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Volume 15: Uranium-238

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J. P. Adams

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Idaho National Engineering Laboratory
Lockheed Idaho Technologies Company
Idaho Falls, Idaho 83415

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ABSTRACT

This report, Volume 15 of the National Low-Level Waste Management Program Radionuclide Report Series, discusses the radiological and chemical characteristics of uranium-238 (^{238}U). This report also includes discussions about waste types and forms in which ^{238}U can be found, and ^{238}U behavior in the environment and in the human body.

DISCLAIMER

FOREWORD

The purpose of the National Low-Level Waste Management Program Radionuclide Report Series is to provide information to state representatives and developers of low-level radioactive waste disposal facilities about the radiological, chemical, and physical characteristics of selected radionuclides and their behavior in the low-level radioactive waste disposal facility environment. Extensive surveys of available literature provided information used to produce this series of reports and an introductory report.

The National Low-Level Waste Management Program Radionuclide Report Series previously addressed the radionuclides technetium-99, carbon-14, iodine-129, tritium, cesium-137, strontium-90, nickel-59, plutonium-241, nickel-63, niobium-94, cobalt-60, curium-242, and americium-241. These radionuclides contribute significantly to the dose estimated during a performance assessment analysis.

This report is Volume 15 of the series. It outlines the basic radiological and chemical characteristics of uranium-238, waste types and forms that contain this radionuclide and its behavior in environmental media such as soils, plants, water, air, animals, and the human body. Additional reports will be generated for other radionuclides.

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National Low-Level Waste Management Program Radionuclide Report Series

Volume 15: Uranium-238

INTRODUCTION

This report outlines the basic radiological and chemical characteristics of uranium-238 (^{238}U) and examines how these characteristics affect the behavior of ^{238}U in various environmental media, such as soils, groundwater, plants, animals, the atmosphere, and the human body. Discussions also include methods of ^{238}U production, waste types, and waste forms that contain ^{238}U.

Because of the long half-life (and consequently low specific activity) of ^{238}U, it is not, itself, a major factor in the ability of a low-level radioactive waste disposal facility to meet performance objectives. The presence of the nuclide, however, can be relatively significant in assessing the long-term performance of such a site due to the quantity, radiotoxicity, and mobility of its daughter products, which include isotopes of radium and radon. For this reason, it was decided to include ^{238}U in this report series.

Uranium was first isolated by Martin Heinrich Klaproth in 1789 from a sample of pitchblende in Saxony.[1] At first, it was assumed that the new "element" was a semimetal due to its apparent chemical and physical characteristics. In 1841, Eugene-Melchior Peligot showed that the original "element" was, in fact, the oxide and was able to reduce the oxide to obtain the first pure metal. In 1896, Antoine-Henri Becquerel discovered that uranium undergoes radioactive decay. In 1939, Otto Hahn and Fritz Strassmann discovered the phenomenon of spontaneous fissioning, which ultimately vaulted uranium from a position of relative obscurity to a role of major importance.

All uranium atoms contain 92 protons ($Z = 92$) and various numbers of neutrons ($N = 135$ to 148) within the atom's nucleus. There are no stable isotopes of uranium; all are radioactive. The half-lives of uranium isotopes range from less than 2 minutes (^{227}U) to billions of years (^{238}U). Even though all of the isotopes of uranium are radioactive, three of the isotopes have sufficiently long lifetimes and exist in nature for long periods of time. (^{234}U half-life $= 2.5 \times 10^5$ yr, 0.0057% of naturally occurring uranium; ^{235}U half-life $= 7.1 \times 10^8$ yr, 0.71% of naturally occurring uranium; and ^{238}U, half-life $= 4.5 \times 10^9$ yr, 99.3% of naturally occurring uranium.) Essentially all of the existing ^{238}U is naturally occurring, although a small fraction is produced by neutron activation of lower mass nuclides. For the purpose of this report, it will be assumed that all of this nuclide is natural and none is manmade.

Before 1942, uranium was used principally as a coloring agent in glass and ceramic glazes, resulting in a yellow, orange, or red (depending on the chemical form) tint to objects where it was used.[2] The principal source for this coloring agent was as a byproduct of the production of radium and vanadium. However, discovery by Otto Hahn and Fritz Strassmann in 1939 of the ability of uranium to fission, creating enormous amounts of energy in the process, led to greatly heightened interest in this element.[1] After demonstrating controlled nuclear fission in 1942, this element was sought after as a source of heat to drive steam generators and as an explosive material for military uses. Additional uses for this element have been discovered and will be discussed in later sections of this report.

Although ^{238}U is the predominant isotope of uranium, a lower mass isotope, ^{235}U, is the nuclide most often used for energy purposes. This has given rise to a new term to describe ^{238}U, namely "depleted uranium." This is because as ^{235}U is extracted from uranium, that which is left is uranium "depleted" of this desirable isotope and contains an even higher fraction of ^{238}U. Thus, depleted uranium and ^{238}U are used interchangeably in this report.

The primary source of ^{238}U in the environment is a natural ore, undisturbed by humans. The principal manmade source of this nuclide is uranium that has been depleted of its original store of ^{235}U either by the enrichment process (whereby the ^{235}U is extracted from the uranium, leaving depleted uranium as a byproduct) or by removal of ^{235}U from uranium in the fissioning process. The enrichment process results in large masses of depleted uranium that can be used for other purposes (discussed later in this report). The fissioning process results in large amounts of ^{238}U that are contaminated with highly radioactive fission and activation products and which are generally disposed of as high-level wastes. Because it does not generally result in disposal of ^{238}U as low-level waste, the second process (fissioning) is discussed only briefly in this report.

The specific activity of ^{238}U is relatively small. Therefore, this nuclide can be handled with standard laboratory equipment, including ventilated hoods or enclosures used for general laboratory purposes.[3] However, as discussed later in this report, the radiotoxicity of this nuclide is classified as high. This may be because of the radiotoxicity of its decay daughters.

The rest of this report includes discussions of radiological, chemical, and physical characteristics of this nuclide, production and waste types, and the behavior of ^{238}U in the environment, in animals, and in humans. In most cases, only brief, summary information has been extracted from the references in writing this report. Additional information can be obtained from the references and bibliography provided at the end of this report.

RADIOLOGICAL CHARACTERISTICS

The most recent information regarding ^{238}U reports a radiological half-life of 4.47×10^9 yr.[4] Despite the fact that this nuclide is radioactive, the principal source is natural uranium ore, of which ^{238}U comprises more than 99%.

As is the case for curium-242 (^{242}Cm), reported in a previous report in this series[5], the decay of ^{238}U is very complex. The principal decay chain from ^{238}U to the stable nuclide lead-206 (^{206}Pb) is[6]

$$\begin{aligned}
&{}^{238}_{92}U \underset{\alpha}{\overset{4.47x10^{9}y}{\rightarrow}} {}^{234}_{90}Th \underset{\beta}{\overset{24d}{\rightarrow}} {}^{234m}_{91}Pa \underset{\beta}{\overset{1.18m}{\rightarrow}} {}^{234}_{92}U \underset{\alpha}{\overset{247{,}000y}{\rightarrow}} {}^{230}_{90}Th \underset{\alpha}{\overset{77{,}000\ y}{\rightarrow}} {}^{226}_{88}Ra \underset{\alpha}{\overset{1602y}{\rightarrow}} \\
&{}^{222}_{86}Rn \underset{\alpha}{\overset{3.8d}{\rightarrow}} {}^{218}_{84}Po \underset{\alpha}{\overset{3.1\ \mathrm{min}}{\rightarrow}} {}^{214}_{82}Pb \underset{\beta^-}{\overset{26.8\mathrm{min}}{\rightarrow}} {}^{214}_{83}Bi \underset{\beta^-}{\overset{19.7\mathrm{min}}{\rightarrow}} {}^{214}_{84}Po \underset{\alpha}{\overset{164\mu s}{\rightarrow}} \\
&{}^{210}_{82}Pb \underset{\beta^-}{\overset{22y}{\rightarrow}} {}^{210}_{83}Bi \underset{\beta^-}{\overset{5d}{\rightarrow}} {}^{210}_{84}Po \underset{\alpha}{\overset{138d}{\rightarrow}} {}^{206}_{82}Pb\ (stable)
\end{aligned} \tag{1}$$

In fact, with the exception of the first three nuclides, this decay chain is identical to the decay of ^{242}Cm (see Equation 3 of Reference 5). This is the most likely decay process, based on the principal branch for the decay of each nuclide ground state. For several of these decay processes, there is a competing process. Many of the alpha-decay processes also result in the emission of various energy gamma-rays, and the beta-decay processes result in the emission of various energy beta particles and gamma-rays. A very small fraction ($<1\%$) of the decays of ^{238}U result in spontaneous fission.[7] The fraction of decays resulting in spontaneous fission is not documented, but the half-life of this process is of the order of 10^{15} years, or approximately one million times as long as for alpha decay, which gives an indication of this fraction.

With the exceptions of plutonium-241 (^{241}Pu),[8] americium-241 (^{241}Am),[9] and ^{242}Cm[5], all other nuclides included in this report series decay either directly to a stable nuclide or reach a stable nuclide within two decays. Therefore, this (^{238}U) decay scheme is one of the most complex included in this report series thus far.

No attempt has been made in this study to correlate the potential effects of radiation associated with the nuclide decay listed in Equation (1) because that would have been beyond the scope of the task. Only the effects associated with the direct decay of ^{238}U will be included. Obviously, these other nuclides will affect the behavior of uranium in a low-level radioactive waste site (these daughter nuclides dominate the radiological consequences)[a] and their effects need to be considered in any comprehensive study of the behavior of uranium. The types and energies of the radiation associated with the subsequent decays are provided in the references.

The decay of ^{238}U results in the emission of alpha particles with three principal energies, 4.04 million electron volts (MeV)—1% of decays; 4.15 MeV—23% of decays; and 4.20 MeV—77% of

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[a] Private communication between J. P. Adams (INEL) and D. F. Harmon (INEL), August 8, 1995.

decays.[5] There is electromagnetic radiation that is emitted during the decay of a ^{238}U nucleus. The x-ray and gamma-ray energies range from 11 to 110 thousand electron volts (keV), although the most probable energy is 16 keV. Finally, electrons may be emitted during the decay of ^{238}U, with principal energies of 30 and 43 keV.[7] Again, these emissions are only from the direct decay of ^{238}U and do not include the emissions from the subsequent decay of the ground states of resultant nuclides, as shown in Equation (1).

Since a released electron is often emitted with an energy less than its theoretical maximum, an antineutrino particle is simultaneously emitted and this particle carries off the energy difference between the released electron and the total energy given off by the associated decay process. Consequently, electrons are emitted with a continuous energy spectrum ranging from zero to the maximum energy. Since neutrinos and antineutrinos rarely interact with matter, they are not considered radiologically important. Therefore, the antineutrino particle is usually not shown in the overall decay equation.

As stated above, approximately 77% of the decays of ^{238}U result in the emission of an alpha particle with an energy of 4.20 MeV. The range of these alpha particles is approximately 2.6 cm in air. Thus, even a small distance removed from a ^{238}U source would provide sufficient shielding from the alpha radiation. The 43 keV electron that can be emitted during the decay would have a range of 3.0 mg/cm^2 (to obtain the linear range in a given material, divide this value by the density of the material).

In addition to alpha and electron particle radiation, the decay of ^{238}U can also result in x-ray and gamma-ray emission, with a maximum listed energy of 110 keV. The probability that the decay of a ^{238}U nucleus would result in the emission of a 110 keV gamma-ray is approximately 0.02%, which is small but still finite. Table 1 lists the half-value layers (the thickness of a given material that would result in the reduction of a stream of 110 keV gamma-rays by one-half) for a photon of this energy in several common materials. The values in this table were derived using data in Tables 5.1 and 5.4 of Reference 10.

Table 2 lists the radiotoxicity of several important radionuclides and Table 3 compares the electron emission energy associated with ^{238}U decay together with the maximum and average electron energies associated with electron emitters. ^{238}U is listed with other nuclides that have "High Toxicity" in Table 2. This designation is based on Table I in Reference 11. The basis is not given for listing this nuclide with the others in this classification, although the definition of toxicity is stated. It is assumed to be principally due to the emission of relatively high energy alpha particles since neither the gamma-ray nor electron emissions would be considered high energy. Another possible reason for this designation is the presence of one or more of the decay daughters. The most energetic (and most prevalent) alpha particle energy of 4.2 MeV is less than the maximum alpha particle energies for the other nuclides in this classification [4.78 MeV for neptunium-237 (^{237}Np); 4.85 MeV for ^{241}Pu; 5.49 MeV for ^{241}Am; and 6.12 MeV for ^{242}Cm]. The specific activity of ^{238}U is 3.39×10^{-7} Ci/g.[6]

Table 1. Half-value layers for the 110 keV gamma-ray emitted from the decay of a ^{238}U nucleus.[a]

Material	Half-value layer thickness (cm)[b]
Lead	0.013
Aluminum	1.57
Bone	2.15
Water	4.15
Concrete	17.5
Air	3,570

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a. The half-value layer for a given material is the thickness of the material that will attenuate a beam of gamma-rays (of a specific energy) to one-half of its intensity.

b. These values were taken from Tables 5.1 and 5.4 of Reference 10.

Table 2. Comparison of the radiotoxicity[a] of several important radionuclides.

Radiotoxicity	Group	Nuclides
High toxicity		^{237}Np, ^{242}Cm, ^{241}Am, ^{238}U
Medium toxicity	Upper subgroup A	^{129}I, ^{60}Co, ^{137}Cs
Medium toxicity	Lower subgroup B	^{63}Ni, ^{99}Tc, ^{59}Ni, ^{14}C
Low toxicity		^{3}H, ^{99m}Tc

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a. The definition of radiotoxicity, as stated in the reference, is "...the ability of the nuclide to produce injury, by virtue of its emitted radiations, when incorporated in a body."

Table 3. Average and maximum kinetic energies of electrons released during decay of several important radionuclides. (Information compiled from data presented in References 7, 9, 12, 13, and 14.)

Radionuclide	Released electron energy Average energy (keV)[a]	 Maximum energy (keV)
Nickel-59	4.1[b]	~7.7
Tritium	5.7	19.0
Nickel-63	17.1	67.0
Curium-242	39[c]	—
Iodine-129	40.0	150.0
Uranium-238	**43[c]**	—
Carbon-14	49.0	156.0
Technetium-99	85.0	293.0
Niobium-94	156.0	500.0
Cesium-137	195.0	1176.0
Cobalt-60	94.0	1480.0

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a. 1,000 keV = 1 MeV

b. The data for ^{59}Ni represent Auger electrons and not electrons emitted from the nucleus.

c. The electron energies for ^{242}Cm and ^{238}U represent the highest energy major emission electron group and may not represent either the average or maximum energy.

CHEMICAL AND PHYSICAL CHARACTERISTICS

Uranium is a member of the actinide series of the periodic table, which includes actinium, thorium, protactinium, neptunium, plutonium, americium, curium, berkelium, californium, einsteinium, fermium, mendelevium, nobelium, and lawrencium. This series lies between radium and rutherfordium and is chemically similar to the lanthanide series, which is the group of elements between barium and hafnium. In both cases, the series consists of elements whose inner electron shell of 14 "f" electrons is being filled. In the case of the actinides, the 5f shell is being filled, and in the case of the lanthanides, the 4f shell is being filled. This results in two series, each of whose members are chemically similar to a member of the other series. For the case of uranium, it is chemically similar to neodymium. Although the members of the actinite and lanthanide series are chemically very similar, there are significant differences. One difference is that the members of the actinide series are much more prone to complex formation whereas most compounds formed with members of the lanthanide series are ionic in nature.[15]

Uranium has one of the highest densities of all metals and has three crystalline structures in the solid phase. It forms a wide range of intermetallic compounds (e.g., with manganese, nickel, and tin). It is chemically reactive and combines directly with most of the elements in the periodic table. In air, the surface is rapidly converted to a yellow and subsequently a black nonprotective film. Powdered uranium metal is frequently pyrophoric, that is, if it is finely divided (e.g., as a powder), it can be ignited and sustain combustion.

The chemistry of uranium has been widely studied. In fact, it may be the best known of all metal chemistries. For example, metallic uranium reacts rapidly, and reversibly, with hydrogen isotopes to form a stable black powder, uranium hydride (UH_3). This reaction can be easily reversed simply by elevating the temperature of the powder. This characteristic has been extensively exploited in the nuclear industry by using uranium (principally depleted uranium) as a storage medium for hydrogen isotopes, such as tritium. Whenever the tritium needs to be available, the uranium bed is heated up and the tritium evolves. These beds can be cycled multiple times, making uranium an ideal storage medium.

The uranium-oxygen system is one of the most complex oxide systems known. This is because uranium can exist in many oxidation states. One of the most common oxides is UO_2, which is used extensively in nuclear fuel. Another common oxide is U_3O_8. In all, there are six stoichiometric oxides of uranium, UO_2, U_4O_9, U_3O_7, U_2O_5, U_3O_8, and UO_3.[16] In addition, the uranium oxide can deviate from stoichiometry (for example, additional oxygen atoms can be added to UO_2 without affecting notable structural changes in the oxide—extending the phase from UO_2 to $UO_{2.25}$).

Uranium also chemically reacts with several of the halides to form compounds. Oxidation states vary from +3 (e.g., UF_3, UCl_3, UBr_3, and UI_3) to +6 (e.g., UF_6 and UCl_6). UF_6 is a readily accessible volatile uranium compound and is used extensively in the uranium enrichment process, wherein ^{235}U is separated from ^{238}U by a series of membrane filter units.

Uranium also forms oxyhalides and many other binary compounds. For example, uranium forms stable compounds with boron, carbon, silicon, nitrogen, phosphorus, tin, selenium, and sulfur. It also can form uranyl salts.

Uranium metal can be melted by any of several different techniques. However, because uranium is very reactive when heated in air, melting must be done either under a protective inert atmosphere or in a vacuum.[17] Uranium and its alloys are considered difficult to machine and almost all machining of uranium results in some sparking or burning because of its pyrophoricity. Problems associated with machining are due to a combination of characteristics, including toughness, stringiness, abrasiveness, galling, and work hardening.

Health and safety considerations must be carefully considered when using uranium because of its high toxicity and pyrophoricity. The main hazard to health occurs where finely divided particles can become airborne and inhaled. For this reason, vents and fume hoods should be used, or the workers should use respirator equipment to avoid inhalation. Additionally, special clothing and footwear should be used to minimize spread of contamination.

Uranium is the heaviest naturally occurring element that exists in recoverable amounts, averaging about 4 ppm in the earth's crust.[14] The majority of leaching processes for processing of uranium ore involves leaching the crushed and ground ore with sulfuric acid. This means that uranium tailings are often acidic.

There are several types of uranium ore. These are named after some distinguishing feature, such as the host rock, mineralizing process, or structural and stratigraphic setting. They include quartz-pebble conglomerate, unconformity-related vein, disseminated, contact, classical vein, sandstone, volcanogenic, and surface deposits. They are widely distributed through the earth's crust, and economical mining of uranium has taken place in Australia, North America, South America, Asia, Europe, and Africa.

Table 4 summarizes some of the important chemical and physical characteristics of this element.

Table 4. Chemical and physical characteristics of uranium (from References 15, 18, and 19).

Physical property (units)	Uranium data
Melting point (K)[a]	2,123
Boiling point (K)	4,091
Heat of fusion (kJ/mole)	19.7
Heat of vaporization (kJ/mole)	446.4
Heat of sublimation @ 0 K (kJ/mole)	487.9
Heat capacity @ 298 K (kJ/mole)	27.7
Thermal conductivity @ 343 K (J/cm • s)	0.29
Electrical conductivity $(\Omega \bullet \text{cm})^{-4}$	$2\text{-}4 \times 10^4$
Density (g/cm^3)	19
Crystal form[b]	Alpha-orthorhombic Beta-complex tetragonal Gamma-body-centered cubic

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a. Zero Kelvin equals -273.15°C, -459.67°F.

b. The transition temperatures between crystalline phases are 941 K for alpha to beta and 1,048 K for beta to gamma.

URANIUM-238 PRODUCTION

^{238}U is not produced by humans. It exists as a naturally-occurring nuclide in nature. This nuclide comprises more than 99% of all existing natural uranium and makes up approximately 4 ppm of the earth's crust. For use in nuclear reactors, uranium (enriched to a higher concentration of ^{235}U) is typically formed into UO_2 pellets, though in some reactors metallic uranium is used (often alloyed with another metal such as aluminum).

Economically minable ores (defined as containing at least 500 ppm uranium) are available, though not abundant.[14] Deposits occur in nearly every major rock type. The majority of uranium ore processing techniques involve leaching with sulfuric acid. This means that uranium tailings typically are acidic in pH, which affects the solubility of uranium in groundwater and soil.

Between the years of 1948 and 1982, the production of uranium in the free world totaled about 828,000 tons of U_3O_8. Of this, 42% was mined in the United States, 22% in Canada, and 15% in South Africa. The total known resources in the free world (defined as reasonably assured resources available using economic mining techniques) total 1.68 million tons of U_3O_8 in 1983.[14] Estimates for the total uranium content of the earth's crust, to a depth of 25 km is calculated to be 4.5×10^{17} kg (5.0×10^{14} tons) and in the oceans, 4.5×10^{13} kg (5.0×10^{10} tons).[1]

WASTE TYPES AND FORMS THAT CONTAIN URANIUM-238

Depleted uranium (^{238}U) has been deposited in the two low-level radioactive waste sites in Richland, Washington, and Barnwell, South Carolina. Figures 1 and 2 show the activity deposited in these two sites as a function of time for the years 1986–1994. The data for these two figures, as well as for Figures 3 and 4, were obtained from the Manifest Information Management System data base.[b] As can be seen, most of the depleted uranium was deposited in the Barnwell site, more than 10 times the activity stored at the Richland site. Deposition at both sites has decreased since 1990. Figures 3 and 4 show the sources of the total depleted uranium disposed at these two sites. In both cases, industrial uranium represents approximately 90% of the total depleted uranium stored at these two sites. Nearly all of the remainder is brokered uranium. Utility and medical sources of depleted uranium represent much less than 1% of the total at either site.

The definition of disposal categories given in Figures 3 and 4 are as follows: academic—sources associated with university laboratories; government—sources associated with the government, such as military hospitals; industrial—sources associated with manufacturing, other than utilities; medical—sources associated with hospitals and medical laboratories, excluding pharmaceutical manufacturing; brokered—sources accepted by middlemen who then ship the waste to the disposal sites. Waste treatment facilities are included in the brokered category.

In this section, the types of activities that generate depleted uranium low-level radioactive waste are discussed.

Nuclear Reactors

Much of the uranium that is used in commercial applications is used to produce nuclear fuel. A typical ^{235}U enrichment for commercial reactors is approximately 3% (i.e., 3% of the uranium in a nuclear fuel rod is ^{235}U, the rest being ^{238}U). Since it is the "fissile" ^{235}U that is "burned" up in the fission reaction, most of the ^{238}U remains when the fuel rods are taken out of the reactor and sent to storage. However, this ^{238}U is mixed with highly radioactive fission products and activation products that result from the fissioning process. The presence of these other species means that this ^{238}U cannot be disposed of as low-level radioactive waste without extensively reprocessing the fuel to remove the fission and activation products. As a consequence, this source of ^{238}U does not end up in low-level radioactive wastestreams. However, the manufacture of these enriched fuel elements results in the production of depleted uranium (uranium with less than the usual mass of ^{235}U). For each kilogram of uranium enriched to 3% ^{235}U, 5–6 kg of depleted uranium are produced, which contain between 0.2 and 0.3% ^{235}U. This is a major source of ^{238}U for nonnuclear applications, discussed below.

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[b] Private communication between R. L. Fuchs, INEL, and J. P. Adams, INEL, June 14, 1995, and July 24, 1995.

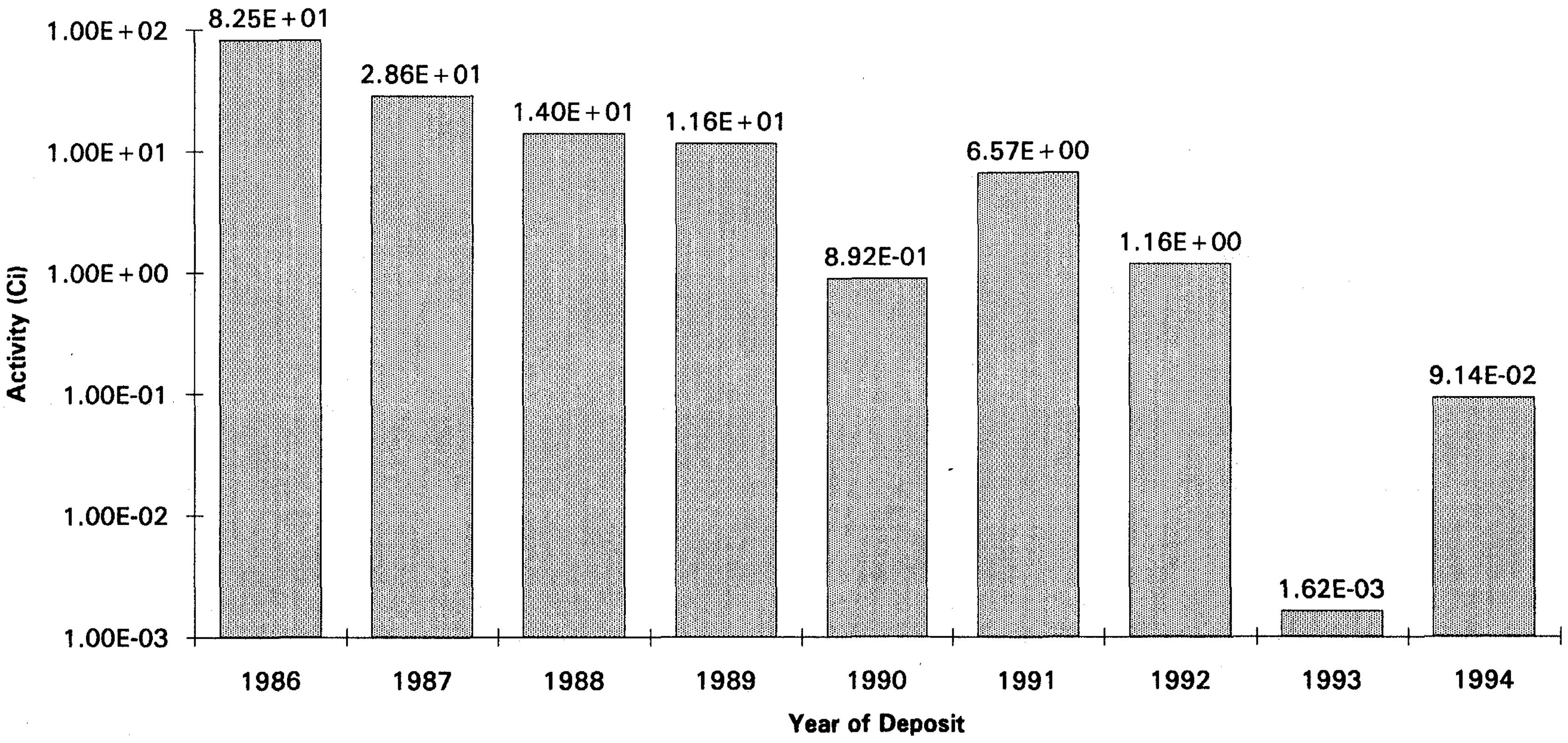

Figure 1. Uranium-238 deposited in the Richland Low-Level Waste Disposal Site as a function of time (log scale).

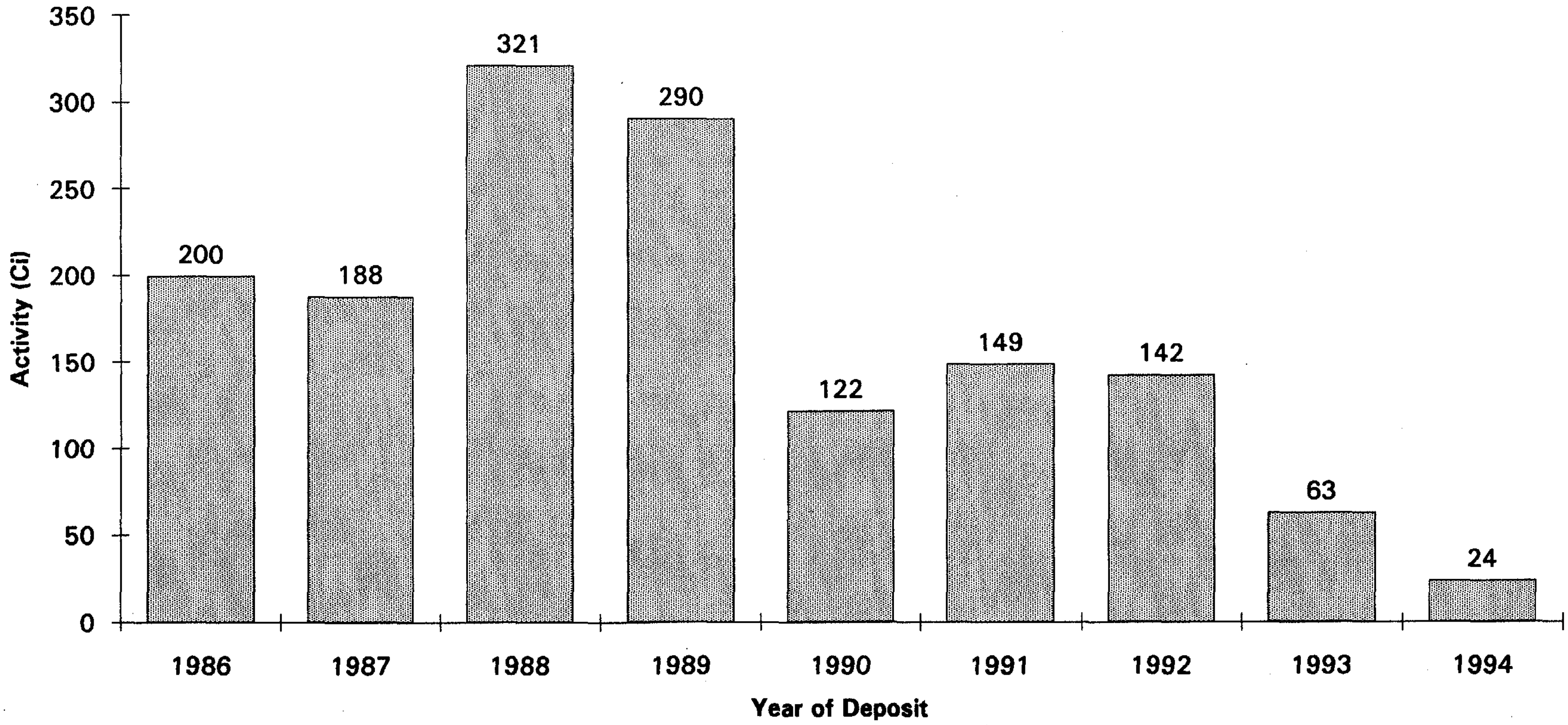

Figure 2. Uranium-238 deposited in the Barnwell Low-Level Waste Disposal Site as a function of time (log scale).

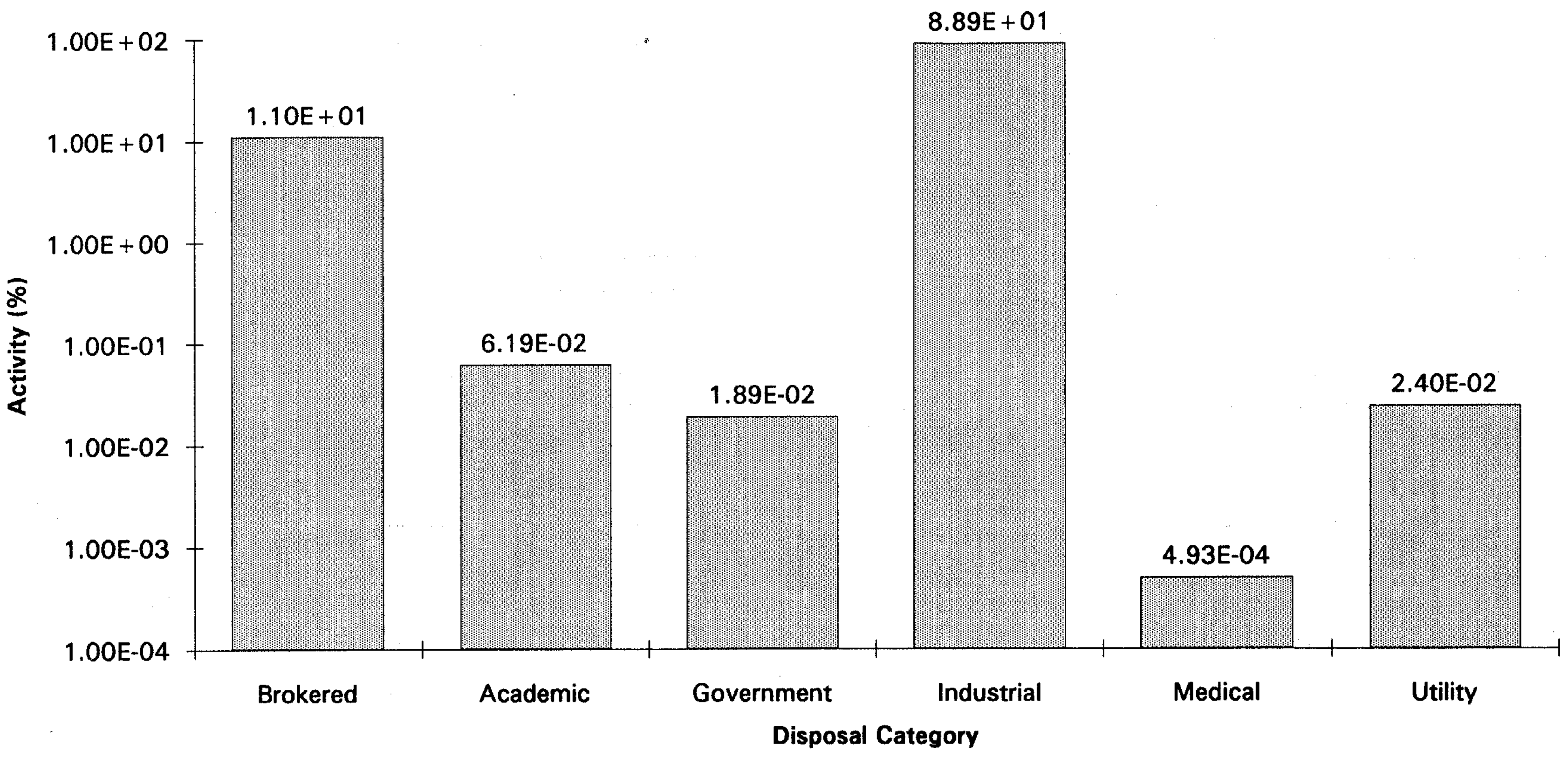

Figure 3. Uranium-238 deposited in the Richland Low-Level Waste Disposal Site as a function of disposal category (log scale).

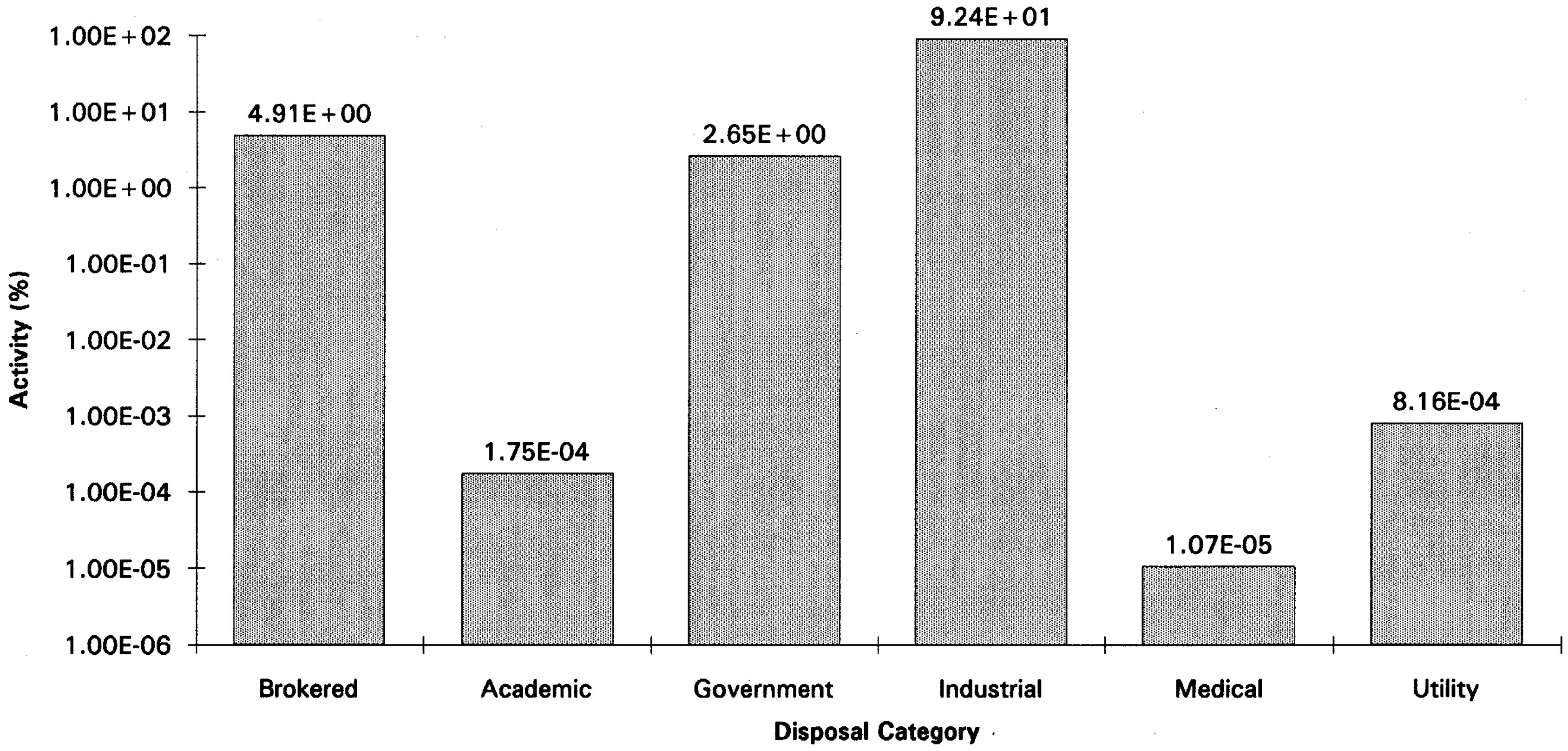

Figure 4. Uranium-238 deposited in the Barnwell Low-Level Waste Disposal Site as a function of disposal category (log scale).

Medical, Academic Institutions, and Commercial Waste

There are no known medical uses for ^{238}U, either for diagnosis or treatment of diseases (this was confirmed by a communication with personnel at the American College of Radiologists).[c]

There are, however, several commercial and military uses for this nuclide. These arise due to its high density, abundance, relatively low cost, and pyrophoricity. There are four principal nonnuclear uses of depleted uranium: radiation shielding; counterweights in airplanes, helicopters, and missiles; armor (for example, on tanks); and armor-piercing projectiles for military ordinance.[15]

Shipping containers made of depleted uranium are used as spent fuel casks for transporting highly radioactive spent fuel elements from commercial reactors to disposal sites. These casks are heavy (up to several metric tons) and have to withstand and dissipate the heat generated by the decaying fission products. These casks are usually clad with stainless steel to minimize corrosion and contamination. Other containers are used to ship radioactive isotopes for medical and industrial applications. These are much smaller and lighter. In these applications, uranium can be used as shielding instead of lead (which is much more bulky due to its smaller density) or tungsten (which is harder and more expensive to fabricate). In fact, uranium can be used as both the shielding and structural material, greatly reducing the size and improving the mobility of these containers. Additionally, depleted uranium is extensively used to house and shield industrial radioactive sources such as iridium-192, cobalt-60, and cesium-137. In these applications, the shielding can either contain an aperture to collimate a beam of gamma-rays or be used as a container out of which the source is removed for irradiation purposes.

Depleted uranium is used as counterweights in airplanes, missiles, and helicopters to maintain the center of gravity when these devices are moved. These counterweights need to be small and shaped to fit in spaces where they will not be in the way of the normal payload. Depleted uranium is ideal because is it dense (minimizing the size) and can be machined into various shapes to fit in unusual spaces. As an example, counterweights of 1,500 kg depleted uranium are used in all Boeing 747 airplanes.

The largest single use of depleted uranium is kinetic-energy projectiles for the military. Again, the high density of this element provides excellent penetrability, and this with its pyrophoricity, ease of fabrication, abundant availability, and low cost, make this material ideal for this application. Each of the three military services (army, air force, and navy) has at least one specific weapons platform that uses a depleted uranium projectile.

Other uses for deleted uranium include sinker bars for inserting instruments into oil wells, special high-performance gyroscope rotors, flywheels for large inertial energy-storage devices, vibration damping bars, and tritium storage beds.

Calibrated sealed sources of ^{238}U are available in a variety of strengths. Reference 20 lists several, primarily European, suppliers of radioactive sources including Czechoslovakian and English

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c. Private communication between C. Sperry, American College of Radiologists, and J. P. Adams Idaho National Engineering Laboratory, May 15, 1995.

sources of ^{238}U. The Czechoslovakian sources range from 8.3×10^{-5} to 3.3×10^{-3} μCi sources with aluminum backing. The English sources range from 10^{-3} to 10^{5} μCi with either thin foil or copper backing. Both of these suppliers indicate the use of these sources for alpha particles. The U.S. Department of Energy (DOE) is also a supplier of calibrated ^{238}U sources, with isotopic enrichment >99.9% and ^{235}U content less than 500 ppm.[21] The source strength varies according to order and the price depends on the ^{235}U contamination level.

BEHAVIOR OF URANIUM-238 IN THE ENVIRONMENT

Uranium in Soils

Understanding the behavior of radioactive materials, such as ^{238}U, in the soil is important in assessing the possibility of transport of these radionuclides through the biosphere. It is important to understand the interactions between radionuclides and various media along the path to the biosphere, whether disposal is in deep or shallow rock caverns or in shallow overburden facilities.

Four parameters were reported to be essential in accurately predicting soil concentrations from either contaminated groundwater or irrigation water.[22] The four parameters, in order of decreasing importance, are (a) soil retention, (b) annual precipitation, (c) soil texture, and (d) depth to the water table. The soil retention parameter represents the solid/liquid partition coefficient and is denoted by the symbol K_d. K_d is defined by $K_d = C_s/C_l$, where C_s is the uranium concentration in the soil (μg [U]/gram of soil) and C_l is the uranium concentration in the groundwater (μg [U]/mL of liquid). Therefore, the units of K_d are mL/g (i.e., mL of water per gram of soil). This empirical model combines all soil retention mechanisms into a simple linear partition relation between the soil and surrounding groundwater. This model assumes that the uranium concentrations in groundwater and soil are in equilibrium with each other. Using this definition for K_d, it follows that the larger the value (i.e., the higher the radionuclide concentration in the soil relative to that in the groundwater) the slower the migration of the radionuclide relative to groundwater flow. Therefore, K_d can be thought of as a measure of the amount of "fixing" or holdup in the soil.

Typically, K_d is measured under laboratory conditions, using samples that are relatively homogenous, and where it can be ensured that equilibrium conditions are met. Applying these values to specific soils in the field can be difficult because actual soils are generally inhomogeneous and there are uncertainties as to how long it takes for the uranium concentrations in the soil and groundwater (C_s and C_l) to reach equilibrium with each other. Therefore, one must be careful to ensure that soil samples used in laboratory studies are as closely representative of the field as possible. Even though the soil samples do not precisely match conditions in the field, K_d values from laboratory studies can be used in computer models that extrapolate data from laboratory experiments and field studies. In addition, some experiments are conducted using intact field samples to validate the laboratory K_d values and to study the effects associated with soil inhomogeneity.

Reference 23 lists ranges of K_d values for various elements in soils and clays. For uranium, this range is from 11 to 4,400 mL/g, which is a large range. Although no explanation is given for this large range, one possibility is the range of soil pH since it is known that the solubility of uranium is strongly dependent on this parameter. This reference also lists native soil concentration ranges for several elements. The typical concentration range for uranium is 0.9 to 9.0 ppm and the extreme range is up to 250 ppm.

A study was made of the ^{238}U content in soils in Venezuela and adjacent islands by scientists of the Instituto Venezolano de Investigaciones Cientificas.[24] This study was conducted to determine the cumulative fallout from nuclear weapons tests and nuclear accidents as well as the natural radioactivity from primordial radionuclides. For the case of ^{238}U, the manmade and primordial sources could not be differentiated. Samples were taken from a large number of different locations and at several depths in order to obtain the best information on the distribution of these nuclides. The ^{238}U concentrations ranged from less than 1 to 64 Bq/kg (1.7 nCi/kg) of soil with a mean concentration of 27 ± 16 Bq/kg (0.73 ± 0.43 nCi/kg). This mean translated to 2 ppm of uranium. Concentrations showed a strong dependence on altitude and proximity to the sea. Values for high altitudes and large (<1 km) distances from the sea were about one-quarter of those for low altitudes and closer distances from the sea, probably due to different ratios of sand and clay minerals in the soils. In general, the external gamma-ray dose due to these nuclides (excepting ^{137}Cs) was comparable to that due to cosmic radiation.

A similar study was performed by scientists of the Okayama Prefectural Institute for Environmental Science and Public Health in Japan.[25] Samples representing four weathering conditions were taken in the vicinity of Kamisaibara village near Okayama, Japan. Samples were taken of river bed soil, paddy field soil, field soil, and uncropped soil. The mean activity concentration of ^{238}U ranged from 0.028 to 0.041 Bq/g (0.76 to 1.1 pCi/g). The concentrations in river bed soil and field soil were similar, approximately 0.028 Bq/g (0.76 pCi/g) but the concentrations in paddy field soil and in uncropped soil were significantly higher, approximately 0.04 Bq/g (1.1 pCi/g). Although a type of phosphatic manure (containing high concentrations of uranium) is used as fertilizer, it is unclear how this affected the soil concentrations.

Phosphate ores contain concentrations of uranium that are significantly higher than average soil concentrations. Since phosphate slag is sometimes used as road bed material, scientists from the University of Idaho decided to measure uranium concentrations on and near some of these roads.[26] Soil samples were taken at three depths (up to 15 cm) and five distances from the roadbeds (up to 12 m) at 12 different locations and the distribution was mapped. Surface samples averaged from 3.5 to 9 ppm, depending on distance from the road (higher concentrations were from samples taken near the road and lower concentrations, from samples taken further away). There was little difference between 0–10 cm depth and 10–15 depth samples, with concentrations ranging from 3.6 to 6.8 ppm, depending on the distance from the road.

Scientists from the Institute of Hygiene and Public Health, Bucharest, Romania, measured the concentrations of uranium in soil samples from areas close to and far from high natural uranium concentrations.[27] The measured average concentrations of uranium were 386 ppm and 62 ppm for the close and far away locations, respectively. This study was conducted to determine the uptake into the food chain and, ultimately, into the human population.

Members of the Radiology and Radiation Biology Department at Colorado State University measured uranium concentrations in soil samples and sediment from a tailings pond near a uranium mine and mill as part of a study of the uptake into the food chain.[28] The uranium mine and mill operation that was studied is located in the southeastern plains of Wyoming at an elevation of 2,200 m. This is a semiarid environment with about 25 cm annual precipitation that is dominated by grasses, forbs, and sagebrush. The uranium ore is approximately 100 m below the surface and mining operations have been conducted since 1969. Soil concentrations of ^{238}U in samples taken several meters downwind from the tailings ranged from 200 to 363 mBq/g (5.4 to 9.8 nCi/g). The sediment had a mean uranium concentration of 0.4 Bq/g (11 pCi/g) and a K_d of 1.0 (Bq/g)/(Bq/mL). The relatively low value for K_d is believed to be a function of the high water solubility of the uranium compound (probably a sulfate due to the acid leaching process). The pH values ranged from 2 to 5, indicating the presence of residual sulfuric acid from the leaching process.

In summary, ^{238}U, as the overwhelming isotope of uranium, is naturally present in soils with concentrations in the few parts per million range. In addition, this nuclide can be present due to human activities. In the presence of acidic soil conditions (for example, in uranium tailings), uranium is very soluble in water and is not readily adsorbed in the soil.

Uranium in Water

The natural concentration of uranium in groundwater is listed in Reference 21, along with natural concentrations of several other elements and compounds. The natural concentration of uranium ranges from 0.1 to 40 parts per billion (ppb).

The migration of ^{238}U in the groundwater of sandstone aquifers was studied by scientists in Germany.[29] Based on their study, the retardation factor (defined in the reference) was measured to be 8.6×10^6 and the removal rate was measured to be 1.3×10^{-3} yr^{-1} for a Triassic sandstone aquifer in the German Democratic Republic.

The concentration of ^{238}U in the water and suspended sediments of the Miami River (near the Mound Laboratory in Ohio) were measured in a study conducted by scientists of the Argonne National Laboratory.[30] The average concentration of ^{238}U in the river water was 0.59 pCi/L, which is approximately six times the average concentration of this nuclide in Lake Michigan. The distribution coefficient of this nuclide between the suspended solids and the surrounding water was 1,800.

A study was made of the activity of certain actinides and beta emitters in the process water used in the Loviisa VVER-440-design nuclear power plant in Finland.[31] The process waters and the ion exchange resins used for purification of the process waters were sampled and analyzed for ^{238}U concentration. The coolant and ion exchange resin samples from the primary circuit represent the normal operation state and a fuel leakage situation. ^{238}U was found in several of the anion and cation exchange resins, both from the column and in the waste tanks. This nuclide was also found in primary coolant samples, both in plants with and without known fuel failures. The concentrations of this nuclide in ion exchange resin samples were all <8 kBq/m^3 (<0.22 μCi/m^3). Though this nuclide was measured in coolant samples, the concentrations were small, resulting in large statistical uncertainties.

An environmental study was made of the radionuclides in the effluent from the Argonne National Laboratory (ANL) that was discharged into streams in the vicinity.[32] The concentration of ^{238}U in the Sawmill Creek ranged from 0.5 to 2.0 μCi/mL, with an average concentration of 1.1 μCi/mL. The concentration of ^{238}U in the Des Plaines River ranged from 0.5 to 2.0 μCi/mL, with an average concentration of 1.3 μCi/mL. The concentrations of this nuclide in domestic wells at the ANL were also measured and ranged from 0.05 to 0.14×10^{-9} μCi/mL. This study contains a great deal of additional information on this nuclide plus many other nuclides included in the ANL measurement plan. Based on the measured activities of ^{238}U, as well as the activities of numerous other radioactive nuclides at the ANL, it was determined that the maximum dose to any member of the public did not exceed 0.1 mrem/yr, or 0.02% of the 500 mrem/yr standard.

The effects of an uranium mining facility on the concentration of uranium in the aquatic food chain were studied by scientists of the Jozef Stefan Institute of the E. Kardelj University in

Yugoslavia.[33] In this study, surface waters, river and stream sediments, and biological samples were taken from several sampling points in the vicinity of the uranium mine at Zirovski and the Nuclear Power Plant at Krsko, both in Slovenia. The concentrations of ^{238}U ranged from 0.11 to 12.5 mg/m^3 and in the corresponding sediment, from 2.04 to 9.83 mg/kg. Accumulation factors ranged from 540 to 30,000 for the same set of samples.

Reference 25 also included measurements of the concentrations of uranium in groundwaters near and far from locations of high uranium concentrations. The average uranium concentrations were 166 and 6.7 μg/L for the near and far away locations, respectively.

The water concentrations used in the Reference 26 study were measured to be 0.5 Bq/mL (0.1 pCi/mL). As stated earlier, this resulted in a partitioning between soil and water with a $K_d = 1.0$.

A researcher at the Pacific Northwest Laboratory conducted a study into the waterborne release of uranium during the enrichment process at the Oak Ridge (Tennessee), Paducah (Kentucky), and Portsmouth (Ohio) facilities.[34] The reported water releases ranged from 12–26 GBq (0.32–0.70 Ci) from Oak Ridge, 1.5–25 GBq (0.041–0.68 Ci) from Paducah, and 11–29 GBq (-.30–0.78 Ci) for Portsmouth. The time frame for these releases was from 1982 to 1986.

Researchers at the DOE Environmental Measurements Laboratory conducted a study into the daily intake of various radioactive nuclides (including ^{238}U) in New York.[35] The reason for this study was that, although it was recognized that uptake of naturally occurring radionuclides represents a major source of radiation exposure to humans, fundamental information was lacking concerning the chronic intake of these radionuclides by humans. Therefore, this study included measurements of the concentration of ^{238}U in city tap water. Typical concentrations ranged from 0.67–1.1 mBq/L (18–30 fCi/L).

In summary, uranium is naturally present in groundwater with concentrations in the parts per billion range. Concentrations may be much larger near uranium processing facilities, such as mines and fuel reprocessing plants. Concentrations near such facilities can be in the microcurie per millimeter range.

Uranium in Plants

Scientists at the Oak Ridge National Laboratory measured the accumulation of various actinides, including ^{238}U, by plants in a contaminated fresh-water pond.[36] This particular pond, Pond 3513, was used for receiving low-level radioactive waste for a period of 30 years, ending in 1976. ^{238}U was measured in grass that grew on the shore of the pond. It was determined that of the four elements studied (with multiple radioactive isotopes for some of the elements), uranium resulted in the highest relative uptake.

Reference 25 included measurements of the concentration of uranium in vegetation growing in locations both near to and far from areas of high uranium backgrounds. The vegetation used in this study was referred to as "fodder vegetation," presumedly plants used to feed livestock, and the samples were first turned into ash prior to measuring the uranium content. The average concentrations of uranium were 78 and 9 ppm for the plant material for samples taken near to and far away from the areas of high uranium background, respectively.

Reference 26 concluded their study with measurements of the plant uptake from contaminated soils. The vegetation collected and measured in this study were mixed grasses, mixed forbs, and big sagebrush. The uranium concentrations ranged from 5.6–407 mBq/g (0.15–11 pCi/g), depending on location. This compared to a value of 4.4 mBq/g (0.12 pCi/g) for a background sample. The concentration ratio (ratio of vegetation concentration to soil concentration of uranium) ranged from 0.04 to 0.8, with no apparent correlation between soil concentration and concentration ratio.

Scientists from the Pacific Northwest Laboratory conducted a survey of vegetation along the stretch of the Columbia River near the DOE's Hanford Site.[37] The shoreline vegetation at this location consists of a narrow zone of broad-leafed deciduous trees (e.g., the mulberry, which provides food for mule deer as well as other mammals) and shrubs (e.g., willows) intermingled with perennial grasses (e.g., reed canary grass, which provides food for wild Canada geese) and forbs. These plants, which provide indirect food for humans via wild animals, remain green despite the dry, desert-like environment in the summer because of their access to the river. This analysis did not result in unusually high levels of ^{238}U. The highest concentrations of uranium in vegetation were found in onions, and even there, there were no significant differences in concentrations between those onions located near the Hanford Site and those harvested upstream of the site. Measured concentrations were of the order of picocuries per gram dry weight.

Researchers at the Environmental Measurements Laboratory of the DOE conducted a study into the daily intake of various radioactive nuclides (including ^{238}U) in New York.[35] Therefore, in addition to water concentrations, this study also included measurements of the amount of ^{238}U ingested by a typical New Yorker in various food products. The amounts of ^{238}U ranged from 9 mBq/yr (0.2 pCi/yr) for rice to 1,935 mBq/yr (52 pCi/yr) for shellfish.

In summary, uranium is taken up into plants relatively easily. Typical uranium concentrations in plants are in the picocurie per gram range. Concentration ratios, the ratio of uranium concentration in a plant to the concentration in the surrounding soil, are generally less than 1.0.

Uranium in Air

From a biological hazard standpoint, one of the most important pathways for actinides to contaminate humans is inhalation. For this reason, it is important to understand how uranium can become suspended in air where it can be inhaled. Reference 26 included a study of the possibility that radioactive dust particles could be abraded from road surfaces (constructed using phosphate slag, which contains relatively high concentrations of uranium) and become suspended in air as aerosols. To do this, aerosols were collected at several sites by pumping air through filters and differentiating between aerosol sizes. Based on the belief that the critical aerosol size for retention in the lungs was 0.01 to 5.0 μm, this was the aerosol size the authors concentrated on in their study. They determined that the concentration of ^{238}U in aerosols of this size was less than 0.03 μg/m^3 of air. The authors concluded that this was not sufficient to cause serious concern about the use of phosphate slag in road construction.

Researchers at the Pacific Northwest Laboratory also conducted a study into the airborne release of uranium during the enrichment process at the Oak Ridge, Paducah, and Portsmouth facilities.[34] The reported atmospheric releases ranged from <0.07–0.37 GBq (4–10 mCi) from Oak Ridge, 0.07–4.8 GBq (2–130 mCi) from Paducah, and 1.4–3.3 GBq (38–89 mCi) for Portsmouth. The time frame for these releases was from 1982 to 1986.

Researchers at the Environmental Measurements Laboratory of the DOE conducted a study into the daily intake of various radioactive nuclides (including ^{238}U) in New York.[35] Therefore, in addition to water and food concentrations (discussed earlier), this study also included measurements of the amount of ^{238}U in New York air. The mean concentration in the air was 0.94 μBq/m^3 (0.03 fCi/m^3).

A similar study was performed by scientists at the Okayama Prefectural Institute for Environmental Science and Public Health for the area north of the city of Okayama.[38] Specifically, these scientists assumed that a principal source of uranium-laden aerosols is soil erosion caused by wind. To maintain scientific integrity, they maintained, as closely as possible, similar sampling conditions between sampling times, including rainfall, snowfall, wind speed, and direction. Typical activity concentrations for ^{238}U ranged from 0.5–9.3 μBq/m^3 (0.01–0.3 fCi/m^3) with a geometric mean of 2.2 μBq/m^3 (0.06 fCi/m^3).

In summary, inhalation of ^{238}U is one of the most important pathways of exposure to this nuclide. Uranium can be suspended in air either due to leakage of volatile uranium (e.g., UF_6) or by weathering of uranium tailings and wind-driven suspension. Examples of the latter mechanism can result in air concentrations of the order of microbecquerel per cubic meter.

BEHAVIOR OF URANIUM-238 IN THE HUMAN BODY AND IN ANIMALS

This section summarizes several studies of the behavior of ^{238}U in the human body and in animals. Since ^{238}U is principally an alpha emitter, the human hazard from this nuclide is dominated by internal dose when the nuclide is either ingested or inhaled. Both exposure pathways are important in understanding the hazard this nuclide presents to humans. This nuclide can become airborne either by atmospheric release (e.g., from an enrichment facility if a leak in the volatile UF_6 line were to occur) or by weathering and resuspension due to wind (e.g., at a uranium tailings site). Ingestion can occur when plants take up the nuclide from contaminated soil and then are themselves either directly ingested by humans or first ingested by animals, which are in turn eaten by humans. A number of factors may influence the pathway to humans, including chemical form of the nuclide, particle size distribution (for the case of aerosols), and water and/or plant concentration effects. Where the specific studies included a discussion of these effects, they are also summarized.

Scientists at the Institute of Biophysics, Ministry of Public Health in Moscow, Russia, conducted a study into the effects of various chelating agents on the removal of various radionuclides that had been ingested by dogs and rats.[39] This work was performed to determine how effective these agents are in reducing the dose due to these nuclides on the human body. A number of various agents were studied, all related to polyaminepolyalkylphosphonic acids, as possible substitutes for other agents that, while successful in removing the radionuclides, are harmful to animals in general and the kidneys in particular. All of the compounds were successful in reducing uranium from the contaminated animals, reducing the skeleton burden by 50–84% and the kidney burden by 82–95%, compared to a control group.

The study, documented in Reference 35, was performed to determine the extent to which radioactive contamination in a radioactive waste settling basin could enter the food chain. Both invertebrates (dragonfly nymphs, water bugs, and snails) and vertebrates (goldfish, catfish, and tadpoles) were included in this study. In addition, rats that populate the shoreline were also trapped and studied. This study concluded that the overall comparative uptake of actinides by these animals indicated that uranium is the most easily absorbed actinide.

A similar study was performed for the area near a uranium mine in Solvenia.[33] In this study, the uptake of uranium (that contaminated surface waters) into fish food and fish was measured. The uranium concentrations in the fish varied from 1.07–42.9 μg/kg, depending on the location relative to the mine. The ratio of uranium concentration in the fish (μg/kg) to that in the surrounding water (mg/m^3) varied from 0.7–38, with the lower ratios tending to be associated with smaller fish concentrations.

A study of the effects of inhaling uranium dust on the pulmonary function was conducted using dogs.[40] In this study, sets of young and old dogs were exposed to various lengths of time to 14 mg/m^3 ore dust. The pulmonary function (including effective lung volume) was measured on these dogs as well as control sets of dogs (also both old and young). This study concluded that there was a significant decrease in the uniformity of ventilation in the lung as a function of exposure to uranium ore dust, although the total lung volume showed no significant effects of either age or aerosol exposure.

The ^{238}U contamination in beavers, grouse, and muskrats was studied by a researcher at the Laurentian University for the Atomic Energy Control Board in Ottawa, Canada.[41] The location for the study was the Serpent River drainage basin, which is near a uranium tailings site at Elliot Lake. Several radionuclides were included in the study, in addition to ^{238}U. ^{238}U was detected in measurable amounts in the bone, muscle, and liver issues of both beavers and grouse. Concentration ratios (defined as the ration of ^{238}U concentration in an animal's diet to that in a specific tissue) were measured. This ratio exceeded unity between some vegetation items and beaver bone (concentration ratios were less than zero for other beaver tissues) at the Elliot Lake tailings site and between aspen leaves and grouse bones. All other ratios were less than 1. Estimated yearly intakes of radionuclides by people eating beaver and grouse were calculated to be below current limits established by regulating authorities in Canada.

Reference 34 cumulated the study by calculating the estimated dose to the public due to exposure to operations of the DOE's three uranium enrichment facilities. This study did not differentiate between the two principal isotopes of uranium, ^{235}U and ^{238}U, in presenting the results of the dose calculation. In addition, the effects of the long-lived radionuclide ^{99}Tc, which is also present in significant quantities in these facilities and which, along with the uranium isotopes, represent the major radioactive hazards due to this type of operation, are included in the dose calculations. Therefore, the doses represent upper bounds on the doses due to ^{238}U alone. The doses account for inhalation (due to airborne effluents, both gaseous or particulate) and ingestion (due to inadvertent release of radioactive effluent from maintenance and decontamination processes). The measured atmospheric and surface water releases were discussed earlier in this report. The total dose to the public due to operation of these three enrichment facilities during the years from 1982 to 1986 were (1) for the Oak Ridge, Tennessee, facility, the dose ranged from 9×10^{-6} - $<8 \times 10^{-5}$ man-Sv, (2) for the Paducah, Kentucky, facility, the dose ranged from 2×10^{-4} - 5×10^{-2} man-Sv, (3) and for the Portsmouth, Ohio, facility, the dose ranged from 2×0^{-4} - 0.1 man-Sv.

The purpose of the study documented in Reference 27 was to investigate the contamination of a population in Romania due to living near an area of elevated uranium deposits. Owing to the specific local ecological processes, natural uranium is expected to be taken up into the members of a local population, which are therefore exposed to an increased level of natural irradiation. This was a group of unoccupationally exposed people, as contrasted with many other studies that are restricted to occupationally exposed subjects, such as uranium mine workers. Urine samples were collected from people living near an area of increased natural uranium deposits as well as a control group of people living away from this area. There were 70 people in the first group and 30 in the second group. In the exposed group, the uranium concentrations ranged from 0.8–94.5 μg/L with an average value of 32.5 μg/L. For the control group, the uranium concentrations ranged from 0.8–26.2 μg/L with an average value of 6.7 μg/L. Thus, the exposed group experienced significantly higher exposures to uranium than did the control group.

The principal excretion paths for ^{238}U in humans was studied by scientists at the Hines Veterans Administration Hospital in Illinois.[42] Four patients in the hospital were included in the study. Their diets were strictly controlled so that an accurate uptake could be measured. The uranium ingested by these patients was of natural origin in the diet, including drinking water and milk. Although the diets for these patients was controlled during the study, prior ingestion of uranium could not be quantified. It was determined that the principal excretion path was via the intestine and that elimination via the urinary tract contributed less than 2% of the total excretion. The total average daily intake of ^{238}U

ranged from 24–46 mBq/d (0.7 to 1 pCi/d) for these patients. During the study, an average of approximately 70% of the intake of ^{238}U was excreted.

The concentrations of various nuclides were measured, including ^{238}U, in the bones and soft tissues of uranium miners, millers, and the general population in the state of Colorado.[43] In this study, samples of lung, lymph node, liver, kidney, spleen, testis, heart, and vertebrae were collected from three uranium miners, two millers, and 15 members of the general population by autopsy and the ^{238}U concentrations were measured for each sample. Of the total body content, the skeletal burdens of this nuclide ranged from 13–79% for the miners, 9–81% for millers, and approximately 85% for the general population. In general, the skeleton accounted for 83–99% of the uranium content in all three groups. It is difficult to determine from this report if there were any other general trends, although it appeared that the heart could also concentrate this nuclide.

Researchers at the DOE Environmental Measurements Laboratory have examined bone samples from Nepal and Australia for evidence of natural uranium and fallout plutonium.[44] The concentration of ^{238}U ranged from 4 to 14 pCi/kg of ash for the remains from Nepal and from 0.2 to 3.5 pCi/kg for the remains from Australia. The samples were from people who died in the 1970s and were from people who lived up to 82 years in Nepal and 34 years in Australia. Bone ash was used because of the belief that natural uranium resides primarily in the skeleton. There was no substantial increase in natural uranium content with age, indicating that equilibrium in the body is established quickly, inferring a short biological half-life. The average dose to the bone (due to alpha radiation) was 0.039 mrad/yr in Nepal and 0.009 mrad/yr in Australia.

Scientists at the University of Utah's Department of Pharmacology studied the concentrations of various radionuclides in the lungs of uranium miners.[45] Lung tissue samples were collected from the remains of eight miners and the ^{238}U concentrations were measured. These concentrations ranged from 0.6–11.5 Bq/kg (16–310 pCi/kg). The lung doses were calculated to range from 0.01 to 0.72 mGy/yr (this included the effects of ^{234}U and ^{230}Th as well as ^{238}U).

A study was made of the inhalation of uranium due to working in an ore crushing facility.[46] Both urine and fecal samples were obtained from active crushermen (men who work to crush uranium ore prior to its being processed to remove the uranium), retired crusher workers (retired at least 4 years), and control individuals who live near but who do not work at the mills. The mills studied were judged to be among the dustiest in the industry and the workers normally do not wear respirators during operations. Therefore, it was judged that these results would represent some of the highest exposures currently to be found. The urine concentrations of ^{238}U ranged from 0.2–11 pCi/L for active workers, 0.04–0.19 pCi/L for retired workers, and 0.04–0.2 pCi/L for the control group. As seen, there was little difference between the retired workers and the control group, indicating that the biological half-life of uranium in the body is fairly short. The uranium levels in the feces ranged from 2–1,930 pCi/L for active workers, from 2–73 pCi/L for retired workers, and from 5–15 pCi/L for the control group. Again, the contamination levels for the retired workers were reasonably similar to those for the control group. In addition, it is evident that the feces represents the principal removal path for this nuclide.

The dose that is received due to intake (either inhalation or ingestion) of a radionuclide is a strong function of the rate at which this nuclide either radioactively decays or is biologically removed from the body. For the case of ^{238}U, the biological half-life (time for the body to biologically remove one half of the uranium burden in the body) is 300 days for contamination of the bones, 15 days for contamination of the kidneys, and 100 days for general contamination of the body.[28] This reference also lists the organs most likely to be affected by intake of various radionuclides. Per this reference, ^{238}U tends to become concentrated in the lungs and kidneys and not in the liver or bones (although another isotope of uranium, ^{233}U is concentrated in the bones). No reason was given for the apparent discrepancies (e.g., biological half-life given for bones and the fact that ^{233}U but not ^{238}U, tends to concentrate in the bones).

Reference 47 lists the organ dose that results as a result of a specific intake of ^{238}U. For exposure due to inhalation (either of a gaseous uranium compound or an aerosol), the lungs receive the largest dose, followed by the bone surface, and bone marrow—the remaining organs receive little dose. For exposure due to ingestion, the bone surface receives the largest dose, followed by the bone marrow—the remaining organs receive little dose. This reference also lists the dose rate factors for external radiation due to ^{238}U. Both for immersion in contaminated air and for ground-surface exposure, these factors are relatively small.

The annual limits on ^{238}U intake (ALI) and the derived air concentration (DAC) for ^{238}U are shown in Table 5. The ALI is defined as that annual intake of a radionuclide that would result in a radiation dose to a human (for calculational purposes, the characteristics of the "Reference Man" are used) equal to the allowable limit. The DAC is defined as that concentration of a radionuclide in air which, if breathed for a work-year, would result in an intake corresponding to the ALI (or, in the case of submersion, to an external exposure corresponding to the primary guide for limiting annual dose). DACs are, thus, used for limiting radionuclide intake through the breathing of, or submersion in, contaminated air. ALIs are used primarily for assessing doses due to accidental ingestion of radionuclides.

Historically, maximum permissible concentrations (MPC) in air and water have been used to determine safety guidelines for released radionuclide concentrations. Currently, the derived guidelines are presented in terms of ALIs for inhalation or ingestion and DACs for inhalation (or submersion). For a radionuclide whose derived value does not change from the old definition, the DAC is numerically equal to the MPC value in air. The information in Table 6 applies only to ^{238}U. In the case of multiple radionuclides released in a mixture, additional guidelines outlined in the Code of Federal Regulations, Part 20 of Title 10[48] must be followed.

In summary, due to its long half-life, ^{238}U is removed by the body exclusively via biological means with excretion via the gastrointestinal tract the principal removal mechanism. This nuclide can be taken up by humans by eating contaminated meat or by inhaling contaminated air. Usually, contamination levels are low. The bone surface appears to be the organ most likely to receive long-term dose, although the lungs can also receive a dose when the exposure is due to inhalation.

Table 5. Annual limits on intake (ALI) and the derived air concentrations (DAC) for ^{238}U (data obtained from Reference 49).

Radionuclide	Component	Ingestion	Inhalation[a]
^{238}U	ALI (Bq)	5×10^5 (5×10^{-2})[b] 8×10^5 (5×10^{-2}) (bone surface) 8×10^6 (2×10^{-3})	50,000/D 80,000/D (bone surface) 30,000/W 2,000/Y
	DAC (Bq/m^3)	—[c]	40,000/D 30,000/W 20,000/Y

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a. Clearance from the lung directly to the blood stream or to the gastrointestinal tract depends on the chemical form of the radionuclide and is classified as D, W, and Y, respectively, for clearance times of the order of days, weeks, and years. For ^{238}U, the classification is W.

b. The removal class, f_1 for each ALI is in parenthesis. A simple model of the lung was used to describe the translocation and retention of material by the body after inhalation. In this model, 25% of the inhaled activity was exhaled and 25% was deposited in the lower respiratory tract. The 50% that was deposited in the upper respiratory tract was eventually cleared by means of mucociliary processes and swallowed. What happens then depends on whether the inhaled material was in either a soluble or insoluble chemical form. Any soluble material deposited in the lower respiratory tract is assumed to be transferred directly to the blood stream. The insoluble activity is cleared from the upper respiratory tract and then swallowed; a fraction (f_1) enters the blood stream via the gastrointestinal tract. In the case of ^{238}U, $f_1 = 0.05$ for water-soluble inorganic compounds of uranium and 0.002 for relatively insoluble compounds such as UF_4, UO_2, and U_3O_8. Additional details of this model are described in References 49 and 50.

c. Data not available.

CONCLUSIONS

^{238}U is a naturally occurring, radioactive nuclide with a half-life of 4.47×10^9 years. This nuclide decays, by alpha particle emission, to ^{234}Th. It may also spontaneously fission, creating two smaller-massed nuclides plus neutrons, although the probability of this occurring is $<1\%$. After a combination of 14 alpha and beta decays, the stable ^{206}Pb nuclide finally results. Moderately high energy alpha particles, low energy gamma-rays, and low energy beta particles are emitted when this nuclide decays. The total particle and photon spectrum is complex because of the complex decay scheme.

Because of the long half-life of ^{238}U, it is not, itself, a major factor in the ability of a low-level radioactive waste disposal facility to meet performance objectives. The presence of the nuclide, however, can be relatively significant in assessing the long-term performance of such a site due to the quantity, radiotoxicity, and mobility of its daughter products, which include isotopes of radium and radon.

^{238}U is the most abundant isotope of uranium, comprising $>99\%$ of all naturally occurring uranium. Another isotope of this element, ^{235}U, is used widely in energy and weapons production, and ^{238}U is a byproduct of this activity. Therefore, this nuclide is equated with the term "depleted uranium." As such, this nuclide has widespread industrial and military usage, including radioactive shielding, counterweights, and nonnuclear weapons. In addition, materials containing relatively high concentrations of uranium have been used as roadbed material. This nuclide is also available as a radioactive sealed source, although no medical uses were identified.

Uranium is very reactive and forms compounds with many other elements, such as the halides, oxygen, and hydrogen. Its chemistry is dominated by the half-filled 5f electron subshell. The U-O system is one of the most complex of all oxide systems. Uranium is similar to neodymium, a lanthanide with a half-filled 4f electron subshell.

The ability of soil to adsorb ^{238}U out of the groundwater depends on a number of factors, including pH and the presence or absence of complexing agents. In the presence of low pH soil conditions (for example, in uranium tailings that were processed using sulfuric acid), uranium is very soluble and tends to remain in the groundwater rather than being adsorbed by the soil.

The principal radiological hazard associated with this nuclide is due to its relatively high energy alpha particles. Since these alpha particles do not penetrate materials easily, external exposure to this nuclide does not pose a high risk. The principal risk is due to either inhalation or ingestion. Inhalation occurs either from release of a volatile uranium compound (such as UF_6, which is used in uranium enrichment processes) or from suspension of uranium-laden aerosols (such as due to weathering of tailings or ore). Ingestion can occur when the uranium is introduced into the food chain by plant uptake. When ^{238}U is either ingested or inhaled, it is removed from the body with a biological half-life of a few hundred days, depending on which organ has become contaminated by the nuclide. This nuclide tends to concentrate in and irradiate the bones. Additionally, if inhaled, the lungs can receive a dose.

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