed to be 1 L/kg.[45] The minimum uranium concentration in solid waste in soil that would result in a leachate concentration at the solubility limit then is 3.6 pCi/g, or about 5 μCi/m^3 for a soil density of 1.4 g/cm^3. Thus, the proposed exempt concentration of uranium in solid wastes at the Y-12 Plant in Oak Ridge of 30 pCi/g is considerably above the minimum value that would result in solubility-limited leachate concentrations.

For disposal in trenches, Pin *et al.*[45] further assumed that the effects of dilution and retardation in transport from the disposal facility to an underlying aquifer reduce the solubility-limited leachate concentration in the disposal facility by a factor of about 22. Thus, the resulting concentration of ^{238}U in the aquifer for any concentration in the disposal facility above the solubility limit is 1.6×10^{-4} μCi/L.

Application of the annual dose per unit concentration of ^{238}U in water from Table 6 gives an annual committed effective dose equivalent from ^{238}U of 16 mrem for the drinking water pathway. Again, this result is independent of the concentration of ^{238}U in the disposal facility, provided the concentration exceeds the solubility limit.

For the pathways involving activity in the disposal facility itself, the annual dose from ^{238}U is proportional to the concentration in the solid wastes, and the annual doses in Table 7 apply to all concentrations. Thus, the results of the Oak Ridge methodology for ^{238}U can be summarized as follows:

^{238}U concentrations in trench above 5 $\mu Ci/m^3$ –
Drinking water pathway: 16 mrem/y independent of concentration;
Other pathways: 0.035 mrem/y per $\mu Ci/m^3$.

Again, the doses are given as annual committed effective dose equivalents. The estimated doses apply to ^{238}U only and do not include possible contributions from ^{234}U as in the methodology of Gilbert *et al.*[46]

5.3.3 *Comparison of different methodologies for U-238*

In comparing the results of EG&G Idaho[30] and Gilbert *et al.*[46] given in Section 5.3.1 with those from the model for the Oak Ridge Reservation given in Section 5.3.2, we see that the comparison depends on the assumed concentration of ^{238}U in the disposal facility. For example, for a ^{238}U concentration of 30 pCi/g, which corresponds approximately to 40 $\mu Ci/m^3$, the following annual committed effective dose equivalents are obtained from the different methodologies:

^{238}U concentration in trench of 30 pCi/g –
EG&G Idaho: 23 mrem (dose from ^{238}U only);
Gilbert *et al.*: 72 mrem (dose from ^{238}U + ^{234}U);
Oak Ridge: 17 mrem (dose from ^{238}U only).

Again, in the calculations of Gilbert *et al.*, the dose from ^{238}U alone is about half of the value listed above. At the low concentration of 30 pCi/g, the dose from the Oak Ridge methodology is determined almost entirely by the drinking water pathway. As the concentration of ^{238}U in the disposal facility increases, the doses from the EG&G Idaho and Gilbert *et al.* methodologies increase in proportion to the concentration, but the Oak Ridge methodology gives doses that increase more slowly because the dose from the drinking water pathway is independent of ^{238}U concentration in the disposal facility.

5.3.4 *Calculations for U-234 and U-235*

The estimation of annual doses per unit concentration of ^{234}U and ^{235}U also depends on the model used to estimate uranium concentrations in the leachate in the disposal facility. If no limit on uranium solubility in water is assumed, then the concentrations of the different uranium isotopes in water can be calculated independently of one another and are proportional to the respective isotope concentrations in the solid wastes, as described in Section 5.3.1. Thus, the contributions from the drinking water pathway and the other pathways involving activity in the disposal facility again are additive for each isotope, and the total dose from all pathways for each isotope is proportional to the concentration of that isotope in the solid wastes irrespective of the concentrations of other uranium isotopes.

On the other hand, if the leachate concentration of uranium is solubility limited, then the leachate concentrations of the different uranium isotopes are not independent of one another and are not proportional to the total concentration of uranium in the disposal facility. Rather, the amount of each isotope in the leachate will be proportional to the fractional abundance <u>by mass</u> of each isotope in the solid wastes. Let us consider natural uranium as an example. For concentrations in the disposal facility above the solubility limit of 45 μmol/L, the mass concentrations of the different isotopes in the leachate are proportional to their natural abundances of 0.0055% for ^{234}U, 0.72% for ^{235}U, and 99.27% for ^{238}U. Since ^{238}U and ^{234}U are in secular equilibrium in natural uranium, the resulting activity concentrations for both ^{238}U and ^{234}U in an aquifer below a disposal trench, as obtained from the model for the CWDF on the Oak Ridge Reservation described in Section 5.3.2, are 1.6×10^{-4} μCi/L, and the concentration of ^{235}U is 7.6×10^{-6} μCi/L. Thus, from the results in Table 6, the annual committed effective dose equivalents from the drinking water pathway are 16 mrem for ^{238}U, 18 mrem for ^{234}U, and 0.7 mrem for ^{235}U. The dose from ^{235}U for the drinking water pathway thus can be neglected compared with the contributions from ^{238}U and ^{234}U.

When the concentration of uranium in the leachate is solubility limited, one cannot calculate the dose from the drinking water pathway without knowledge of the fractional abundance by mass of each isotope in the disposal facility. Uranium that is enriched in $^{234}U/^{235}U$ will result in higher doses from the drinking water pathway per unit concentration of ^{238}U than for natural or depleted uranium. Conversely, depleted uranium will result in lower doses from the drinking water pathway per unit concentration of ^{238}U than for natural uranium.

In estimating intruder doses for the exposure pathways involving activity in the disposal facility itself, the contributions from the different uranium isotopes can be obtained independently of one another using the results in Table 7. For natural uranium, for example, the annual committed effective dose equivalents from these pathways per $\mu Ci/m^3$ of ^{238}U in the trench are 0.035 mrem for ^{238}U, 0.012 mrem for ^{234}U, and 0.0064 mrem for ^{235}U. The contribution from ^{238}U thus is the most important for these pathways.

5.3.5 *Effects of buildup of uranium daughter products*

A further complication in estimating doses from disposal of ^{238}U and ^{234}U involves the long-term buildup of radiologically significant daughter products, particularly ^{226}Ra and the daughter products of ^{222}Rn. Possible contributions to intruder doses from ^{234}U daughter products are not considered in any of the calculations presented in Sections 5.3.1-5.3.4. From the results from the Oak Ridge methodology in Tables 6 and 7, it appears that if ^{226}Ra is in secular equilibrium with ^{238}U + ^{234}U at the time exposures occur, then the dose from the uranium daughters will greatly exceed the dose from the uranium isotopes themselves. It is important to note, however, that this conclusion is based on the assumption that none of the ^{222}Rn produced by the decay of ^{226}Ra in soil escapes to the atmosphere, so that the calculated dose from ^{226}Ra and daughter products probably provides a conservative overestimate of actual doses to an intruder.

The potential importance of ^{226}Ra and its daughters to intruder doses depends on the rate at which the original uranium in the wastes is removed from the disposal facility by leaching or other physical processes. As shown in Fig. 2, the buildup of ^{226}Ra and its daughters from an initial inventory of pure ^{238}U requires tens of thousands of years. Similarly, as shown in Fig. 3, buildup from an initial inventory of ^{238}U + ^{234}U in secular equilibrium requires thousands of years. Thus, if the initial inventory of uranium is removed from the disposal facility before significant buildup of the daughter products can occur, then the daughter products will not contribute significantly to doses to an inadvertent intruder from pathways involving activity in the facility. The removal of uranium via leaching is particularly likely to occur at disposal sites with abundant rainfall, such as the Oak Ridge Reservation.

For the drinking water pathway from trench disposal, it is likely that only the uranium isotopes will reach an underlying aquifer in significant concentrations, because the ^{230}Th produced in the disposal facility or during uranium transport through soil should be highly

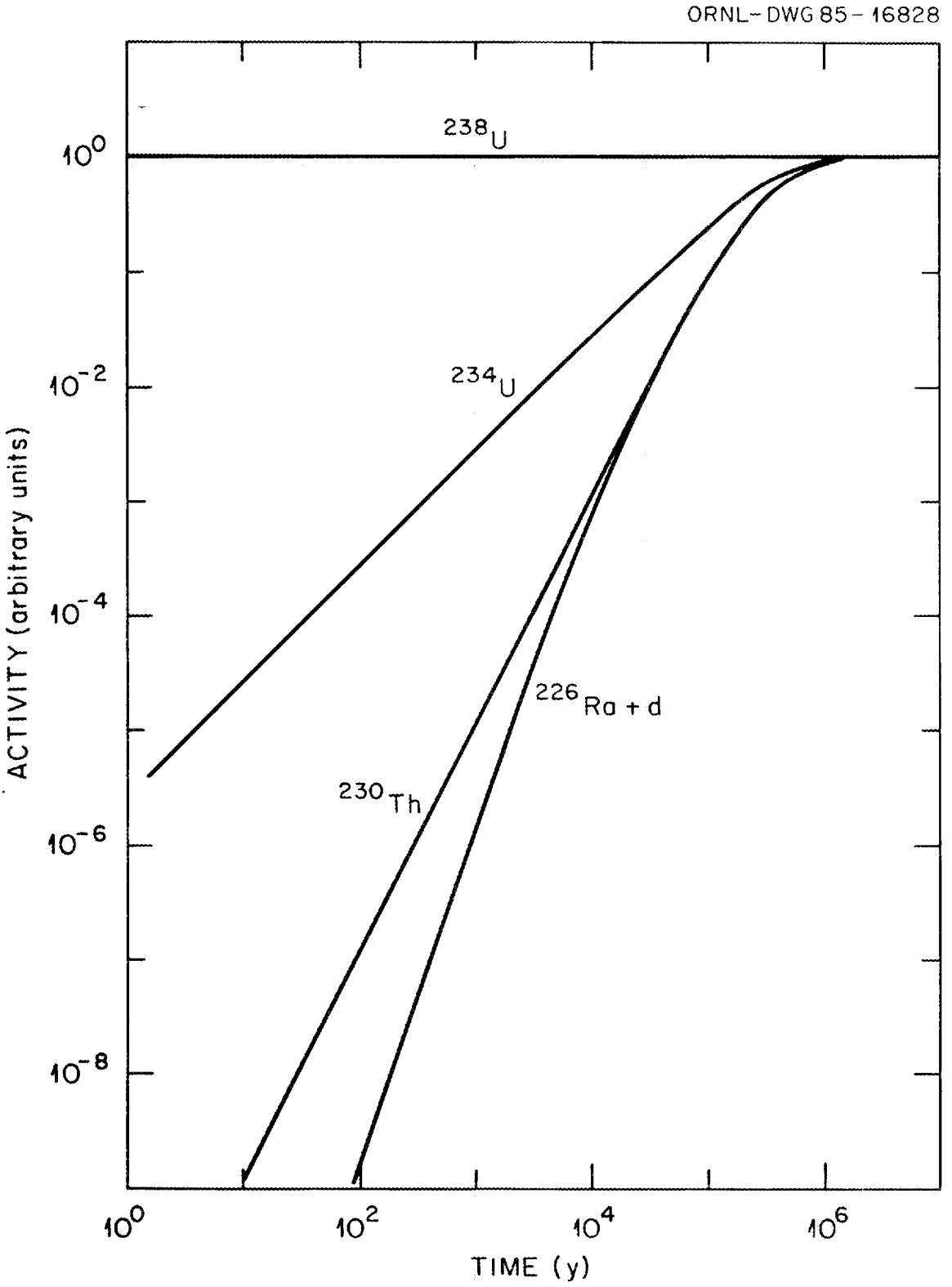

Fig. 2. Activity of U-238 and daughter products vs time for an initial activity of pure U-238.

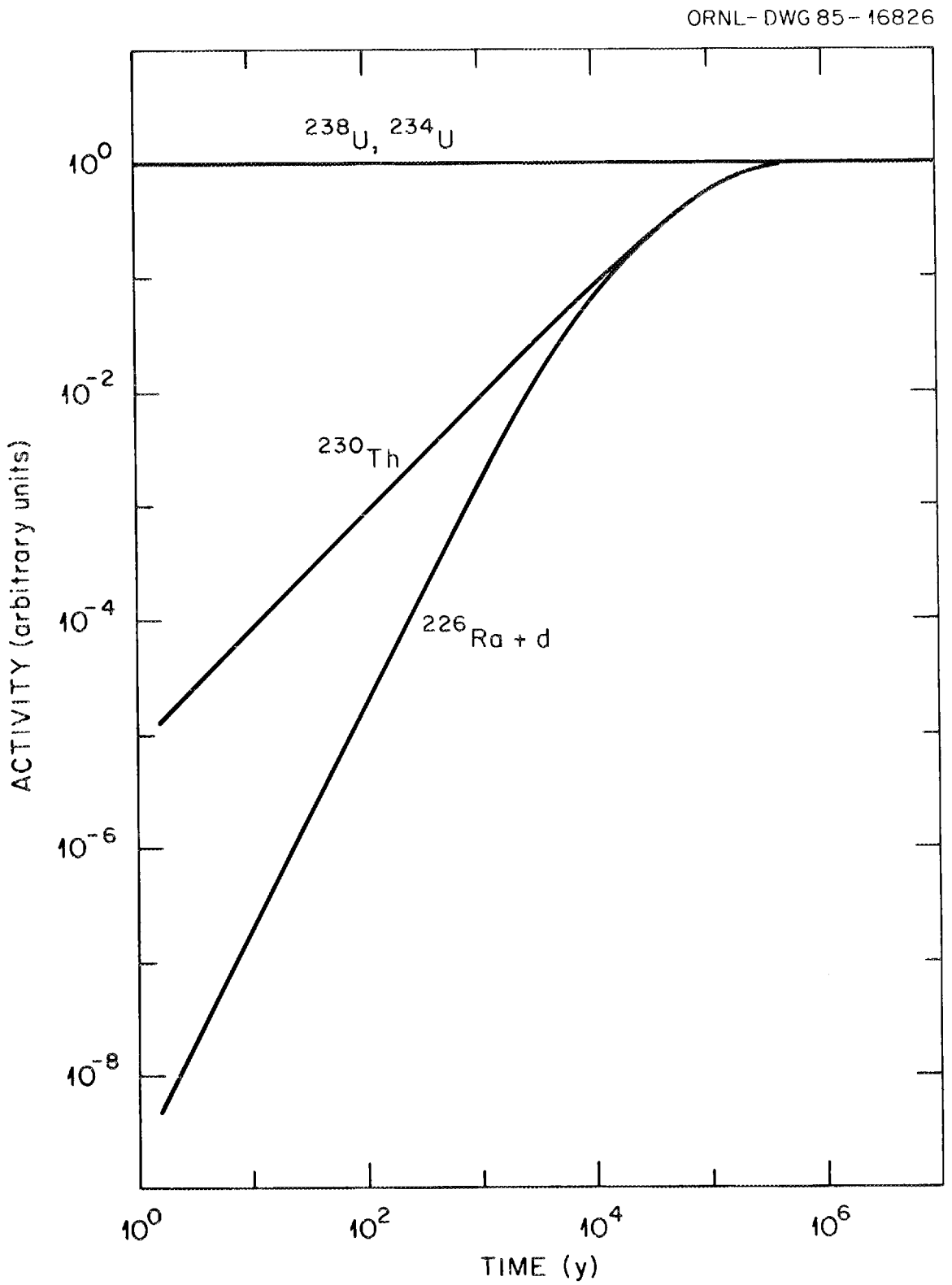

Fig. 3. Activity of U-238 and daughter products vs time for an initial activity of U-238 and U-234 in secular equilibrium.

insoluble and highly retarded[45] and, thus, should not reach an aquifer before decay to ^{226}Ra. Since ^{226}Ra has a relatively short half-life and is also somewhat retarded in water transport through soil,[45] the concentrations of ^{226}Ra and daughter products in an aquifer should be much less than the concentrations that would result from radioactive decay of the uranium parent alone.

If the initial inventory of uranium in the disposal facility is expected to remain there for thousands of years or more, then the contributions from uranium daughters should be considered in evaluating intruder doses from the pathways involving activity in the facility. Since the drinking water pathway is unimportant if no leaching occurs, the maximum dose from ^{238}U and daughter products at any time in the future can be obtained from the results for the Oak Ridge methodology in Table 7. Thus, we obtain the following result for a no-leaching scenario:

> ^{238}U + daughters in secular equilibrium in the disposal facility – Oak Ridge: 4 mrem/y per $\mu Ci/m^3$.

The annual committed effective dose equivalent is normalized to an initial concentration of ^{238}U in the trench. Again, this result assumes no escape of ^{222}Rn from soil. For this scenario, the maximum annual dose to an intruder at any time in the future for an initial concentration of ^{238}U of 30 pCi/g then would be about 160 mrem. We re-emphasize, however, that such a no-leaching scenario is credible only for disposal in arid sites, because isolation of the wastes from infiltrating water using engineered barriers probably cannot be demonstrated for tens of thousands of years for disposal at non-arid sites.

5.4 Summary of Results for Disposal of Uranium on the Oak Ridge Reservation

The estimation of doses to an inadvertent intruder from trench disposal of long-lived isotopes of uranium is complicated by the fact that scenarios for leaching of wastes into water and transport to an underlying aquifer need to be treated on a site-specific basis. The estimated concentrations of uranium in an aquifer depend on such factors as the rate of infiltration of water through the disposal facility, the extent to which water contacts uranium in the wastes, the leachability of the materials containing uranium, the solubility of uranium in water, the equilibrium distribution coefficient for uranium in the soil/water system, the distance of water travel from the disposal facility to the aquifer, the dilution of contaminated water in travel to the aquifer, and the retardation factor for uranium in the soil/water system. If the

mobilization of uranium is solubility limited, then the resulting dose from the drinking water pathway will not be proportional to the concentration of uranium in solid wastes in the disposal facility.

Using the model of Pin *et al.*[45] for mobilization and water transport of uranium for trench disposal at a site on the Oak Ridge Reservation, we estimate an annual committed effective dose equivalent from ^{238}U in drinking water of 16 mrem independent of the concentration in the disposal facility, provided the concentration in the solid waste is greater than about 5 $\mu Ci/m^3$; above this concentration, the leachate concentration in the trench will be solubility limited. This concentration is considerably less than the proposed exempt concentration of 30 pCi/g for uranium wastes at the Y-12 Plant in Oak Ridge. For the pathways involving exposure to ^{238}U in the disposal facility itself, we estimate an annual committed effective dose equivalent of 0.035 mrem/y per $\mu Ci/m^3$. Thus, for a ^{238}U concentration of 30 pCi/g, which is about 40 $\mu Ci/m^3$, the annual committed effective dose equivalent to an inadvertent intruder is estimated to be 17 mrem and is due almost entirely to the dose from the drinking water pathway. For ^{238}U + ^{234}U in secular equilibrium, and for natural uranium as well, the contribution to the dose for the drinking water pathway from ^{234}U is about 10% greater than that from ^{238}U, and the annual committed effective dose equivalent for a ^{238}U concentration in the trench of 30 pCi/g would be about 34 mrem for the drinking water pathway, independent of the total uranium concentration in the disposal facility. At this concentration, the dose from exposures to the contents of the facility itself again would be negligible. For natural uranium, the contributions to the dose from ^{235}U would be relatively small for all pathways compared with the contributions from ^{238}U and ^{234}U.

If uranium in the disposal facility is not removed by leaching or other processes over tens of thousands of years, then the concentrations of uranium daughter products (principally ^{230}Th, ^{226}Ra, and short-lived daughters of ^{226}Ra) in the facility will increase with time until secular equilibrium is achieved, and the resulting dose to an intruder may greatly exceed the dose from the uranium isotopes alone. For ^{226}Ra in equilibrium with ^{238}U and ^{234}U in the facility and assuming no escape of ^{222}Rn from soil, we estimate an annual committed effective dose equivalent of 4 mrem/y per $\mu Ci/m^3$ of ^{238}U, due almost entirely to ^{226}Ra and daughter products. Thus, an initial concentration of ^{238}U of 30 pCi/g in the disposal facility could produce annual doses to an inadvertent intruder in excess of 150 mrem, but only for times after disposal exceeding 10^5 years.

For disposal in a wet environment such as Oak Ridge, however, a no-leaching scenario for disposal of uranium seems highly unlikely. It is more likely that infiltrating water would remove most of the uranium from the disposal facility before the buildup of ^{226}Ra would reach levels near

secular equilibrium with the uranium parents, so the resulting annual dose equivalents to an inadvertent intruder probably would be considerably less than 100 mrem. Thus, it clearly is important to evaluate the long-term retention and transport of uranium placed in the disposal facility on a site-specific basis.

An additional complicating factor for estimating intruder doses at far future times from the buildup of uranium daughter products is the potential for escape of ^{222}Rn from soil, which is not taken into account in the methodology for the Oak Ridge Reservation. This effect would increase inhalation doses, but would also decrease doses for the other pathways from the ^{222}Rn daughter products. With regard to the potential for increases in inhalation doses, we note that the NCRP has estimated that inhalation of radon daughter products during indoor residence presently contributes about half of the total average annual committed effective dose equivalent from natural background radiation in the U.S.[11,12] In addition, if the ^{222}Rn is produced at depths in soil of a few meters or more, then escape from the soil to the atmosphere before radioactive decay occurs is unlikely.

6. SURVEY OF METHODS FOR MEASURING URANIUM CONTENT IN BULK SOLID WASTES

6.1 Introduction

If a limit on exempt concentrations of uranium in solid wastes is established, then reliable measurement techniques probably will be needed to ensure that wastes placed in a sanitary landfill contain uranium concentrations below the limit. This section briefly discusses a variety of methods for assaying bulk solid wastes for uranium content and their associated costs.

The uranium content of bulk solid wastes can be determined either by passive photon detection or by active pulsed thermal-neutron interrogation. These techniques differ both in the type of radiation measured (photons vs fission neutrons) and in the uranium isotope assayed (^{238}U vs ^{235}U). The thermal-neutron method takes advantage of the fact that all uranium (including that which is depleted) contains ^{235}U, and the method is quite sensitive to the presence of small amounts of the fissile material. Both techniques have been well characterized and are in use at a number of facilities. Passive photon detection is being developed for use at the Y-12 Plant in Oak Ridge[54] and is being used in assay systems for transuranic (TRU) wastes at Oak Ridge National Laboratory and elsewhere.[55-57] Active pulsed thermal-neutron interrogation is being used, in conjunction with active and passive photon detection and passive neutron detection, for assaying drums and large containers of TRU wastes.[55,58-60]

6.2 Existing Measurement System at the Y-12 Plant

Passive photon detection systems are being developed at the Y-12 Plant for determining the uranium content of dumpsters and barrels containing solid waste materials. The dumpster assay system uses two shielded and collimated 12.7-cm NaI(Tl) detectors and associated electronics, with each detector facing the opposite sides of a dumpster. The barrel assay system consists of a single detector of the same type and a table for rotating the barrel being assayed. The dumpster and barrel assay systems are based on detection of 766- and 1001-keV photons from beta decay of the short-lived isotope ^{234m}Pa, which is produced in the decay of ^{238}U. Thus, an obvious disadvantage of this method is that the photons being detected are ubiquitous in natural background radiation.

As reported in ref. 54, a test system consisting of a single detector has produced rather uncertain estimates of uranium content in controlled measurements on full and empty dumpsters. For example, the measured uranium content of an empty dumpster varied by as much as a factor of 17 depending on the location of the uranium within the dumpster; i.e., the number of counts above background in the detector was about a factor of 17 higher for a point source located at the center of the dumpster than for the same source located in a corner. Although the data indicated that masses of depleted uranium in the dumpster as low as 37 g could be detected, it was not possible to differentiate quantitatively between amounts of uranium for masses below 100 g. The measurement uncertainty is due primarily to the large size of a dumpster, but variations in the waste composition in a particular dumpster could further increase the uncertainty and the detection limit for the measurements. Although use of two detectors should yield better accuracy than the test system involving a single detector, geometrical contraints (i.e., the use of fixed detectors to assay a large volume of waste) may severely limit the accuracy of the technique.

More recent studies of the detection system at the Y-12 Plant indicate that the lower limit of detection for depleted uranium in a dumpster is slightly less than 200 g at the 95% confidence level (K. F. Symon, private communication). This would correspond to about 120 pCi/g for a typical trash loading, and is about a factor of 4 higher than the proposed exemption level for uranium in solid wastes of 30 pCi/g (see Section 4.5).

The performance of the dumpster and barrel assay systems could be improved by providing for better geometrical coverage of the waste volumes by the photon detectors. Three alternatives appear to warrant further study. First, a sufficient number of detectors can be added to the system to limit the measurement uncertainty for empty dumpsters to an acceptable level (e.g., ±50%). The optimum number of detectors could be predicted from geometrical considerations and verified empirically. For example, the use of three vertically aligned detectors should improve the performance of the barrel assay system. Second, for the dumpster assay system, a significant improvement in measurement capabilities should be obtained by placing a number of vertically aligned detectors on each side of the dumpster and moving the source horizontally past the detectors during a measurement. Third, the two detectors in the present dumpster assay system could be placed above the dumpster and the source moved horizontally past the detectors. For the barrel assay system, the single detector could be placed above the barrel and the source rotated about a vertical axis. The choice of suitable modifications of the present systems presumably would be determined by comparing the costs of using

additional detectors with a fixed source with the costs of relocating the present detectors and, in the case of the dumpsters, moving the source during the measurement. The use of additional detectors is being studied in test measurements (K. F. Symon, private communication).

6.3 Segregation at the Point of Waste Generation

The results of a dumpster survey given in Table 8 of ref. 54 suggest that it could be cost-effective to identify contaminated wastes by surveying and segregating the wastes at the point of generation. In the dumpster survey, components that occupied at least half of the volume of the dumpster apparently were not contaminated and seem unlikely to be contaminated. Portable photon detectors could be used to segregate contaminated and uncontaminated wastes at the point of generation. Several instruments are available for this purpose at costs ranging from $500 to $1,500.[61]

Implementation of a waste segregation program would require the services of a waste surveyor as well as administrative controls on the waste streams from each potential source. The uranium content of dumpsters filled with segregated wastes likely would require verification by one of the bulk assay techniques before final disposal. Nonetheless, waste segregation is a promising method for reducing the volume of contaminated wastes that would be sent to a radioactive waste disposal facility.

6.4 An Alternative System for Uranium Assay

An alternative to photon measurements from the decay of ^{238}U for determining the uranium content of wastes is active pulsed thermal-neutron interrogation of the ^{235}U contained in depleted, natural, or enriched uranium. This technique has been adapted successfully to the determination of quantities of fissile materials in TRU wastes. Systems are currently in operation that can detect 1-mg quantities of fissile materials in standard 208-L barrels filled with a variety of common wastes.[57,58] We note that 1 mg of ^{235}U corresponds to 0.36 g of depleted (0.28%) uranium or 0.033 g of enriched (3%) uranium, so the method apparently is quite sensitive to the presence of small amounts of uranium in the wastes if no other fissile materials are present. Systems capable of assaying containers that are much larger than a standard dumpster also are in use,[59,60] so the size of the source is not a limitation on use of the method. The overall accuracy of the method for a large container

system appears to be about ±50%.

A pulsed thermal-neutron interrogation system consists of a specially designed graphite- and polyethylene-moderated chamber that contains a commercially available source of 14-MeV neutrons, the waste container, a bank of cadmium-shielded fast-neutron detectors, and a bare ^{3}He proportional counter. System operation involves generation of a pulse of 14-MeV neutrons that are rapidly moderated to thermal velocities. The thermal neutrons then are absorbed by the fissile material in the wastes, e.g., by the ^{235}U present in all uranium. The resulting neutrons from prompt fission of the ^{235}U are detected with a high probability. Further details of system operation can be found in refs. 57-60.

The costs of the pulsed thermal-neutron interrogation system will vary with the size of the containers to be assayed. A barrel assay system developed in 1981 cost $136,000.[58] A more recent estimate is $250,000 for the barrel assay system at Oak Ridge National Laboratory and $450,000 for a comparable dumpster assay system (F. J. Schultz, private communication). These costs do not include those associated with operation and maintanence of the system.

6.5 Summary

From our review of current efforts to establish a system for assaying large volumes of uranium-contaminated wastes at the Y-12 Plant based on passive photon detection of a decay product of ^{238}U, it is not yet evident that the proposed system is capable of measuring with reasonable accuracy and on a routine basis concentrations as low as 30 pCi/g, which is the proposed limit on exempt concentrations of uranium in solid wastes. In order for an exempt concentration of 30 pCi/g to achieve widespread acceptance, it probably will be necessary to measure such concentrations with reasonable accuracy (e.g., ±50%).

The present passive photon detection system probably could be improved significantly by use of an increased number of detectors, different detector locations relative to the source volume, and movement of the sources past a fixed detector system. A system involving assay and segregation of the wastes at the source of generation also may lead to substantial reductions in the volume of materials that would need to be sent to a radioactive waste disposal facility. These methods would not entail great costs or major changes in system design.

A system based on active pulsed thermal-neutron interrogation of ^{235}U in the wastes has been developed and used successfully by a number of investigators. This system appears to be capable of detecting much smaller quantities of uranium than the passive photon detection system,

because problems with relatively high background levels of radiation that limit the capabilities of the photon detection system are largely eliminated. While the costs of existing thermal-neutron systems are high relative to the costs of the present photon detection system, a system that is satisfactory for use at the Y-12 Plant may be considerably less expensive than the systems that have been developed elsewhere.

7. SUMMARY AND CONCLUSIONS

This report has discussed a number of issues associated with the determination of exempt quantities of radionuclides in solid waste materials. General acceptance of exemption levels for radioactivity in solid wastes could lead to significant reductions in the required capacity and associated costs of facilities for storage and disposal of low-level radioactive wastes, because exempt materials could be handled in all respects as if they were nonradioactive.

An important aspect of this report has been the discussion of a methodology developed for application to waste disposal on the Oak Ridge Reservation that relates concentrations of radionuclides in solid waste materials to radiation doses that might be received by individuals who inadvertently intrude onto the disposal site after loss of institutional controls. The methodology provides estimates of annual committed effective dose equivalents per unit concentration of radionuclides placed in a disposal facility and can be applied in two ways. First, if regulatory authorities establish a generally applicable limit on radiation dose to members of the general public that is *de minimis* or below regulatory concern, then the methodology can be used to derive limits on exempt concentrations of radionuclides for purposes of waste disposal. Alternatively, if limits on exempt concentrations of radionuclides for purposes of waste disposal can be determined on some basis other than a *de minimis* dose, e.g., by application of the ALARA principle to waste disposal, then the methodology can be used to estimate annual dose equivalents that would result from disposal of exempt concentrations of different radionuclides.

While the dose assessment methodology for waste disposal on the Oak Ridge Reservation can be applied to any radionuclide, this report has focused on the determination of annual dose equivalents per unit concentration that would result from disposal of uranium-bearing wastes generated at the Y-12 Plant. A limit on exempt concentrations of 30 pCi/g for depleted, natural, or enriched uranium has been proposed for use within DOE exclusion areas.[51] This concentration limit is based, first, on the cleanup standard of 5 pCi/g established by the EPA for ^{226}Ra in soils at inactive uranium processing sites[15,19] and, second, on an analysis by the NRC which concluded that annual doses associated with disposal of 30 pCi/g of uranium would be comparable to those from disposal of 5 pCi/g of radium. The proposed limit on exempt concentrations of uranium in solid wastes thus results from an application of the ALARA principle to the cleanup of naturally occurring radionuclides, rather than a generally applicable *de minimis* dose.

The principal conclusions and recommendations obtained from the analyses presented in this report are summarized in the following paragraphs.

[1] The concept of a generally applicable *de minimis* dose for members of the general public must be understood within the context of established limits on acceptable dose from all sources of exposure, limits on dose from specific practices (e.g., low-level waste disposal), and application of the ALARA principle to specific sites and specific practices. A *de minimis* dose corresponds to a level of risk that would be regarded as negligible by most individuals and is a dose below which control of radiation exposures would be deliberately and specifically curtailed.[1] Thus, a *de minimis* dose constitutes a lower limit for application of the ALARA principle to any site and any practice, and a *de minimis* dose must be set well below the limit on acceptable dose from all sources of exposure and below any dose from specific practices that is of concern to regulatory authorities. The establishment of a generally applicable *de minimis* dose would provide the most defensible basis for determining exempt concentrations of radionuclides for purposes of waste disposal.

[2] Exempt levels of radioactivity for specific practices can be based on application of the ALARA principle to those practices. This procedure may result in doses that greatly exceed a generally applicable *de minimis* dose. For example, the EPA's cleanup standard for radium in soil described above results in an estimated annual dose equivalent of about 80 mrem.[19] Furthermore, the doses associated with exempt quantities of radionuclides may differ greatly from one practice to another, because of the wide variation in costs associated with achieving a given dose limit for different practices.

[3] Various national and international authorities are involved in efforts to establish a generally applicable *de minimis* dose. The values currently under consideration are an annual dose equivalent in the range 0.001-0.1 mSv (0.1-10 mrem). We believe that a value above the upper end of this range would be too high to be reasonable for waste disposal, because of the existing limit on annual dose equivalent for this practice of 25 mrem (0.25 mSv).[17] A value toward the lower end of this range may be too low for practical application, because of the difficulties that would be encountered in measuring associated quantities of radionuclides.

[4] As a generally applicable *de minimis* dose for the general public, we recommend a principal limit on annual committed effective dose equivalent averaged over a lifetime of 0.01 mSv (1 mrem), with a subsidiary limit on committed effective dose equivalent in any year of 0.05 mSv (5 mrem). This proposal is based on the recommendation of the NCRP that a limit on *de minimis* dose be set at 1% of the limit on acceptable dose from all sources of exposure,[32] and that the limit on acceptable dose be set at an annual committed effective dose equivalent of 1 mSv (0.1 rem) for continuous exposures and 5 mSv (0.5 rem) for occasional exposures.[9,10] The proposed *de minimis* dose corresponds to a risk from continuous lifetime exposure of about 10^{-5}.

[5] The proposed *de minimis* dose presented above should be applied to the determination of exempt concentrations for waste disposal only for those radionuclides that are man-made. As noted by advisory groups of the IAEA,[36,37] it is illogical to apply a limit on annual dose equivalent of 0.01 mSv (1 mrem) to naturally occurring radionuclides which, in their undisturbed state, lead to doses much greater than the *de minimis* value. Thus, in particular, the establishment of a generally applicable *de minimis* dose would not be useful in establishing exempt concentrations of uranium in solid wastes.

[6] Estimates of doses that could be received by an inadvertent intruder at a waste disposal site can be obtained from standard methodologies that are based on appropriate definitions of scenarios for intruder activities and pathways for radiation exposure. However, the doses estimated by various investigators may vary by 1-2 orders of magnitude or more for a given radionuclide because of differences in the disposal site, exposure scenarios, environmental parameters for the various exposure pathways, and dosimetric data assumed in each case. Thus, unless site-specific data are obtained, estimated doses to an inadvertent intruder from radioactive waste disposal at any site probably will have large uncertainties for many radionuclides.

[7] Application of the dose assessment methodology for the Oak Ridge Reservation to the disposal of uranium-bearing wastes involved the important assumption that the concentration of uranium in waste leachate will be solubility limited. Thus, for uranium concentrations in the disposal facility greater than those that would result in solubility-limited leachate concentrations, the predicted uranium concentration in the leachate is a constant

independent of the concentration in the solid wastes. For trench disposal, the leachate then is assumed to be transported to an aquifer below the trench from which an inadvertent intruder obtains all drinking water. We assumed a solubility limit for uranium of 45 μmol/L and an equilibrium distribution coefficient for uranium in the soil/water system of 1 L/kg,[45] which correspond to a minimum concentration of ^{238}U in solid wastes in the trench that would result in solubility-limited leachate concentrations of about 5 $\mu Ci/m^3$. This concentration is about an order of magnitude less than the proposed exempt concentration of 30 pCi/g for depleted, natural, and enriched uranium from the Y-12 Plant.[51] We further assumed that the uranium concentrations in the aquifer would be reduced by a factor of 22 from those in the trench leachate, due to the effects of dilution of infiltrating water and retardation of uranium in the soil/water system.[45] Thus, the solubility-limited concentration of ^{238}U in the aquifer was assumed to be 1.6×10^{-4} μCi/L. For mixtures of uranium isotopes in the wastes, the activity concentration of each isotope in the trench leachate is determined by the fractional abundance by mass of each isotope in the solid wastes and their known specific activities. Thus, for example, the concentrations in μCi/L in the aquifer for the different isotopes in natural uranium would be 1.6×10^{-4} for ^{238}U and ^{234}U and 7.6×10^{-6} for ^{235}U. The factors for converting concentrations in the aquifer to annual committed effective dose equivalents for the uranium isotopes, in units of mrem/y per μCi/L, are as follows: 1.1×10^5 for ^{234}U, 9.8×10^4 for ^{235}U, and 9.9×10^4 for ^{238}U.

[8] The dose assessment methodology for the Oak Ridge Reservation also included several exposure pathways involving activity in the disposal facility itself; i.e., ingestion of vegetables grown in contaminated soil, ingestion of contaminated soil from the vegetable garden, external exposure to contaminated soil while working in the vegetable garden and living in a house constructed on top of the trench, and inhalation of suspended activity from contaminated soil while working in the vegetable garden and living in the house. For any radionuclide, the annual dose equivalent from these pathways is directly proportional to the concentration in the disposal facility at the time intrusion occurs. For the uranium isotopes in depleted, natural, or enriched uranium, the factors for converting concentrations in the disposal facility to annual committed effective dose equivalents, in units of mrem/y per $\mu Ci/m^3$, are as follows: 0.012 for ^{234}U, 0.14 for ^{235}U, and 0.035 for ^{238}U.

[9] Based on the results in the preceding two paragraphs, the following estimates of annual committed effective dose equivalents to an inadvertent intruder resulting from disposal of natural uranium at the proposed exempt concentration of 30 pCi/g (i.e., approximately 15 pCi/g each of ^{238}U and ^{234}U) are obtained: 18 mrem from ^{234}U, 0.7 mrem from ^{235}U, and 16 mrem from ^{238}U. Thus, the total annual committed effective dose equivalent from disposal of this concentration of natural uranium is about 35 mrem. Because of the solubility limit for uranium assumed in the calculations, the dose from each isotope for this concentration of uranium in the disposal trench is due almost entirely to the dose from the drinking water pathway and, thus, is essentially independent of concentration in the trench.

[10] As the concentration of uranium in the trench increases above the proposed exempt concentration of 30 pCi/g, the dose from the drinking water pathway is unchanged but the contribution from the other pathways increases in proportion to the concentration. Thus, for example, if a limit on concentrations of uranium that would be acceptable for disposal in a near-surface facility for low-level radioactive wastes were based on a limit on annual committed effective dose equivalent to an inadvertent intruder of 1 mSv (0.1 rem),[38] then about two-thirds of the total dose (i.e., 65 mrem) would result from direct intrusion into the waste and the concentration limit for natural uranium would be about 2.4×10^3 $\mu Ci/m^3$ (1.7×10^3 pCi/g), or nearly two orders of magnitude greater than the proposed exempt concentration of 30 pCi/g.

[11] A potentially important factor that usually is not taken into account in estimating doses from disposal of uranium is the long-term buildup of ^{226}Ra and short-lived daughter products from the decay of ^{234}U and ^{230}Th. If the uranium placed in the trench remains there for tens of thousands of years or more, then the buildup of ^{226}Ra and daughters can result in doses to an inadvertent intruder that are considerably higher than the doses from exposure to the uranium alone. For ^{226}Ra and daughters in secular equilibrium with ^{238}U and ^{234}U in the disposal facility and assuming no escape of ^{222}Rn from soil, we have estimated an annual committed effective dose equivalent of about 4 mrem/y per $\mu Ci/m^3$ of ^{238}U. Under these conditions, a uranium concentration of 30 pCi/g would result in an annual committed effective dose equivalent to an inadvertent intruder of about 160 mrem, a value that is considerably in excess of natural background levels in Oak Ridge. Thus, if

uranium is not removed from the trench by infiltration of water or other physical removal processes before significant buildup of ^{226}Ra and daughters occurs, then 30 pCi/g of uranium probably could not be regarded as an exempt concentration. In this case, the limit on exempt concentrations for depleted, natural, and enriched uranium probably should be reduced to 5 pCi/g, which is the EPA's cleanup standard for ^{226}Ra in soils.[15,19] In any event, a cost-benefit analysis probably would be required to justify an exemption level for uranium that is considerably above naturally occurring levels uranium in soils on the Oak Ridge Reservation. We would also emphasize, however, that any appreciable infiltration of water into the solid wastes, such as would be expected at a site with considerable rainfall, and an appreciable solubility of uranium in water probably will result in most of the uranium being removed from the disposal facility before a significant buildup of ^{226}Ra and daughter products occurs.

[12] Given the importance of transport in water for determining the dose to an inadvertent intruder from disposal of uranium and for determining the dose from the long-term buildup of ^{226}Ra and daughter products, a site-specific determination of the long-term behavior of uranium in the soil/water system clearly is quite important. Reliable information is needed on such factors as the infiltration rate of water through the trench, the leachability of uranium in the solid wastes, the solubility limit of uranium in water, the equilibrium distribution coefficient for uranium in the soil/water system, the water travel time to an underlying aquifer, and the retardation factor for transport of uranium in the soil/water system. In the absence of such information, any estimated doses from disposal of uranium are quite uncertain.

[13] The establishment of a limit on exempt concentrations of uranium in solid wastes, e.g., 30 pCi/g, probably requires a reliable technique for measuring such concentrations in order for the wastes to be acceptable for disposal in a sanitary landfill. The Y-12 Plant currently is developing a uranium assay system based on passive detection of high-energy photons emitted in the decay of a short-lived daughter product of ^{238}U. The information we have received on this system indicates that it does not yet provide a reliable means of assaying small quantities of uranium in large containers. The concentration inferred from the measurements varies greatly with the location of uranium within the source volume, and the detection of small quantities of ^{238}U is hindered by the ubiquitous background

from natural sources. The relationship between the measurements of this system and the proposed exempt concentration of 30 pCi/g for uranium thus does not appear to have been established satisfactorily.

[14] A number of suggestions for improving the gamma-ray measurement system being developed at the Y-12 Plant were considered. These involved using additional detectors and moving the source volume through the detection system during the measurements. These modifications should alleviate some of the geometrical problems arising from use of only one or two fixed detectors in conjunction with large stationary volumes of waste materials. We also suggested that the segregation of contaminated and uncontaminated wastes at the point of generation using portable photon detectors could be a cost-effective means of reducing the volume of wastes that would be sent to a radioactive waste-disposal facility.

[15] A promising alternative for assaying uranium wastes involves active pulsed thermal-neutron interrogation of the ^{235}U contained in depleted, natural, or enriched uranium. This technique has been used successfully in assaying very small quantities of fissile materials in TRU wastes. This method largely eliminates the problem of competing natural background radiation associated with the passive gamma-ray technique. The costs of a pulsed thermal-neutron system are somewhat higher than for the passive gamma-ray system, but the detection limits for uranium appear to be much lower and the accuracy quite acceptable even for large source volumes. Thus, this method may be considerably more satisfactory for verifying that concentrations of uranium in bulk solid wastes are below any proposed exemption level.

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APPENDIX A

DOSE ANALYSIS FOR AN INADVERTENT INTRUDER

This appendix presents a methodology for estimating annual dose equivalents to an inadvertent intruder at a near-surface disposal facility containing radioactive materials. The methodology assumes that all radionuclides may be transported in the environment according to the postulated scenarios.

Section A.1 presents the general equation for estimating annual committed effective dose equivalents to an inadvertent intruder. In Section A.2, the general equation is applied to the different exposure pathways that are assumed to occur. In Section A.3, tables of factors are developed for each exposure scenario that convert radionuclide concentrations in the various environmental media to annual committed effective dose equivalents. These factors can be used to derive limits on concentrations of radionuclides that can be placed in the disposal facility on the basis of a limit on annual dose equivalents to an inadvertent intruder, an assumed time delay between disposal and the occurrence of intruder exposures, and estimates of removal of radionuclides from the disposal facility prior to the exposures. The results of the intruder dose analysis are summarized in Section A.4.

A.1 General Equation for Radiation Dose

The general equation for estimating the annual dose equivalent to an individual at time t from radionuclide i and exposure pathway p is[1]

$$H_{ip}(t) = C_{ip}(t)U_pD_{ip} \ , \qquad \text{(A-1)}$$

where

- H = annual dose equivalent,
- C = radionuclide concentration in medium of exposure (air, water, soil, or foodstuffs),
- U = usage parameter (annual exposure time or annual intake of contaminated material), and
- D = dose conversion factor (annual dose equivalent per unit radionuclide concentration in environment for external exposure or committed dose equivalent per unit radionuclide intake for internal exposure).

We express H as the annual committed effective dose equivalent,[2] and the time dependence of the annual dose equivalent and the environmental

concentration C generally is suppressed in writing eq. (A-1) for each exposure pathway. The dose conversion factors for internal exposures via inhalation or ingestion are the same for all pathways leading to intakes by the given route. For external exposures, however, the dose conversion factors depend on the assumed distribution of sources in the environment and the location of the exposed individual relative to the source region.

A.2 Equations for Specific Exposure Pathways

In this section, eq. (A-1) is applied to the specific pathways for exposures of inadvertent intruders that are considered in this analysis. We assume an intruder-agriculture scenario in which the following exposure pathways occur:

- ingestion of contaminated drinking water;
- ingestion of vegetables grown in contaminated soil;
- ingestion of contaminated soil from the vegetable garden;
- ingestion of milk and meat from dairy and beef cattle that drink contaminated water;
- external exposure to contaminated soil; and
- inhalation of suspended activity from contaminated soil.

Sections A.2.1-A.2.6 present the model equations for estimating annual dose equivalents from each exposure pathway.

A.2.1 *Ingestion of contaminated drinking water*

The annual committed effective dose equivalent in rem per year from ingestion of radionuclide i in drinking water (w) is given by

$$H_{iw} = C_{iw} U_w D_i \ , \qquad \text{(A-2)}$$

where

C_{iw} = concentration of radionuclide i in drinking water (μCi/L),
U_w = annual consumption of drinking water (liters per year), and

D_i = dose conversion factor for ingestion of radionuclide i (rem per μCi ingested).

We assume that all drinking water consumed by an intruder comes from a contaminated source on the disposal site.

A.2.2 *Ingestion of contaminated vegetables*

The annual committed effective dose equivalent in rem per year from ingestion of radionuclide i in vegetables (v) is given by

$$H_{iv} = C_{iv} U_v D_i \ , \tag{A-3}$$

where

C_{iv} = concentration of radionuclide i in vegetables (μCi/kg),
U_v = annual consumption of vegetables (kg per year), and
D_i = dose conversion factor for ingestion of radionuclide i (rem per μCi ingested).

We assume an exposure scenario in which an intruder mixes contaminated soil from the disposal facility with native soil in the vegetable garden, and the radionuclides are transferred to the vegetation via root uptake. Radionuclide concentrations in vegetables then are given by

$$\begin{aligned} C_{iv} &= B_{iv} C_{is} / \rho_s \\ &= B_{iv} f_{is} C_{it} / \rho_s \ , \end{aligned} \tag{A-4}$$

where

B_{iv} = plant-to-soil concentration ratio for radionuclide i (μCi/kg wet weight in vegetation per μCi/kg dry weight in soil),
C_{is} = concentration of radionuclide i in soil in vegetable garden (μCi/m^3),
ρ_s = density of soil (kg/m^3),
C_{it} = concentration of radionuclide i in disposal facility (μCi/m^3), and
f_{is} = dilution factor for mixing of radionuclide i from disposal facility into soil in vegetable garden.

We assume that all vegetables consumed by an intruder are grown in the contaminated garden.

A.2.3 *Ingestion of contaminated soil*

The annual committed effective dose equivalent in rem per year from direct ingestion of radionuclide i in contaminated soil (s) is given by

$$H_{is} = C_{is}U_sD_i \ , \tag{A-5}$$

where

C_{is} = concentration of radionuclide i in soil in vegetable garden (μCi/kg),
U_s = annual consumption of contaminated soil (kg per year), and
D_i = dose conversion factor for ingestion of radionuclide i (rem per μCi ingested).

As with the scenario for ingestion of contaminated vegetables, we assume that an intruder mixes contaminated soil from the disposal facility with native soil in the vegetable garden, and that a quantity of contaminated soil is ingested in conjunction with vegetable intakes. As in eq. (A-4), radionuclide concentrations in contaminated soil in the vegetable garden are given by

$$C_{is} = f_{is}C_{it}/\rho_s \ , \tag{A-6}$$

where

C_{it} = concentration of radionuclide i in disposal facility (μCi/m^3),
f_{is} = dilution factor for mixing of radionuclide i from disposal facility into soil in vegetable garden, and
ρ_s = density of soil (kg/m^3).

A.2.4 *Ingestion of contaminated milk and meat*

The annual committed effective dose equivalents in rem per year from ingestion of radionuclide i in milk (m) and meat (f) are given by

$$H_{im} = C_{im}U_mD_i \ , \tag{A-7}$$

$$H_{if} = C_{if}U_fD_i \ , \tag{A-8}$$

respectively, where

C_{im} = concentration of radionuclide i in milk (μCi/L),
C_{if} = concentration of radionuclide i in meat (μCi/kg),
U_m = annual consumption of milk (liters per year),
U_f = annual consumption of meat (kg per year), and

D_i = dose conversion factor for ingestion of radionuclide i (rem per μCi ingested).

We assume an exposure scenario in which dairy and beef cattle drink contaminated water from the disposal site. Radionuclide concentrations in milk and meat then are given by

$$C_{im} = C_{iw}Q_{wm}F_{im} \ , \tag{A-9}$$

$$C_{if} = C_{iw}Q_{wf}F_{if} \ , \tag{A-10}$$

respectively, where

C_{iw} = concentration of radionuclide i in drinking water (μCi/L),
Q_{wm} = daily consumption of drinking water by dairy cattle (liters per day),
Q_{wf} = daily consumption of drinking water by beef cattle (liters per day),
F_{im} = ratio of equilibrium concentration of radionuclide i in milk to daily intake by dairy cattle (μCi/L in milk per μCi/d intake), and
F_{if} = ratio of equilibrium concentration of radionuclide i in meat to daily intake by beef cattle (μCi/kg in meat per μCi/d intake).

We assume that all milk and meat consumed by an intruder come from contaminated dairy and beef cattle and that the dairy and beef cattle obtain all drinking water from the same contaminated source on the disposal site that supplies drinking water for the intruder.

A.2.5 *External exposure to contaminated soil*

We assume two scenarios for external exposure of an inadvertent intruder. In the first, an intruder spends some fraction of the time working in the vegetable garden that is contaminated with soil from the disposal facility. In the second, an intruder spends some fraction of the time indoors in a house that is constructed immediately on top of the disposal facility.

For external exposure (e) to contaminated soil in the vegetable garden, the annual effective dose equivalent in rem per year from radionuclide i is given by

$$H_{ie} = C_{is}U_sD_{is} \ , \tag{A-11}$$

where

C_{is} = concentration of radionuclide i in soil in vegetable garden ($\mu Ci/m^3$),

U_s = fraction of the year during which external exposure to contaminated soil in vegetable garden occurs, and

D_{is} = dose conversion factor for external exposure to radionuclide i in soil (rem/y per $\mu Ci/m^3$).

As in Sections A.2.2 and A.2.3, the concentration of radionuclide i in soil in the vegetable garden is given by

$$C_{is} = f_{is}C_{it} \ , \qquad \text{(A-12)}$$

where

C_{it} = concentration of radionuclide i in disposal facility ($\mu Ci/m^3$), and

f_{is} = dilution factor for mixing of radionuclide i from disposal facility into soil in vegetable garden.

The dose conversion factors, D_{is}, are based on the assumption that the activity is uniformly distributed within a slab of soil of depth 15 cm, i.e., within the depth of the plowed layer, and that the exposed individual is standing on the ground surface.

For external exposure to the contents of the disposal facility during indoor residence, the annual effective dose equivalent in rem per year from radionuclide i is given by

$$H_{ie} = C_{it}U_tD_{it}S_i \ , \qquad \text{(A-13)}$$

where

C_{it} = concentration of radionuclide i in disposal facility ($\mu Ci/m^3$),

U_t = fraction of the year during which external exposure to disposal facility during indoor residence occurs,

D_{it} = dose conversion factor for external exposure to radionuclide i in disposal facility (rem/y per $\mu Ci/m^3$), and

S_i = shielding factor for radionuclide i during indoor residence.

The dose conversion factors, D_{it}, are based on the assumption that the activity is uniformly distributed in a semi-infinite slab source beginning at the ground surface, and that the exposed individual is standing at ground level. We note that a depth of the source region below ground greater than about 1 m is effectively semi-infinite for the purposes of estimating external dose above ground.[3]

A.2.6 *Inhalation of suspended activity from contaminated soil*

We assume two scenarios for inhalation exposure that are similar to those for external exposure described in Section A.2.5. In the first, an intruder spends some fraction of the time working in the vegetable garden that is contaminated with soil from the disposal facility. In the second, an intruder spends some fraction of the time indoors in a house that is constructed immediately on top of the disposal facility.

The annual committed effective dose equivalent in rem per year from inhalation of radionuclide i in air (a) is given by

$$H_{ia} = C_{ia} f_a U_a D_i \ , \qquad \text{(A-14)}$$

where

C_{ia} = concentration of radionuclide i in air (μCi/m^3),
f_a = fraction of the year during which inhalation exposure occurs,
U_a = annual air intake (m^3 per year), and
D_i = dose conversion factor for inhalation of radionuclide i (rem per μCi inhaled).

We estimate concentrations of suspended material in air using a mass-loading approach,[4] which is based on observations of airborne concentrations of naturally occurring materials, such as uranium and thorium, relative to their concentrations in surface soils. In this model, the airborne concentration of radionuclide i is given by

$$C_{ia} = C_{is} L_a / \rho_s \ , \qquad \text{(A-15)}$$

where

C_{is} = concentration of radionuclide i in soil (μCi/m^3),
L_a = mass loading of soil in the atmosphere (kg/m^3), and
ρ_s = density of soil (kg/m^3).

For inhalation of contaminated soil suspended from the vegetable garden, the concentration of radionuclide i in soil again is given by

$$C_{is} = f_{is} C_{it} \ , \qquad \text{(A-16)}$$

where

C_{it} = concentration of radionuclide i in disposal facility (μCi/m^3), and
f_{is} = dilution factor for mixing of radionuclide i from disposal facility into soil in vegetable garden.

For inhalation exposures during indoor residence, the airborne concentration of radionuclide i is given by

$$C_{ia} = S_i C_{it} L_a / \rho_s \ , \qquad \text{(A-17)}$$

where

S_i = ratio of indoor to outdoor air concentration for radionuclide i,
C_{it} = concentration of radionuclide i in disposal facility ($\mu Ci/m^3$),
L_a = mass loading of soil in the atmosphere (kg/m^3), and
ρ_s = density of soil (kg/m^3).

A.3 Dose Calculations for Specific Exposure Pathways

This section presents the data used in estimating annual committed effective dose equivalents for the different exposure pathways assumed in this analysis. These data then are used to produce tables of annual committed effective dose equivalents per unit radionuclide concentration for each pathway; for external exposure pathways, the committed dose is the same as the dose received.

The dose analysis assumes that the probability of occurrence for each exposure scenario is unity, and that inadvertent intrusion may occur at any time after loss of institutional controls over the facility. Because these assumptions probably result in overestimates of risk to inadvertent intruders, we believe it is most appropriate to choose reasonable average values for parameters that describe transport of radionuclides through terrestrial foodchains and the annual intakes and exposure times for an inadvertent intruder, rather than maximum possible values. That is, we believe it is unreasonable to assume not only that an inadvertent intruder will be exposed with unit probability at any time according to each of the pathway scenarios, but also that an intruder will experience extreme intakes and exposure times. This approach is consistent with the intent of the International Commission on Radiological Protection (ICRP) that limits on radiation dose should apply to average individuals within the critical group of maximally exposed individuals, rather than the single individual who might receive the highest dose.[2]

A.3.1 *Radionuclides of potential importance to intruder exposures*

The radionuclides that are assumed to be of potential importance to exposures of inadvertent intruders are listed in Table A-1. These radionuclides are expected to be contained in waste streams generated at the three plants on the Oak Ridge Reservation. Since institutional controls are expected to prevent intruder exposures for at least 100 years

Table A-1. Radionuclides of potential importance to exposures of inadvertent intruders

Nuclide[a]	Half-life[b]	Nuclide[a]	Half-life[b]
H-3	12.28 y	U-232	72 y
Be-10	1.6E6 y	Th-228	1.9132 y
C-14	5730 y	Ra-224	3.62 d
Co-60	5.271 y	Pb-212	10.643 h
Ni-63	100.1 y	Bi-212	60.55 m
Sr-90	28.6 y	Tl-208	3.053 m
Y-90	64.1 h	U-233	1.592E5 y
Zr-93	1.53E6 y	U-234	2.445E5 y
Nb-93m	14.6 y	U-235	7.038E8 y
Tc-99	2.13E5 y	Th-231	25.52 h
Cd-113m	13.7 y	U-236	2.3415E7 y
Sn-121m	55 y	U-238	4.468E9 y
Cs-137	30.17 y	Th-234	24.10 d
Sm-151	90 y	Pa-234m	1.17 m
Eu-152	13.6 y	Pa-234	6.70 h
Eu-154	8.8 y	Np-237	2.14E6 y
Eu-155	4.96 y	Pa-233	27.0 d
Ra-226	1600 y	Pu-238	87.75 y
Pb-214	26.8 m	Pu-239	24131 y
Bi-214	19.9 m	Pu-241	14.4 y
Pb-210	22.26 y	Pu-242	3.758E5 y
Po-210	138.378 d	Am-241	432.2 y
Th-232	1.405E10 y	Am-243	7.38E3 y
Ra-228	5.75 y	Np-239	2.355 d
Ac-228	6.13 h	Cm-244	18.11 y
Th-228	1.9132 y	Cf-249	350.6 y
Ra-224	3.62 d		
Pb-212	10.643 h		
Bi-212	60.55 m		
Tl-208	3.053 m		

[a]Indented entries are radiologically significant daughter products.

[b]Values from ref. 5.

after disposal, a radionuclide is listed in Table A-1 only if its half-life is sufficiently long that the inventory of the radionuclide would not be depleted within 100 years due to radioactive decay. Table A-1 also lists any relatively short-lived radioactive daughter products that could contribute significantly to either external or internal exposures following the period of institutional controls. In most cases, it is reasonable to assume that the short-lived daughters are in secular equilibrium with the long-lived parent for the purpose of estimating environmental concentrations of the daughters, and the tables of annual dose equivalents per unit concentration of radionuclides in the environment presented in this report are based on this assumption. Thus, the annual dose equivalents from short-lived daughter products assume unit concentrations of the longer-lived parent and the known decay branching fractions.[5] It is important to note, however, that ^{241}Am also is produced in the decay of ^{241}Pu, and this production should be taken into account in estimating concentrations of ^{241}Am over time.

A.3.2 *Dose conversion factors for inhalation and ingestion*

The dose conversion factors for inhalation and ingestion of radionuclides are given in Tables A-2 and A-3, respectively. These data represent 50-year committed effective dose equivalents in rem from an acute intake of 1 μCi by each intake route and are calculated using models and data bases developed by the ICRP.[6] Table A-2 gives the respiratory clearance class[6] assumed for each inhaled radionuclide, and Table A-3 gives the GI-tract uptake fraction assumed for each ingested radionuclide. Dose conversion factors are listed for radioactive daughter products only if the daughter could contribute significantly to internal dose from intakes of a parent and its daughters in secular equilibrium.

A.3.3 *Annual doses from ingestion of contaminated drinking water*

The annual committed effective dose equivalents per unit concentration of radionuclides in drinking water are given in Table A-4. These data were obtained from eq. (A-2) and assume an annual consumption of contaminated water by an average adult of[7]

U_W = 370 liters per year (1 liter per day),

and the dose conversion factors in Table A-3.

Table A-2. Committed effective dose equivalents from inhalation of radionuclides

Nuclide[a]	Clearance class[b]	Rem per μCi inhaled[c]	Nuclide[a]	Clearance class[b]	Rem per μCi inhaled[c]
H-3		6.30E-5	Th-232	Y	1.15E3
Be-10	W	3.55E-1	Th-228	Y	3.42E2
C-14		2.36E-5	U-232[d]	Y	6.59E2
Co-60	Y	2.19E-1	U-233	Y	1.35E2
Ni-63	W	2.30E-3	U-234	Y	1.33E2
Sr-90	Y	1.30	U-235	Y	1.23E2
Zr-93	Y	7.40E-2	U-236	Y	1.26E2
Nb-93m	Y	2.93E-2	U-238	Y	1.18E2
Tc-99	W	8.32E-3	Np-237	W	4.80E2
Cd-113m	D	1.53	Pu-238	W	4.63E2
Sn-121m	W	1.15E-2	Pu-239	W	5.17E2
Cs-137	D	3.20E-2	Pu-241	W	1.04E1
Sm-151	W	3.00E-2	Pu-242	W	4.92E2
Eu-152	W	2.21E-1	Am-241	W	5.31E2
Eu-154	W	2.86E-1	Am-243	W	5.30E2
Eu-155	W	4.15E-2	Cm-244	W	2.82E2
Ra-226	W	8.58	Cf-249	W	5.62E2
Pb-210	D	1.36E1			
Po-210	D	9.41			

[a]Indented entries are radiologically significant daughter products.

[b]Clearance from respiratory passages for radionuclides in particulate form in a matter of days (D), weeks (W), or years (Y).

[c]50-year dose commitment for particle size of 1 μm.

[d]Entry for Th-228 daughter product is given under Th-232 above.

Table A-3. Committed effective dose equivalents from ingestion of radionuclides

Nuclide[a]	f_1[b]	Rem per μCi ingested[c]	Nuclide[a]	f_1[b]	Rem per μCi ingested[c]
H-3	1.0	6.30E-5	Th-232	0.0002	2.73
Be-10	0.005	4.67E-3	Ra-228	0.2	1.44
C-14	0.95	1.54E-3	Th-228	0.0002	3.97E-1
Co-60	0.3	2.69E-2	Ra-224	0.2	3.66E-1
Ni-63	0.05	5.76E-4	Pb-212	0.2	4.55E-2
Sr-90	0.3	1.43E-1	U-232[d]	0.05	1.31
Y-90	0.0001	1.08E-2	U-233	0.05	2.89E-1
Zr-93	0.002	1.66E-3	U-234	0.05	2.84E-1
Nb-93m	0.01	5.23E-4	U-235	0.05	2.66E-1
Tc-99	0.8	1.46E-3	U-236	0.05	2.69E-1
Cd-113m	0.05	1.61E-1	U-238	0.05	2.55E-1
Sn-121m	0.02	1.55E-3	Th-234	0.0002	1.37E-2
Cs-137	1.0	5.02E-2	Np-237	0.001	4.00
Sm-151	0.0003	3.88E-4	Pu-238	0.001	3.81
Eu-152	0.001	6.50E-3	Pu-239	0.001	4.27
Eu-154	0.001	9.56E-3	Pu-241	0.001	8.66E-2
Eu-155	0.001	1.53E-3	Pu-242	0.001	4.06
Ra-226	0.2	1.33	Am-241	0.001	4.37
Pb-210	0.2	5.37	Am-243	0.001	4.34
Po-210	0.1	1.90	Cm-244	0.001	2.30
			Cf-249	0.0005	2.33

[a]Indented entries are radiologically significant daughter products.

[b]Fraction of ingested radionuclide absorbed into blood from the GI tract.

[c]50-year dose commitment.

[d]Entries for Th-228, Ra-224, and Pb-212 daughter products are given under Th-232 above.

Table A-4. Annual committed effective dose equivalents per unit concentration of radionuclides in drinking water[a]

Nuclide[b]	Annual dose (rem/y per μCi/L)	Nuclide[b]	Annual dose (rem/y per μCi/L)
H-3	2.3E-2	Th-232	1.0E3
Be-10	1.7	Ra-228	5.3E2
C-14	5.7E-1	Th-228	1.5E2
Co-60	1.0E1	Ra-224	1.4E2
Ni-63	2.1E-1	Pb-212	1.7E1
Sr-90	5.3E1	U-232[c]	4.8E2
Y-90	4.0	U-233	1.1E2
Zr-93	6.1E-1	U-234	1.1E2
Nb-93m	1.9E-1	U-235	9.8E1
Tc-99	5.4E-1	U-236	1.0E2
Cd-113m	6.0E1	U-238	9.4E1
Sn-121m	5.7E-1	Th-234	5.1
Cs-137	1.9E1	Np-237	1.5E3
Sm-151	1.4E-1	Pu-238	1.4E3
Eu-152	2.4	Pu-239	1.6E3
Eu-154	3.5	Pu-241	3.2E1
Eu-155	5.7E-1	Pu-242	1.5E3
Ra-226	5.7E-1	Am-241	1.6E3
Pb-210	2.0E3	Am-243	1.6E3
Po-210	7.0E2	Cm-244	8.5E2
		Cf-249	8.6E2

[a]Assumptions for calculations are described in Sections A.2.1 and A.3.3.

[b]Indented entries are radiologically significant daughter products; doses from daughters assume secular equilibrium between parent and daughters.

[c]Entries for Th-228, Ra-224, and Pb-212 daughter products are given under Th-232 above.

A.3.4 *Annual doses from ingestion of contaminated vegetables*

From eqs. (A-3) and (A-4), the annual committed effective dose equivalents from ingestion of contaminated vegetables depend on the plant-to-soil concentration ratios, B_{iv}, for each radionuclide. The concentration ratios assumed in this analysis are given in Table A-5. The values from Table 4.1 of ref. 8 are the reported arithmetic means. The values from Tables 5.16 and 5.18 of ref. 10 are the reported arithmetic means for different edible foods but exclude the values that represent gross plant-to-soil concentration ratios including external contamination of the plants by deposited and resuspended materials; for the exposure scenario assumed in this analysis, we are interested only in direct transfer from soil to plants via root uptake. The value from Table 10 of ref. 9 is an estimated arithmetic mean for the reported range of values. The values for Sn and Cf are estimated from the values for other elements that are chemical analogs. The value for C is obtained from the value in Table E-1 of ref. 7 and the assumption that 1% of the carbon in plants comes from root uptake from soil; the remaining 99% is assumed to come from photosynthesis of atmospheric carbon.

The adopted plant-to-soil concentration ratios in Table A-5 are subject to large uncertainties that may approach 1-2 orders of magnitude for some elements, because reported values often show large variations depending on the particular food crop and properties of the soil. In selecting the values in Table A-5, our objective was to chose a reasonable mean value from the available data rather than the largest value for any type of food or soil condition. The resulting doses from consumption of contaminated vegetables still should be somewhat conservative, however, because the mean value of B_{iv} generally lies toward the upper end of the range of reported values and all vegetables consumed by an inadvertent intruder are assumed to be grown in the contaminated garden.

The annual committed effective dose equivalents for the vegetable pathway per unit concentration of radionuclides in the disposal facility at the time intrusion occurs are given in Table A-6. These data were obtained from eqs. (A-3) and (A-4) and assume the plant-to-soil concentration ratios in Table A-5, a dilution factor for mixing of all radionuclides from the disposal facility into the native soil of the vegetable garden of[11]

$$f_{is} = 0.2,$$

a soil density of[12]

$$\rho_s = 1.4 \times 10^3 \text{ kg/m}^3,$$

an annual consumption of contaminated vegetables by an average adult

Table A-5. Elemental plant-to-soil concentration ratios in vegetables

Element	B_{iv}[a]	Source
H	4.8	Ref. 7, Table E-1
Be	3.7E-1	b
C	5.5E-2	c
Co	3.0E-2	Ref. 8, Table 4.1
Ni	3.3E-2	Ref. 8, Table 4.1
Sr	1.6E-1	Ref. 8, Table 4.1
Y	2.4E-2	Ref. 8, Section 5.3
Zr	3.2E-3	Ref. 8, Table 4.1
Nb	5.0E-2	Ref. 8, Section 5.4
Tc	5.0	Ref. 8, Section 5.5
Cd	1.6E-1	d
Sn	3.2E-3	e
Cs	9.5E-3	Ref. 8, Table 4.1
Sm	4.8E-3	Ref. 8, Section 5.3
Eu	4.8E-3	Ref. 8, Section 5.3
Pb	1.0E-2	From ref. 9, Table 10
Po	2.4E-4	From ref. 10, Tables 5.16 and 5.18
Ra	2.6E-2	From ref. 10, Tables 5.16 and 5.18
Th	5.0E-4	From ref. 10, Tables 5.16 and 5.18
U	1.7E-4	From ref. 10, Tables 5.16 and 5.18
Np	7.9E-2	Ref. 8, Table 4.1
Pu	8.0E-5	From ref. 10, Tables 5.16 and 5.18
Am	2.2E-5	From ref. 10, Tables 5.16 and 5.18
Cm	1.5E-5	From ref. 10, Tables 5.16 and 5.18
Cf	8.0E-5	f

[a]μCi/kg wet weight of vegetation per μCi/kg dry weight of soil.

[b]Value is assumed to be the same as for Zn.

[c]Value is assumed to be 1% of the value given in ref. 7, Table E-1.

[d]Value is assumed to be the same as for Sr.

[e]Value is assumed to be the same as for Zr.

Table A-6. Annual committed effective dose equivalents from vegetable pathway per unit concentration of radionuclides in disposal facility at time intrusion occurs[a]

Nuclide[b]	Annual dose (rem/y per $\mu Ci/m^3$)	Nuclide[b]	Annual dose (rem/y per $\mu Ci/m^3$)
H-3	3.9E-6	Th-232	1.8E-5
Be-10	2.2E-5	Ra-228	4.8E-4
C-14	1.1E-6	Th-228	2.6E-6
Co-60	1.0E-5	Ra-224	1.2E-4
Ni-63	2.4E-7	Pb-212	5.9E-6
Sr-90	2.9E-4	U-232[c]	2.9E-6
Y-90	3.3E-6	U-233	6.3E-7
Zr-93	6.8E-8	U-234	6.2E-7
Nb-93m	3.4E-7	U-235	5.8E-7
Tc-99	9.4E-5	U-236	5.9E-7
Cd-113m	3.3E-4	U-238	5.6E-7
Sn-121m	6.4E-8	Th-234	8.8E-8
Cs-137	6.1E-6	Np-237	4.1E-3
Sm-151	2.4E-8	Pu-238	3.9E-6
Eu-152	4.0E-7	Pu-239	4.4E-6
Eu-154	5.9E-7	Pu-241	8.9E-8
Eu-155	9.4E-8	Pu-242	4.2E-6
Ra-226	4.4E-4	Am-241	1.2E-6
Pb-210	6.9E-4	Am-243	1.2E-6
		Cm-244	4.4E-7
		Cf-249	2.4E-6

[a]Assumptions for calculations are described in Sections A.2.2 and A.3.4.

[b]Indented entries are radiologically significant daughter products; doses for daughters assume secular equilibrium between parent and daughters.

[c]Entries for Th-228, Ra-224, and Pb-212 daughter products are given under Th-232 above.

of[7,13]

U_v = 90 kg per year,

and the dose conversion factors in Table A-3.

A.3.5 *Annual doses from ingestion of contaminated soil*

The annual committed effective dose equivalents from direct ingestion of contaminated soil from the vegetable garden per unit concentration of radionuclides in the disposal facility are given in Table A-7. These data were obtained from eqs. (A-5) and (A-6) and assume a dilution factor for mixing of all radionuclides from the disposal facility into the native soil of the vegetable garden of[11]

f_{is} = 0.2,

a soil density of[12]

$\rho_s = 1.4 \times 10^3$ kg/m^3,

an annual consumption of contaminated soil from the vegetable garden by an average adult of[14]

U_s = 0.037 kg per year (0.1 g per day),

and the dose conversion factors in Table A-3.

A.3.6 *Annual doses from ingestion of contaminated milk and meat*

From eqs. (A-7)-(A-10), the annual committed effective dose equivalents from ingestion of contaminated milk and meat depend on the transfer coefficients of radionuclides from intake to milk for dairy cattle, F_{im}, and from intake to meat for beef cattle, F_{if}, respectively. The values of the milk and meat transfer coefficients assumed in this analysis are given in Tables A-8 and A-9, respectively. The adopted values are means of the reported data.

The annual committed effective dose equivalents for the milk pathway per unit concentration of radionuclides in water consumed by dairy cattle are given in Table A-10. These data were obtained from eqs. (A-7) and (A-9) and assume the milk transfer coefficients in Table A-8, a daily consumption of contaminated water by dairy cattle of[7]

Q_{wm} = 60 liters per day,

an annual consumption of contaminated milk by an average adult of[7,13]

Table A-7. Annual committed effective dose equivalents from soil ingestion pathway per unit concentration of radionuclides in disposal facility at time intrusion occurs[a]

Nuclide[b]	Annual dose (rem/y per $\mu Ci/m^3$)	Nuclide[b]	Annual dose (rem/y per $\mu Ci/m^3$)
H-3	3.3E-10	Th-232	1.4E-5
Be-10	2.5E-8	Ra-228	7.6E-6
C-14	8.1E-9	Th-228	2.1E-6
Co-60	1.4E-7	Ra-224	1.9E-6
Ni-63	3.0E-9	Pb-212	2.4E-7
Sr-90	7.6E-7	U-232[c]	6.9E-6
Y-90	5.7E-8	U-233	1.5E-6
Zr-93	8.8E-9	U-234	1.5E-6
Nb-93m	2.8E-9	U-235	1.4E-6
Tc-99	7.7E-9	U-236	1.4E-6
Cd-113m	8.5E-7	U-238	1.3E-6
Sn-121m	8.2E-9	Th-234	7.2E-8
Cs-137	2.7E-7	Np-237	2.1E-5
Sm-151	2.1E-9	Pu-238	2.0E-5
Eu-152	3.4E-8	Pu-239	2.3E-5
Eu-154	5.1E-8	Pu-241	4.6E-7
Eu-155	8.1E-9	Pu-242	2.1E-5
Ra-226	7.0E-6	Am-241	2.3E-5
Pb-210	2.8E-5	Am-243	2.3E-5
Po-210	1.0E-5	Cm-244	1.2E-5
		Cf-249	1.2E-5

[a]Assumptions for calculations are described in Sections A.2.3 and A.3.5.

[b]Indented entries are radiologically significant daughter products; doses for daughters assume secular equilibrium between parent and daughters.

[c]Entries for Th-228, Ra-224, and Pb-212 daughter products are given under Th-232 above.

Table A-8. Elemental intake-to-milk transfer coefficients for dairy cattle

Element	$F_{im}(d/L)$[a]	Element	$F_{im}(d/L)$[a]
H	1.4E-2	Sm	6.0E-5[c]
Be	9.1E-7	Eu	6.0E-5[c]
C	1.5E-2	Pb	2.6E-4
Co	2.9E-3	Po	3.4E-4
Ni	1.0E-3	Ra	4.0E-4
Sr	1.4E-3	Th	5.0E-6[b]
Y	2.0E-5[b]	U	3.7E-4
Zr	3.0E-5	Np	5.0E-6[b]
Nb	2.0E-2[b]	Pu	1.0E-7
Tc	9.9E-3[b]	Am	4.1E-7
Cd	1.5E-3	Cm	2.0E-5[b]
Sn	1.2E-3	Cf	2.0E-5[b]
Cs	7.1E-3		

[a]Values from Table 4 of ref. 9, unless otherwise noted.

[b]Value from Table 7 of ref. 15.

[c]Value is assumed to be the same as for Ce in Table 4 of ref. 9.

Table A-9. Elemental intake-to-meat transfer coefficients for beef cattle

Element	F_{if}(d/kg)	Source
H	1.2E-2	Ref. 7, Table E-1
Be	1.8E-2	a
C	3.1E-2	Ref. 7, Table E-1
Co	9.7E-3	Ref. 9, Table 7
Ni	2.0E-3	Ref. 9, Table 7
Sr	3.0E-4	Ref. 9, Table 5
Y	1.0E-3	Ref. 16, Table 7.1
Zr	2.1E-2	Ref. 9, Table 7
Nb	2.5E-1	Ref. 9, Table 7
Tc	1.0E-3	Ref. 16, Section 8
Cd	3.5E-4	Ref. 16, Table 7.1
Sn	2.1E-2	b
Cs	2.0E-2	Ref. 9, Table 5
Sm	7.5E-4	c
Eu	7.5E-4	c
Pb	1.0E-3	Ref. 9, Table 8
Po	4.0E-3	Ref. 9, Table 8
Ra	5.0E-4	Ref. 9, Table 8
Th	2.0E-4	Ref. 10, Table 5.37
U	3.4E-4	Ref. 10, Table 5.37
Np	1.0E-6	d
Pu	1.0E-6	Ref. 9, Table 5
Am	3.6E-6	Ref. 9, Table 6
Cm	3.6E-6	Ref. 10, Table 5.37
Cf	3.6E-6	Ref. 10, Table 5.37

[a]Value is assumed to be the same as for Mg from Table 7.1 of ref. 16.

[b]Value is assumed to be the same as for Zr.

[c]Value is assumed to be the same as for Ce in Table 5 of ref. 9.

[d]Value is assumed to be the same as for Pu.

Table A-10. Annual committed effective dose equivalents from milk pathway per unit concentration of radionuclides in water[a]

Nuclide[b]	Annual dose (rem/y per μCi/L)	Nuclide[b]	Annual dose (rem/y per μCi/L)
H-3	5.8E-3	Th-232	9.0E-2
Be-10	2.8E-5	Ra-228	3.8
C-14	1.5E-1	Ra-224	9.7E-1
Co-60	5.1E-1	U-232[c]	3.2
Ni-63	3.8E-3	U-233	7.1E-1
Sr-90	1.3	U-234	6.9E-1
Zr-93	3.3E-4	U-235	6.5E-1
Nb-93m	6.9E-2	U-236	6.6E-1
Tc-99	9.5E-2	U-238	6.2E-1
Cd-113m	1.6	Np-237	1.3E-1
Sn-121m	1.2E-2	Pu-238	2.5E-3
Cs-137	2.4	Pu-239	2.8E-3
Sm-151	1.5E-4	Pu-241	5.7E-5
Eu-152	2.6E-3	Pu-242	2.7E-3
Eu-154	3.8E-3	Am-241	1.2E-2
Eu-155	6.1E-4	Am-243	1.2E-2
Ra-226	3.5	Cm-244	3.0E-1
Pb-210	9.2	Cf-249	3.1E-1
Po-210	4.3		

[a]Assumptions for calculations are described in Sections A.2.4 and A.3.6.

[b]Indented entries are radiologically significant daughter products; doses for daughters assume secular equilibrium between parent and daughters.

[c]Entry for Ra-224 daughter product is given under Th-232 above.

U_m = 110 liters per year,

and the dose conversion factors in Table A-3.

The annual committed effective dose equivalents for the meat pathway per unit concentration of radionuclides in water consumed by beef cattle are given in Table A-11. These data were obtained from eqs. (A-8) and (A-10) and assume the meat transfer coefficients in Table A-9, a daily consumption of contaminated water by beef cattle of[7]

Q_{wf} = 50 liters per day,

an annual consumption of contaminated meat by an average adult of[7,13]

U_f = 90 kg per year,

and the dose conversion factors in Table A-3.

A.3.7 *Annual doses from external exposure to contaminated soil*

The annual effective dose equivalents from external exposure to photon-emitting radionuclides in soil depend on the assumed vertical distribution of the sources in soil. As described in Section A.2.5, two exposure scenarios with different source distributions were assumed: (1) a slab source with upper boundary at the ground surface and thickness of 15 cm for exposures while working in the vegetable garden, and (2) a slab source with upper boundary at the ground surface and a thickness that is effectively infinite for exposures during indoor residence. In both cases, the radionuclides are assumed to be uniformly distributed throughout the source region.

The annual effective dose equivalents for the two source distributions per unit concentration of photon-emitting radionuclides in soil for an individual standing on the ground surface are given in Tables A-12 and A-13. These results assume that the ratio of effective dose equivalent to absorbed dose in air for a given radionuclide is the same for sources in soil as for immersion in a semi-infinite atmospheric cloud. The absorbed dose in air for slab sources in soil is based on calculations for monoenergetic sources[3] and the known photon spectrum for each radionuclide.[5] The ratio of effective dose equivalent to absorbed dose in air for immersion in a semi-infinite atmospheric cloud is obtained from available data,[17-19] and the va .e is about 0.65-0.70 for most radionuclides.

The annual effective dose equivalents from external exposure per unit concentration of radionuclides in the disposal facility for an individual working in the contaminated vegetable garden are given in Table A-14.

Table A-11. Annual committed effective dose equivalents from meat pathway per unit concentration of radionuclides in water[a]

Nuclide[b]	Annual dose (rem/y per μCi/L)	Nuclide[b]	Annual dose (rem/y per μCi/L)
H-3	3.4E-3	Th-232	2.5
Be-10	3.8E-1	Ra-228	3.2
C-14	2.1E-1	Th-228	3.6E-1
Co-60	1.2	Ra-224	8.2E-1
Ni-63	5.2E-3	Pb-212	2.0E-1
Sr-90	1.9E-1	U-232[c]	2.0
Y-90	4.9E-2	U-233	4.4E-1
Zr-93	1.6E-1	U-234	4.3E-1
Nb-93m	5.9E-1	U-235	4.1E-1
Tc-99	6.6E-3	U-236	4.1E-1
Cd-113m	2.5E-1	U-238	3.9E-1
Sn-121m	1.5E-1	Th-234	1.2E-2
Cs-137	4.5	Np-237	1.8E-2
Sm-151	1.3E-3	Pu-238	1.7E-2
Eu-152	2.2E-2	Pu-239	1.9E-2
Eu-154	3.2E-2	Pu-241	3.9E-4
Eu-155	5.2E-3	Pu-242	1.8E-2
Ra-226	3.0	Am-241	7.1E-2
Pb-210	2.4E1	Am-243	7.0E-2
Po-120	3.4E1	Cm-244	3.7E-2
		Cf-249	3.8E-2

[a]Assumptions for calculations are described in Sections A.2.4 and A.3.6.

[b]Indented entries are radiologically significant daughter products; doses for daughters assume secular equilibrium between parent and daughters.

[c]Entries for Th-228, Ra-224, and Pb-212 daughter products are given under Th-232 above.

Table A-12. Annual effective dose equivalents from external exposure above ground per unit concentration of radionuclides in a 15-cm thickness of soil

Nuclide[a]	Annual dose (rem/y per $\mu Ci/m^3$)	Nuclide[a]	Annual dose (rem/y per $\mu Ci/m^3$)
Co-60	8.30E-3	U-232[b]	-
Cs-137	1.97E-3	U-235	3.68E-4
Eu-152	3.74E-3	Th-231	1.02E-5
Eu-154	4.15E-3	U-238	-
Eu-155	8.05E-5	Th-234	9.41E-6
Ra-226	1.67E-4	Pa-234m	3.88E-5
Pb-214	7.70E-4	Pa-234	1.05E-5
Bi-214	5.13E-3	Np-237	2.79E-5
Th-232	-	Pa-233	6.24E-4
Ac-228	3.14E-3	Am-241	1.45E-5
Pb-212	3.71E-4	Am-243	5.45E-5
Bi-212	6.26E-4	Np-239	3.91E-4
Tl-208	4.17E-3	Cf-249	1.07E-3

[a]Indented entries are radiologically significant daughter products; doses for daughters assume secular equilibrium between parent and daughters.

[b]Entries for Pb-212, Bi-212, and Tl-208 daughter products are given under Th-232 above.

Table A-13. Annual effective dose equivalents from external exposure above ground per unit concentration of radionuclides in an infinite thickness of soil

Nuclide[a]	Annual dose (rem/y per $\mu Ci/m^3$)	Nuclide[a]	Annual dose (rem/y per $\mu Ci/m^3$)
Co-60	1.07E-2	U-232[b]	-
Cs-137	2.33E-3	U-235	3.88E-4
Eu-152	4.63E-3	Th-231	1.03E-5
Eu-154	5.14E-3	U-238	-
Eu-155	8.17E-5	Th-234	9.57E-6
Ra-226	1.76E-5	Pa-234m	4.77E-5
Pb-214	8.65E-4	Pa-234	1.28E-5
Bi-214	6.60E-3	Np-237	2.85E-5
Th-232	-	Pa-233	6.87E-4
Ac-228	3.87E-3	Am-241	1.45E-5
Pb-212	3.97E-4	Am-243	5.49E-5
Bi-212	7.80E-4	Np-239	4.16E-4
Tl-208	5.71E-3	Cf-249	1.20E-3

[a]Indented entries are radiologically significant daughter products; doses for daughters assume secular equilibrium between parent and daughters.

[b]Entries for Pb-212, Bi-212, and Tl-208 daughter products are given under Th-232 above.

Table A-14. Annual effective dose equivalents from external exposure per unit concentration of radionuclides in disposal facility at time intrusion occurs for individual working in contaminated vegetable garden[a]

Nuclide[b]	Annual dose (rem/y per $\mu Ci/m^3$)	Nuclide[b]	Annual dose (rem/y per $\mu Ci/m^3$)
Co-60	1.7E-5	U-232[c]	-
Cs-137	3.9E-6	U-235	7.4E-7
Eu-152	7.5E-6	Th-231	2.0E-8
Eu-154	8.3E-6	U-238	-
Eu-155	1.6E-7	Th-234	1.9E-8
Ra-226	-	Pa-234m	7.8E-8
Pb-214	1.5E-6	Pa-234	2.1E-8
Bi-214	1.0E-5	Np-237	5.6E-8
Th-232	-	Pa-233	1.2E-6
Ac-228	6.3E-6	Am-241	2.9E-8
Pb-212	7.4E-7	Am-243	1.1E-7
Bi-212	1.3E-6	Np-239	7.8E-7
Tl-208	8.3E-6	Cf-249	2.1E-6

[a]Assumptions for calculations are described in Sections A.2.5 and A.3.7.

[b]Indented entries are radiologically significant daughter products; doses from daughters assume secular equilibrium between parent and daughters.

[c]Entries for Pb-212, Bi-212, and Tl-208 daughter products are given under Th-232 above.

These data were obtained from eqs. (A-11) and (A-12) and assume a dilution factor for mixing of all radionuclides from the disposal facility into the native soil of the vegetable garden of[11]

$$f_{is} = 0.2,$$

a fraction of the year during which exposure occurs of

$$U_s = 0.01,$$

which corresponds to an annual exposure time of 100 hours per year,[20] and the dose conversion factors in Table A-12.

The annual effective dose equivalents from external exposure per unit concentration of radionuclides in the disposal facility for an individual living in a house that is constructed immediately on top of the facility are given in Table A-15. These data were obtained from eq. (A-13) and assume a fraction of the year during which exposure occurs of

$$U_t = 0.5,$$

which corresponds to an annual exposure time of 4380 hours per year,[20] a shielding factor from indoor residence for all radionuclides of[7]

$$S_i = 0.7,$$

and the dose conversion factors in Table A-13.

Comparison of the results in Tables A-14 and A-15 shows that the estimated external doses from working in the contaminated vegetable garden are negligible compared with the external doses from residence in a house located on top of the disposal facility. The greater importance of the indoor exposures results primarily from the assumed dilution factor for mixing of radionuclides in the vegetable garden and the much greater time spent indoors.

A.3.8 *Annual doses from inhalation of contaminated soil*

As described in Section A.2.6, the two exposure scenarios assumed for inhalation of suspended activity from soil are essentially the same as the two scenarios for external exposure. The airborne concentration of radionuclides in each case is described using a mass-loading approach.

The annual committed effective dose equivalents from inhalation per unit concentration of radionuclides in the disposal facility for an individual working in the contaminated vegetable garden are given in Table A-16. These data were obtained from eqs. (A-14)-(A-16) and assume a dilution factor for mixing of all radionuclides from the disposal facility into the native soil of the vegetable garden of[11]

Table A-15. Annual effective dose equivalents from external exposure per unit concentration of radionuclides in disposal facility at time intrusion occurs for individual living in house on the facility[a]

Nuclide[b]	Annual dose (rem/y per $\mu Ci/m^3$)	Nuclide[b]	Annual dose (rem/y per $\mu Ci/m^3$)
Co-60	3.7E-3	U-232[c]	-
Cs-137	8.2E-4	U-235	1.4E-4
Eu-152	1.6E-3	Th-231	3.6E-6
Eu-154	1.8E-3	U-238	-
Eu-155	2.9E-5	Th-234	3.3E-6
Ra-226	-	Pa-234m	1.7E-5
Pb-214	3.0E-4	Pa-234	4.5E-6
Bi-214	2.3E-3	Np-237	1.0E-5
Th-232	-	Pa-233	2.4E-4
Ac-228	1.4E-3	Am-241	5.1E-6
Pb-212	1.4E-4	Am-243	1.9E-5
Bi-212	2.7E-4	Np-239	1.5E-4
Tl-208	2.0E-3	Cf-249	4.2E-4

[a]Assumptions for calculations are described in Sections A.2.5 and A.3.7.

[b]Indented entries are radiologically significant daughter products; doses from daughters assume secular equilibrium between parent and daughters.

[c]Entries for Pb-212, Bi-212, and Tl-208 daughter products are given under Th-232 above.

Table A-16. Annual committed effective dose equivalents from inhalation per unit concentration of radionuclides in disposal facility at time intrusion occurs for individual working in contaminated vegetable garden[a]

Nuclide[b]	Annual dose (rem/y per $\mu Ci/m^3$)	Nuclide[b]	Annual dose (rem/y per $\mu Ci/m^3$)
H-3	7.2E-13	Th-232	1.3E-5
Be-10	4.1E-9	Th-228	3.9E-6
C-14	2.7E-13	U-232[c]	7.5E-6
Co-60	2.5E-9	U-233	1.5E-6
Ni-63	2.6E-11	U-234	1.5E-6
Sr-90	1.5E-8	U-235	1.4E-6
Zr-93	8.5E-10	U-236	1.4E-6
Nb-93m	3.3E-10	U-238	1.3E-6
Tc-99	9.5E-11	Np-237	5.5E-6
Cd-113m	1.7E-8	Pu-238	5.3E-6
Sn-121m	1.3E-10	Pu-239	5.9E-6
Cs-137	3.7E-10	Pu-241	1.2E-7
Sm-151	3.4E-10	Pu-242	5.6E-6
Eu-152	2.5E-9	Am-241	6.1E-6
Eu-154	3.3E-9	Am-243	6.1E-6
Eu-155	4.7E-10	Cm-244	3.2E-6
Ra-226	9.8E-8	Cf-249	6.4E-6
Pb-210	1.6E-7		
Po-210	1.1E-7		

[a]Assumptions for calculations are described in Sections A.2.6 and A.3.8.

[b]Indented entries are radiologically significant daughter products; doses from daughters assume secular equilibrium between parent and daughters.

[c]Entry for Th-228 daughter product is given under Th-232 above.

$f_{is} = 0.2$,

a fraction of the year during which exposure occurs of

$f_a = 0.01$,

which corresponds to an annual exposure time of 100 hours per year,[20] an atmospheric mass loading of soil of

$L_a = 10^{-6}$ kg/m^3,

which is ten times greater than the average background value[4] and takes into account the increased suspension during gardening activities, a soil density of[12]

$\rho_s = 1.4 \times 10^3$ kg/m^3,

an annual air intake by an average adult of[7]

$U_a = 8 \times 10^3$ m^3 per year,

and the dose conversion factors in Table A-2.

The annual committed effective dose equivalents from inhalation per unit concentration of radionuclides in the disposal facility for an individual living in a house located immediately on top of the facility are given in Table A-17. These data were obtained from eqs. (A-14) and (A-17) and assume a fraction of the year during which exposure occurs of

$f_a = 0.5$,

which corresponds to an annual exposure time of 4380 hours per year,[20] an atmospheric mass loading of soil of

$L_a = 10^{-7}$ kg/m^3,

which is the average background value,[4] a soil density of[12]

$\rho_s = 1.4 \times 10^3$ kg/m^3,

a ratio of indoor to outdoor air concentration for all radionuclides except ^{3}H and ^{14}C, for which no reduction in indoor concentrations is assumed, of[21]

$S_i = 0.24$,

an annual air intake by an average adult of[7]

$U_a = 8 \times 10^3$ m^3 per year,

and the dose conversion factors in Table A-2.

Table A-17. Annual committed effective dose equivalents from inhalation per unit concentration of radionuclides in disposal facility at time intrusion occurs for individual living in house on the facility[a]

Nuclide[b]	Annual dose (rem/y per $\mu Ci/m^3$)	Nuclide[b]	Annual dose (rem/y per $\mu Ci/m^3$)
H-3	1.8E-11	Th-232	7.9E-5
Be-10	2.4E-8	Th-228	2.3E-5
C-14	6.7E-12	U-232[c]	4.5E-5
Co-60	1.5E-8	U-233	9.3E-6
Ni-63	1.6E-10	U-234	9.1E-6
Sr-90	8.9E-8	U-235	8.4E-6
Zr-93	5.1E-9	U-236	8.6E-6
Nb-93m	2.0E-9	U-238	8.1E-6
Tc-99	5.7E-10	Np-237	3.3E-5
Cd-113m	1.0E-7	Pu-238	3.2E-5
Sn-121m	7.9E-10	Pu-239	3.5E-5
Cs-137	2.2E-9	Pu-241	7.1E-7
Sm-151	2.1E-9	Pu-242	3.4E-5
Eu-152	1.5E-8	Am-241	3.6E-5
Eu-154	2.0E-8	Am-243	3.6E-5
Eu-155	2.8E-9	Cm-244	1.9E-5
Ra-226	5.9E-7	Cf-249	3.9E-5
Pb-210	9.3E-7		
Po-210	6.5E-7		

[a]Assumptions for calculations are described in Sections A.2.6 and A.3.8.

[b]Indented entries are radiologically significant daughter products; doses from daughters assume secular equilibrium between parent and daughters.

[c]Entry for Th-228 daughter product is given under Th-232 above.

Comparison of the results in Tables A-16 and A-17 shows that, except for ^{3}H and ^{14}C, the inhalation exposures from working in the vegetable garden are about one-sixth of those from residing in the house on the disposal facility. In this case, the effects of dilution of the radionuclides in soil and the smaller exposure time while working outdoors are somewhat compensated by the assumptions for indoor exposures of the smaller atmospheric mass loading and the reduction in air concentration provided by the building.

A.3.9 *Comparisons of annual doses for different pathways*

The tables of annual committed effective dose equivalents per unit concentration of radionuclides in the environment developed in Sections A.3.3-A.3.8 are normalized either to a unit concentration in water or to a unit concentration in the disposal facility itself at the time intrusion occurs. Therefore, the results for the different exposure pathways that result from activity in the same environmental compartment can be compared directly to evaluate their relative importance for each radionuclide.

The relative contributions from the drinking water, milk, and meat pathways that result from intakes of radionuclides in contaminated water are given in Table A-18. The contributions from these pathways are obtained from Tables A-4, A-10, and A-11, and the results are normalized to unity for the drinking water pathway. The results for parent and daughter radionuclides are combined into a single entry by assuming that the parent and daughters are in secular equilibrium. The table shows that, for the assumptions used in the analyses, the milk and meat pathways are important relative to the drinking water pathway only for a few of the fission and activation products. For all other radionuclides, the drinking water pathway is the only one that needs to be considered in evaluating annual dose equivalents from contaminated water.

The relative contributions from the vegetable, soil ingestion, external exposure, and inhalation pathways that result from radionuclides in the disposal facility itself at the time intrusion occurs are given in Table A-19. The contributions from these pathways are obtained from Tables A-6, A-7, and A-14 through A-17, and the results are normalized to unity for the vegetable pathway. The entries for parent and daughter radionuclides again are combined by assuming that the parent and daughters are in secular equilibrium. The table shows that while the vegetable pathway is usually the most important for the fission and activation products, with the exception of ^{60}Co, ^{137}Cs, and the Eu isotopes which are all strong photon emitters, ingestion of contaminated soil and external

Table A-18. Relative contribution to intruder doses from exposure pathways resulting from radionuclides in water

Nuclide[a]	Drinking water	Milk	Meat
H-3	1.0	0.25	0.15
Be-10	1.0	2E-5	0.22
C-14	1.0	0.26	0.37
Co-60	1.0	0.05	0.12
Ni-63	1.0	0.02	0.03
Sr-90 + d	1.0	0.02	0.004
Zr-93 + d	1.0	0.09	0.94
Tc-99	1.0	0.18	0.01
Cd-113m	1.0	0.03	0.004
Sn-121m	1.0	0.02	0.26
Cs-137	1.0	0.13	0.24
Sm-151	1.0	0.001	0.009
Eu-152	1.0	0.001	0.009
Eu-154	1.0	0.001	0.009
Eu-155	1.0	0.001	0.009
Ra-226 + d	1.0	0.005	0.02
Th-232 + d	1.0	0.003	0.004
U-232 + d	1.0	0.005	0.004
U-233	1.0	0.006	0.004
U-234	1.0	0.006	0.004
U-235	1.0	0.006	0.004
U-236	1.0	0.006	0.004
U-238 + d	1.0	0.006	0.004
Np-237	1.0	9E-5	1E-5
Pu-238	1.0	2E-6	1E-5
Pu-239	1.0	2E-6	1E-5
Pu-241	1.0	2E-6	1E-5
Pu-242	1.0	2E-6	1E-5
Am-241	1.0	8E-6	4E-5
Am-243	1.0	8E-6	4E-5
Cm-244	1.0	4E-4	4E-5
Cf-249	1.0	4E-4	4E-5

[a]"d" denotes radioactive daughter products that are assumed to be in secular equilibrium with the parent.

Table A-19. Relative contribution to intruder doses from exposure pathways resulting from radionuclides in disposal facility at time intrusion occurs

Nuclide[a]	Vegetables	Soil ingestion	External	Inhalation
H-3	1.0	8E-5	-	5E-6
Be-10	1.0	0.001	-	0.002
C-14	1.0	0.007	-	6E-6
Co-60	1.0	0.01	3.7E2	0.002
Sr-90 + d	1.0	0.003	-	4E-4
Zr-93 + d	1.0	0.03	-	0.02
Tc-99	1.0	8E-5	-	7E-6
Cd-113m	1.0	0.003	-	4E-4
Sn-121m	1.0	0.13	-	0.01
Cs-137	1.0	0.04	1.3E2	4E-4
Sm-151	1.0	0.09	-	0.10
Eu-152	1.0	0.09	4.0E3	0.04
Eu-154	1.0	0.09	3.1E3	0.04
Eu-155	1.0	0.09	3.1E2	0.04
Ra-226 + d	1.0	0.04	2.3	0.002
Th-232 + d	1.0	0.04	6.0	0.19
U-232 + d	1.0	0.08	18	0.61
U-233	1.0	2.4	-	17
U-234	1.0	2.4	-	17
U-235 + d	1.0	2.4	2.4E2	17
U-236	1.0	2.4	-	17
U-238 + d	1.0	2.2	38	14
Np-237 + d	1.0	0.005	0.06	0.009
Pu-238	1.0	5.1	-	9.6
Pu-239	1.0	5.1	-	9.6
Pu-241	1.0	5.1	-	9.6
Pu-242	1.0	5.1	-	9.6
Am-241	1.0	19	4.3	35
Am-243 + d	1.0	19	1.4E2	35
Cm-244	1.0	27	-	50
Cf-249	1.0	5.1	1.8E2	19

[a]"d" denotes radioactive daughter products that are assumed to be in secular equilibrium with the parent.

and inhalation exposures are the most important for all actinides and their daughter products.

A.4 Summary of Dose Calculations

This appendix has presented a methodology for estimating annual committed effective dose equivalents resulting from inadvertent intrusion into a near-surface radioactive waste disposal facility. Six different exposure pathways were assumed to occur: (1) ingestion of contaminated drinking water from a source at the site, (2) ingestion of vegetables grown in native soil that is contaminated by soil from the disposal facility, (3) direct ingestion of contaminated soil from the vegetable garden, (4) ingestion of milk and meat from dairy and beef cattle that drink contaminated water from a source at the the site, (5) external exposure to contaminated soil while working in the vegetable garden or residing in a house located on the disposal facility, and (6) inhalation of suspended activity from contaminated soil while working in the vegetable garden or residing in a house located on the disposal facility. The two scenarios involving ingestion of water by man or by dairy and beef cattle result from release of radionuclides from the disposal facility into a source of drinking water (i.e., an aquifer or a surface stream). The other four scenarios result from exposure to radionuclides retained in the disposal facility itself at the time intrusion occurs.

The dose estimates obtained from this analysis are summarized in Tables A-20 and A-21, which give annual committed effective dose equivalents per unit concentration of radionuclides in water or in the disposal facility itself, respectively. These results then can be multiplied by estimates of the concentrations of each radionuclide in the two environmental media at the time intrusion is assumed to occur, as obtained from considerations of radioactive decay and environmental transport of radionuclides placed in the disposal facility, to obtain estimates of annual dose equivalents to intruders.

The dose analysis for an inadvertent intruder was based on reasonably realistic estimates of parameters for food-chain transport of radionuclides and for exposure times or annual intakes of contaminated materials. However, since the probability that the intrusion scenarios occur is assumed to be unity at any time following loss of institutional controls, the calculations are believed to provide conservative overestimates of actual risks to inadvertent intruders.

The results in Tables A-20 and A-21 can be used to determine limits on concentrations of radionuclides that may be placed in a disposal facility. For the pathways involving exposure to radionuclides in the

Table A-20. Summary of annual committed effective dose equivalents per unit concentration of radionuclides in water[a]

Nuclide[b]	Annual dose (rem/y per μCi/L)	Nuclide[b]	Annual dose (rem/y per μCi/L)
H-3	3.2E-2	Th-232 + d	1.8E3
Be-10	2.1	U-232 + d	7.9E2
C-14	9.3E-1	U-233	1.1E2
Co-60	1.2E1	U-234	1.1E2
Ni-63	2.1E-1	U-235	9.8E1
Sr-90 + d	5.7E1	U-236	1.0E2
Zr-93 + d	1.6	U-238 + d	9.9E1
Tc-99	6.4E-1	Np-237	1.5E3
Cd-113m	6.0E1	Pu-238	1.4E3
Sn-121m	7.2E-1	Pu-239	1.6E3
Cs-137	2.6E1	Pu-241	3.2E1
Sm-151	1.4E-1	Pu-242	1.5E3
Eu-152	2.4	Am-241	1.6E3
Eu-154	3.5	Am-243	1.6E3
Eu-155	5.7E-1	Cm-244	8.5E2
Ra-226 + d	3.2E3	Cf-249	8.6E2

[a]Results are sum of annual dose equivalents per unit concentration from drinking water, milk, and meat pathways in Tables A-4, A-10, and A-11, respectively.

[b]"d" denotes radioactive daughter products that are assumed to be in secular equilibrium with the parent.

Table A-21. Summary of annual committed effective dose equivalents per unit concentration of radionuclides in disposal facility at time intrusion occurs[a]

Nuclide[b]	Annual dose (rem/y per $\mu Ci/m^3$)	Nuclide[b]	Annual dose (rem/y per $\mu Ci/m^3$)
H-3	3.9E-6	Th-232 + d	4.5E-3
Be-10	2.2E-5	U-232 + d	2.4E-3
C-14	1.1E-6	U-233	1.2E-5
Co-60	3.7E-3	U-234	1.2E-5
Ni-63	2.4E-7	U-235 + d	1.4E-4
Sr-90 + d	2.9E-4	U-236	1.2E-5
Zr-93 + d	4.1E-7	U-238 + d	3.5E-5
Tc-99 + d	9.4E-5	Np-237 + d	4.3E-3
Cd-113m	3.3E-4	Pu-238	6.1E-5
Sn-121m	7.2E-8	Pu-239	6.9E-5
Cs-137	8.2E-4	Pu-241	1.4E-6
Sm-151	2.9E-8	Pu-242	6.6E-5
Eu-152	1.6E-3	Am-241	7.0E-5
Eu-154	1.8E-3	Am-243 + d	2.3E-4
Eu-155	2.9E-5	Cm-244	3.4E-5
Ra-226 + d	3.7E-3	Cf-249	4.9E-4

[a]Results are sum of annual dose equivalents per unit concentration from vegetable, soil ingestion, external exposure, and inhalation pathways in Tables A-6, A-7, A-14 and A-15, and A-16 and A-17, respectively.

[b]"d" denotes radioactive daughter products that are assumed to be in secular equilibrium with the parent.

disposal facility itself, for example, a limit on annual dose equivalent to an inadvertent intruder divided by the factor in Table A-21 gives a concentration limit for each radionuclide at the time intrusion is assumed to occur, provided the dose from transport to a source of drinking water is unimportant. These concentration limits then can be increased for those radionuclides that decay significantly over the period of institutional controls to give concentration limits for placement in the disposal facility.

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