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RADIATION HAZARDS TO FISH, WILDLIFE, AND INVERTEBRATES: A SYNOPTIC REVIEW

by
Ronald Eisler

Patuxent Environmental Science Center
U.S. National Biological Service
Laurel, MD 20708

TABLES

FIGURES

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Abstract

This account is a selective review and synthesis of the voluminous technical literature on radiation and radionuclides in the environment and their effects on notably fishes, wildlife, invertebrates, and other natural resources. The subtopics include the physical and biological properties of the electromagnetic spectrum and of charged particles; radiation sources and uses; concentrations of radionuclides in field collections of abiotic materials and living organisms; lethal and sublethal effects, including effects on survival, growth, reproduction, behavior, metabolism, carcinogenicity, and mutagenicity; a synopsis of two case histories of massive releases of radionuclides into the biosphere (military weapons tests at the Pacific Proving Grounds and the Chernobyl nuclear reactor accident); currently proposed radiological criteria for the protection of human health and natural resources; and recommendations for additional research. A glossary is included.

Key words: Radioactivity, radionuclides, radioecology, Chernobyl, Pacific Proving Grounds, wildlife, aquatic organisms, invertebrates, flora, radiological protection criteria.

Life on earth has evolved under the ubiquitous presence of environmental solar, X-ray, gamma, and charged-particle radiation. On a global basis, radiation from natural sources is a far more important contributor to radiation dose to living organisms than radiation from anthropogenic sources (Aarkrog 1990). However, ionizing radiation can harm biological systems (Aarkrog 1990; Nozaki 1991; Severa. and Bar 1991), and this harm can be expressed (1) in a range of syndromes from prompt lethality to reduced vigor, shortened life span, and diminished reproductive rate by the irradiated organism and (2) by the genetic transmission of radiation-altered genes that are most commonly recessive and almost always disadvantageous to their carriers (Bowen et al. 1971). Direct effects of radiation were documented in lampreys in 1896-soon after H. Becquerel discovered radioactivity-and in brine shrimp (*Artemia sp*.) in 1923 (Whicker and Schultz 1982a). Genetic effects of ionizing radiation and thus X-rays as a mutagenic agent were first documented in 1927 in fruit flies, *Drosophila melanogaster* (Evans 1990). The discovery of radioactivity of nuclear particles and the discovery of uranium fission resulted in a great upsurge of nuclear research. During and shortly after World War II, nuclear reactors, nuclear weapons, and radionuclides as tracers in almost all scientific and technical fields were developed rapidly (Severa and Bar 1991). In the early 1940's when fission of uranium and transuranic nuclei became possible in reactors and in explosions of nuclear weapons, environmental radiation from anthropogenic sources began to cause serious concerns (Aarkrog 1990). The first nuclear explosion resulted from a 19-kiloton (TNT-equivalent) source in New Mexico in July 1945 (Whicker and Schultz 1982a). On 6 August 1945, about 75,000 people were killed when the United States Army Air Corps dropped a uranium nuclear bomb on Hiroshima, Japan; on 9 August 1945, about 78,000 Japanese were killed and more than 100,000 injured when a plutonium nuclear bomb was detonated at Nagasaki (Kudo et al. 1991). The former Soviet Union detonated its first nuclear device in August 1949, and in 1952 the United Kingdom exploded a device in Australia (Whicker and Schultz 1982a). Since 1960, nuclear devices have also been detonated by France, India, and The People's Republic of China. Nuclear devices have been developed that can release energy in the megaton range. The first such device was detonated by the United States in 1954 at Bikini Atoll and accidentally contaminated Japanese fishermen and Marshall Island natives. Between 1945 and 1973, an estimated 963 nuclear tests were conducted by The People's Republic of China, France, the former Soviet Union, the United Kingdom, and the United States; 47% of them were atmospheric, and 53% subterraneous (Whicker and Schultz 1982a). Today, the most important environmentally damaging anthropogenic radiation comes from atmospheric testing of nuclear weapons that was conducted 20 to 30 years ago, authorized discharges to the sea from nuclear reprocessing plants, and the Chernobyl accident in 1986 (Aarkrog 1990). By the year 2000, the United States will have an estimated 40,000 tons of spent nuclear fuel that will be stored at some 70 sites and await disposal; by 2035, after all existing nuclear plants have completed 40 years of operation, about 85,000 metric tons will be awaiting disposal (Slovic et al. 1991).

This report was initiated in response to a request for information on radiation from environmental contaminant specialists of the U.S. Fish and Wildlife Service. Specifically, general information was requested on radiation nomenclature, sources and uses, fate, effects, concentrations in field collections, and protection criteria. More detailed information was requested on radiation hazards to living organisms, especially fishes and wildlife. The report is an introduction to the broader fields of radioecology and radiation risk assessment and is intended primarily for use by service personnel, I emphasize that the published literature in these subject areas is particularly voluminous and that I selected for synthesis a comparatively small and highly selective portion of the available information. For more detailed information on various aspects of radiation in the environment, readers are strongly advised to consult at least several of the many reviews [1] that I found particularly useful.

[1] National Academy of Sciences (NAS) 1957, 1971; Glasstone 1958; Schultz and Klement 1963; Nelson and Evans 1969; Nelson 1971; Polikarpov 1973; Cushing 1976; Nelson 1976; International Atomic Energy Agency (IAEA) 1976, 1992; International Commission on Radiological Protection (ICRP) 1977,1991a 1991b; Luckey 1980; Whicker and Schultz 1982a, 1982b; League of Women Voters (LWV) 1985; Hobbs and McClellan 1986; United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) 1988; Becker 1990; Kiefer 1990; Majumdar et al. 1990; Brisbin 1991; Kershaw and Woodhead 1991; Sankaranarayanan 1991a, 1991b, 1991c; National Council on Radiation Protection and Measurements (NCRP) 1991; Severa and Bar 1991.

Physical Properties of Radiation

General

Radiation is usually defined as the emission and propagation of energy through space in the form of waves and subatomic particles (Weast 1985; Kiefer 1990). For regulatory purposes in the United States, radiation is narrowly defined as α, β, , or X-rays; neutrons; and high-energy electrons, protons, or other atomic particles; but not radio-waves or visible, infrared, or ultraviolet light (U.S. Code of Federal Regulations [CFR] 1990). Readers may wish to consult the glossary at this time.

In current atomic theory, all elementary forms of matter consist of small units called *atoms*. All atoms of the same element have the same size and weight; atoms of different elements differ in size and weight. Atoms of the same element or of different elements may unite to form compound substances called *molecules*. Each atom consists of a central nucleus and several negatively charged electrons in a cloud around the nucleus. The nucleus is composed of positively charged particles called *protons*, and particles without charge are called *neutrons*. Electrons are arranged in successive energy levels around the nucleus, and the extranuclear electronic structure of the atom is characteristic of the element. Electrons in the inner shells are tightly bound to the nucleus but can be altered by high energy waves and particles (Weast 1985). Based on the number of protons in their nucleus (= the atomic number), atoms are classified chemically into 92 naturally occurring elements and another dozen or so artificial elements (Rose et al. 1990; Severa and Bar 1991). Atoms of the same element may occur as isotopes that differ in the number of neutrons that accompany the protons in the nucleus. The sum of the number of protons and neutrons in the nucleus is called the *mass number* (see Glossary) and is indicated by a superscript that precedes the chemical symbol of the element. For example, three isotopes of hydrogen (one proton) are denoted as ^{1}H (no neutrons), ^{2}H (1 neutron, also known as deuterium), and ^{3}H (2 neutrons, also known as tritium). A nuclide is an elemental form that is distinguished from others by its atomic and mass numbers. Some nuclides, such as ^{238}U and ^{137}Cs, are radioactive and spontaneously decay to a different nuclide with the emission of characteristic energy particles or electromagnetic waves; isomers of a given nuclide that differ in energy content are metastable (i.e., ^{115m}Cd) and characterized in part by the half-life of the isomer (Rose et al. 1990; Severa and Bar 1991).

Chemical forms with at least one radioactive atomic nucleus are radioactive substances. The capability of atomic nuclei to undergo spontaneous nuclear transformation is called *radioactivity*. Nuclear transformations are accompanied by emissions of nuclear radiation (Severa and Bar 1991). The average number of nuclei that disintegrates per unit time (= activity) is directly proportional to the total number of radioactive nuclei; the time for 50% of the original nuclei to disintegrate (= half-life or Tb ½) is equal to ln 2/decay constant for that element (Kiefer 1990). Radiations with sufficient energy to interact with matter to produce charged particles are called *ionizing radiations* (Hobbs and McClellan 1986; UNSCEAR 1988). Radiation injury is related to the production of ions inside the cell. Ionizing radiations include electromagnetic radiation such as gamma ( ) and X-rays and particulate or corpuscular radiation such as alpha (α) particles, beta (β) particles, electrons, positrons, and neutrons. Ionizing radiation may be produced from manufactured devices such as X-ray tubes or from the disintegration of radioactive nuclides. Some nuclides occur naturally, but others may be produced artificially, for example, in nuclear reactors. The basic reaction of ionizing radiation with molecules is either ionization or excitation. In ionization, an orbital electron is ejected from the molecule and forms an ion pair. Directly ionizing particles are charged and possess the energy to produce ionizations along their paths from impulses imparted to orbital electrons by electrical forces between the charged particles and electrons. In excitation, an electron is raised to a higher energy level. Indirectly ionizing radiations are not charged and penetrate a medium until they collide with elements of the atom and liberate energetically charged ionizing particles.

Electromagnetic Spectrum

The electromagnetic spectrum is defined as the ordered array of known electromagnetic radiations including cosmic rays; gamma rays; X-rays; ultraviolet, visible, and infrared radiations; and radiowaves (Weast 1985). The energy transfer by electromagnetic waves can be described by discrete processes with elementary units called *photons* (Kiefer 1990). Their energy, E, is given by E = hv, where h is Planck's constant and v is the frequency. Because velocity C, wavelength , and frequency v are related (C = v), E = hc/ (Kiefer 1990). The relations between E, v, and for parts of the total spectrum of the electromagnetic waves are shown in Figure 1. The high energy radiation that enters the earth's atmosphere from outer space is known as primary cosmic rays. On

interaction with the nuclei of atoms in the air, secondary cosmic rays and a variety of reaction products (cosmogenic nuclides) such as ^{3}H, ^{7}Be, ^{10}Be, ^{14}C, ^{22}Na, and ^{24}Na are produced (UNSCEAR 1988).

Radionuclides

Radioactive nuclides contain atoms that disintegrate by emission of subatomic particles and gamma or X-ray photons (Weast 1985; Hobbs and McClellan 1986; Kiefer 1990; Rose et al. 1990). In alpha decay, a helium nucleus of 2 protons and 2 neutrons is emitted and reduces the mass number by 4 and the atomic number by 2. In beta decay, an electron--produced by the disintegration of a neutron into a proton, an electron, and an antineutrino--is emitted from the nucleus and increases the atomic number by 1 without changing the mass number. Sometimes a positron together with a neutrino is emitted. And sometimes an electron may be captured from the K (outermost) shell of the atom; the resultant electron hole in the K shell is filled by electrons from outer orbits and causes the emission of X-rays. Alpha and beta decay generally leave the resultant daughter nuclei in an excited state that is deactivated by emission of photons. Although emission accompanies most decays, it is not always detected, especially not with light β emitters such as ^{3}H, ^{14}C, ^{32}P, and ^{35}S. The half-life of individual radionuclides can be measured (i.e., the time during which half the atoms of the radionuclide spontaneously decay to a daughter nuclide). Another form of nuclear breakdown is fission in which the nucleus breaks into two nuclides of approximately half the parent's size (Rose et al. 1990). The symbol, mass number, atomic number, half-life, and decay mode of all radionuclides mentioned herein are listed in Table 1.

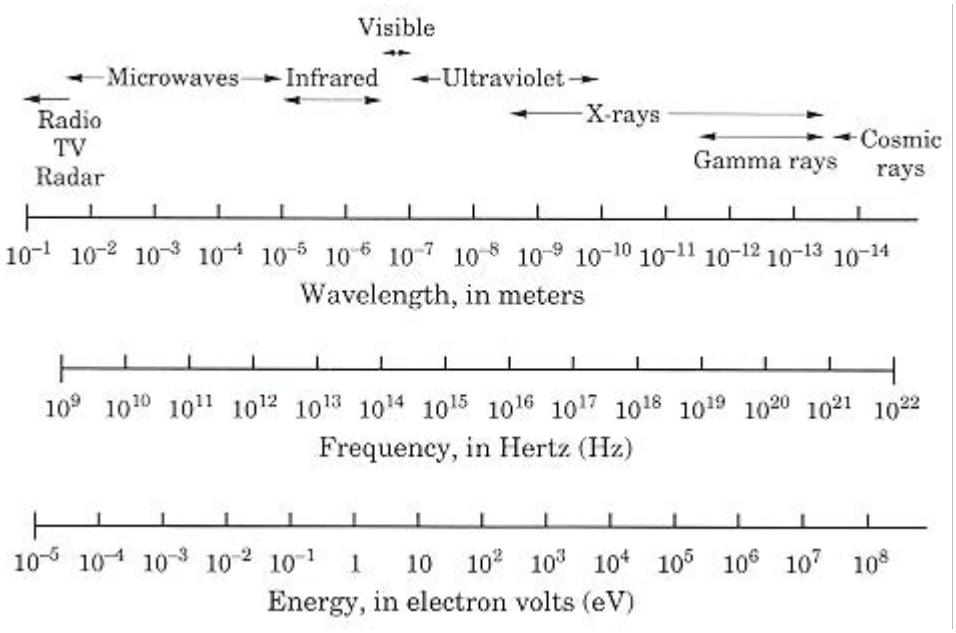

Fig. 1. The spectrum of electromagnetic waves, showing relation between wavelength, frequency, and energy (modified from Kiefer 1990).

Four general groups of radionuclides are distinguished: (1) a long half-life group (i.e., Tb ½ >10^{9} years) of elements including ^{238}U, ^{235}U, ^{232}Th, ^{40}K, ^{87}Rb, and ^{143}Sm that were formed about 4.5 billion years ago; (2) shorter-lived daughters of U and Th such as Ra and Rn that form as a result of the decay of their long-lived parents; (3) nuclides (i.e., ^{14}C and ^{3}H) formed by continuing natural nuclear transformations that are driven by cosmic rays, natural sources of neutrons, or energetic particles that are formed in the upper atmosphere by cosmic rays; and (4) nuclides that are formed as a result of nuclear weapons tests, nuclear reactor operations, and other human activities. Important members of this group include ^{90}Sr, ^{137}Cs, ^{14}C, and ^{3}H; note that many members of the third group such as ^{14}C and ^{3}H are also formed in this fourth fashion (Rose et al. 1990).

Radioactive decay usually does not immediately lead to a stable end product but to other unstable nuclei that form a decay series (Kiefer 1990). The most important examples of unstable nuclei are started by heavy, naturally occurring nuclei. Because the mass number changes only with α-decay, all members of a series may be classified according to their mass numbers (see the uranium-238 decay series in Figure 2). A total of three natural decay series--formed at the birth of our planet--are named after their parent isotope: ^{232}Th, ^{235}U, and ^{238}U (Fig. 3). Several shorter decay series also exist. For example, ^{90}Sr decays with a Tb ½ of 28 years by β emission to ^{90}Y, which in turn disintegrates (β emission) with a Tb ½ of 64 h to the stable ^{90}Zr (Kiefer 1990). Other examples of known radionuclides since the earth's origin include ^{40}K and ^{87}Rb. In hazard assessments, all members of a decay series must be considered.

Linear Energy Transfer

The deposition of energy in an exposed body is mediated almost exclusively by charged particles. These particles cause ionizations but lose energy with each ionization until they reach the end of their ranges. Depending on the type of particle, the ionizations are more or less closely spaced and described by the energy loss of a traversing particle. The linear energy transfer (LET) is defined as the amount of locally absorbed energy per unit length, that is, only the energy fraction that leads to ionizations or excitations in the considered site is counted (Kiefer 1990; ICRP 1991a). Because radiation effects are dependent on the nature of the radiation, a weighting factor is used to modify the absorbed dose and to define the dose equivalent; this factor--now called the Radiation Weighting Factor--is a function of LET. Approximate weighting values range from 1 (X-rays, electrons, gamma rays) to 10 (neutrons, protons, singly charged particles of rest mass greater than one atomic mass of unknown energy) and to 20 (alpha particles and multiply-charged particles of unknown energy; ICRP 1977; Whicker and Schultz 1982a; Hobbs and McClellan 1986; Severa and Bar 1991). The relation between radiation type and energy to weighting factors is shown in Table 2.

Table 1. Selected radionuclides: symbol, mass number, atomic number, half-life, and decay mode (modified from Whicker and Schultz 1982a, 1982b; Weast 1985; Kiefer 1990; Severa and Bar 1991).

Table 1. Nuclide	Symbol	Mass number	Atomic number	Half-life	Major decay mode [a]
Hydrogen-3	^{3}H	3	1	12.26 years	β^-
Beryllium-7	^{7}Be	7	4	53.3 days	EC
Beryllium-10	^{10}Be	10	4	1,600,000 years	β^-
Carbon-14	^{14}C	14	6	5,730 years	β^-
Sodium-22	^{22}Na	22	11	2.6 years	β^+, EC
Sodium-24	^{24}Na	24	11	15 h	β^-
Phosphorus-32	^{32}P	32	15	14.3 days	β^-
Sulphur-35	^{35}S	35	16	87 days	β^-
Argon-39	^{39}Ar	39	18	261 years	β^-
Potassium-40	^{40}K	40	19	1,250,000,000 years	β^- β^+, EC
Potassium-42	^{42}K	42	19	12.4 h	β^-
Calcium-45	^{45}Ca	45	20	164 days	β^-
Chromium-51	^{51}Cr	51	24	28 days	EC
Manganese-54	^{54}Mn	54	25	312 days	EC
Manganese-56	^{56}Mn	56	25	2.6 h	β^-
Iron-55	^{55}Fe	55	26	2.7 years	EC

Table 1. Nuclide	Symbol	Mass number	Atomic number	Half-life	Major decay mode [a]
Iron-59	^{59}Fe	59	26	45 days	β^-
Cobalt-57	^{57}Co	57	27	271 days	EC
Cobalt-58	^{58}Co	58	27	71 days	β^+, EC
Cobalt-60	^{60}Co	60	27	5.3 years	β^-
Nickel-63	^{63}Ni	63	28	100 years	β^-
Nickel-65	^{65}Ni	65	28	2.5 h	β^-
Copper-64	^{64}Cu	64	29	12.7 h	β^- β^+, EC
Zinc-65	^{65}Zn	65	30	244 days	β^+, EC
Selenium-75	^{75}Se	75	34	118 days	EC
Krypton-85	^{85}Kr	85	36	10.72 years	β^-
Rubidium-86	^{86}Rb	86	37	18.6 days	β^-
Rubidium-87	^{87}Rb	87	37	49,000,000,000 years	β^-
Strontium-85	^{85}Sr	85	38	64.8 days	EC
Strontium-89	^{89}Sr	89	38	50.5 days	β^-
Strontium-90	^{90}Sr	90	38	29 years	β^-
Yttrium-90	^{90}Y	90	39	64 h	β^-
Yttrium-91	^{91}Y	91	39	59 days	β^-
Zirconium-95	^{95}Zr	95	40	65 days	β^-
Niobium-95	^{95}Nb	95	41	35 days	β^-
Molybdenum-99	^{99}Mo	99	42	66 h	β^-
Technetium-99	^{99}Tc	99	43	213,000 years	β^-
Technetium-99m	^{99m}Tc	99	43	6 h	IT
Ruthenium-103	^{103}Ru	103	44	40 days	β^-
Ruthenium-106	^{106}Ru	106	44	373 days	β^-
Rhodium-106	^{106}Rh	106	45	29.8 s	β^-
Palladium-109	^{109}Pd	109	46	14 h	β^-
Silver-108m	^{108m}Ag	108	47	130 years	EC, IT
Silver-110m	^{110m}Ag	110	47	250 days	β^-, IT
Silver-110	^{110m}Ag	110	47	24.6 s	β^-, IT
Silver-111	^{111}Ag	111	47	7.5 days	β^-
Silver-113	^{113}Ag	113	47	5.3 h	β^-
Cadmium-109	^{109}Cd	109	48	462 days	EC
Cadmium-113m	^{113m}Cd	113	48	13.7 years	β^-
Cadmium-115m	^{115m}Cd	115	48	44.6 days	β^-
Cadmium-115	^{115}Cd	115	48	54 h	β^-
Tin-123	^{123}Sn	123	50	129 days	β^-
Tin-126	^{126}Sn	126	50	100,000 years	β^-

Table 1. Nuclide	Symbol	Mass number	Atomic number	Half-life	Major decay mode [a]
Antimony-124	^{124}Sb	124	51	60 days	β^-
Antimony-125	^{125}Sb	125	51	2.7 years	β^-
Antimony-127	^{127}Sb	127	51	3.8 days	β^-
Tellurium-127m	^{127m}Te	127	52	109 days	IT, β^-
Tellurium-129m	^{129m}Te	129	52	33 days	IT, β^-
Tellurium-129	^{129}Te	129	52	69.5 min	β^-
Tellurium-132	^{132}Te	132	52	78.2 h	β^-
Iodine-125	^{125}I	125	53	60 days	β^-, EC
Iodine-129	^{129}I	129	53	16,000,000 years	β^-
Iodine-130	$^{130}1$	130	53	12.4 h	β^-
Iodine-131	^{131}I	131	53	8 days	β^-
Xenon-131	^{131}Xe	131	54	11.9 days	IT
Xenon-133	^{133}Xe	133	54	5.3 days	β^-
Xenon-135	^{135}Xe	135	54	9.1 h	β^-
Cesium-134	^{134}Cs	134	55	2.06 years	β^-
Cesium-135	^{135}Cs	135	55	3,000,000 years	β^-
Cesium-137	^{137}Cs	137	55	30.2 years	β^-
Barium-140	^{140}Ba	140	56	12.8 days	β^-
Lanthanum-140	^{140}La	140	57	40 h	β^-
Cerium-141	^{141}Ce	141	58	33 days	β^-
Cerium-143	^{143}Ce	143	58	33 h	β^-
Cerium-144	^{144}Ce	144	58	284 days	β^-
Praseodymium-143	^{143}Pr	143	59	13.6 days	β^-
Praseodymium-144	^{144}Pr	144	59	7.2 min	IT
Praseodymium-147	^{147}Pr	147	59	13.4 min	β^-
Neodymium-147	^{147}Nd	147	60	11 days	β^-
Promethium-147	^{147}Pm	147	61	2.6 years	β^-
Samarium-143	^{143}Sm	143	62	8.8 min	β^-, EC
Samarium-151	^{151}Sm	151	62	90 years	β^-
Europium-152	^{152}Eu	152	63	13.4 years	EC, β^-
Europium-155	^{155}Eu	155	63	15.2 days	β^-
Tungsten-181	^{181}W	181	74	121 days	EC
Tungsten-185	^{185}W	185	74	75 days	β^-
Tungsten-187	^{187}W	187	74	24 h	β^-
Gold-198	^{198}Au	198	79	2.7 days	β^-
Mercury-203	^{203}Hg	203	80	47 days	β^-
Mercury-206	^{206}Hg	206	80	8.1 min	β^-

Table 1. Nuclide	Symbol	Mass number	Atomic number	Half-life	Major decay mode [a]
Thallium-206	$^{106}_{Tl}$	206	81	4.3 min	β^-
Thallium-207	$^{207}_{Tl}$	207	81	4.8 min	β^-
Thallium-208	$^{208}_{Tl}$	208	81	3 min	β^-
Thallium-210	$^{210}_{Tl}$	210	81	1.3 min	β^-
Lead-210	$^{210}_{Pb}$	210	82	22.3 years	β^-
Lead-211	$^{211}_{Pb}$	211	82	36.1 min	β^-
Lead-212	$^{212}_{Pb}$	212	82	10.6 h	β^-
Lead-214	$^{214}_{Pb}$	214	82	26.8 min	β^-
Bismuth-210	$^{210}_{Bi}$	210	83	5.0 days	β^-
Bismuth-211	$^{211}_{Bi}$	211	83	2.2 min	β^-
Bismuth-212	$^{212}_{Bi}$	212	83	1.0 h	β^-, α
Bismuth-214	$^{214}_{Bi}$	214	83	19.9 min	β^-
Bismuth-215	$^{215}_{Bi}$	215	83	7.4 min	β^-
Polonium-210	$^{210}_{Po}$	210	84	138.4 days	α
Polonium-211	$^{211}_{Po}$	211	84	0.52 s	α
Polonium-212	$^{212}_{Po}$	212	84	0.0000003 s	α
Polonium-214	$^{214}_{Po}$	214	84	0.000163 s	α
Polonium-215	$^{215}_{Po}$	215	84	0.00178 s	α
Polonium-216	$^{216}_{Po}$	216	84	0.15 s	α
Polonium-218	$^{218}_{Po}$	218	84	3.1 min	α
Astatine-215	$^{215}_{At}$	215	85	0.0001 s	α
Astatine-218	$^{218}_{At}$	218	85	1.6 s	α
Astatine-219	$^{219}_{At}$	219	85	0.9 min	α
Radon-218	$^{218}_{Rn}$	218	86	0.0356 s	α
Radon-219	$^{219}_{Rn}$	219	86	3.96 s	α
Radon-220	$^{220}_{Rn}$	220	86	56 s	α
Radon-222	$^{222}_{Rn}$	222	86	3.8 days	α
Francium-223	$^{223}_{Fr}$	223	87	21.8 min	β^-
Radium-223	$^{223}_{Ra}$	223	88	11.4 days	α
Radium-224	$^{224}_{Ra}$	224	88	3.7 days	α
Radium-226	$^{226}_{Ra}$	226	88	1,620 years	α
Radium-228	$^{228}_{Ra}$	228	88	5.75 years	β^-
Actinium-227	$^{227}_{Ac}$	227	89	21.8 years	β^-
Actinium-228	$^{228}_{Ac}$	228	89	6.13 h	β^-
Thorium-227	$^{227}_{Th}$	227	90	18.8 days	α
Thorium-228	$^{228}_{Th}$	228	90	1.91 years	α
Thorium-230	$^{230}_{Th}$	230	90	75,400 years	α

Table 1. Nuclide	Symbol	Mass number	Atomic number	Half-life	Major decay mode [a]
Thorium-231	^{231}Th	231	90	25.6 h	β^-
Thorium-232	^{232}Th	232	90	14,000,000,000 years	α
Thorium-234	^{234}Th	234	90	24 days	β^-
Protactinium-231	^{231}Pa	231	91	32,700 years	α
Protactinium-234	^{234}Pa	234	91	6.7 h	β^-
Protactinium-234m	^{234m}Pa	234	91	1.17 min	β^-, IT
Uranium-233	^{233}U	233	92	160,000 years	α
Uranium-234	^{234}U	234	92	245,000 years	α
Uranium-235	^{235}U	235	92	710,000,000 years	α
Uranium-236	^{236}U	236	92	23,400,000 years	α
Uranium-238	^{238}U	238	92	4,470,000,000 years	α
Neptunium-235	^{235}Np	235	93	1.08 years	EC
Neptunium-237	^{237}Np	237	93	2,140,000 years	α
Neptunium-239	^{239}Np	239	93	2.35 days	β^-
Neptunium-241	^{241}Np	241	93	13.9 min	β^-
Plutonium-238	^{238}Pu	238	94	87.7 years	α
Plutonium-239	^{239}Pu	239	94	24,110 years	α
Plutonium-240	^{240}Pu	240	94	6,537 years	α
Plutonium-241	^{241}Pu	241	94	14.4 years	β^-
Plutonium-242	^{242}Pu	242	94	376,000 years	α
Plutonium-244	^{244}Pu	244	94	82,000,000 years	α
Americium-241	^{241}Am	241	95	458 years	α
Americium-243	^{243}Am	243	95	7,370 years	α
Curium-241	^{241}Cm	241	96	33 days	EC
Curium-242	^{242}Cm	242	96	463 days	α
Curium-243	^{243}Cm	243	96	28.5 years	α
Curium-244	^{244}Cm	244	96	18.1 years	α
Curium-247	^{247}Cm	247	96	15,600,000 years	α
Curium-248	^{248}Cm	248	96	340,000 years	α
Curium-250	^{250}Cm	250	96	7,400 years	SF
Californium-252	^{252}Cf	252	98	2.6 years	α, SF

[a] Observed modes of decay for all radioactive species: α = particle emission; β = beta emission, β^+ = positron emission; EC = electron capture resulting in X-ray emission; IT = isomeric transition from higher to lower energy state; SF = spontaneous fission.

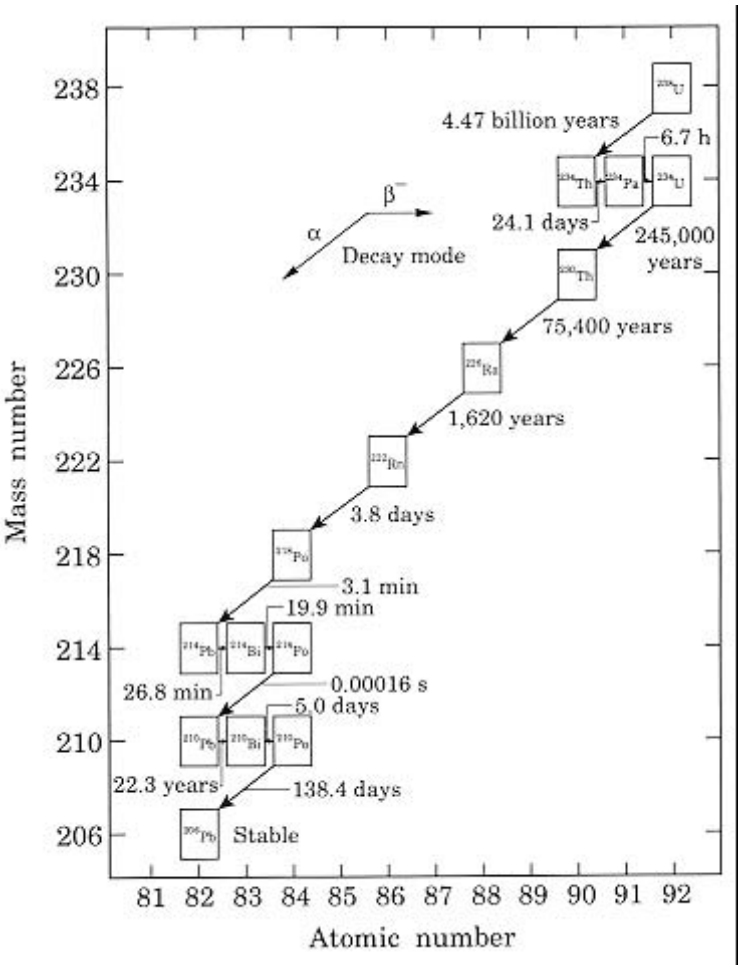

Fig. 2. The principal uranium-238 decay series, indicating major decay mode and physical half-time of persistence (modified from Cecil and Gessell 1992).

New Units of Measurement

A variety of units have been used for the assessment of exposures to ionizing radiation. The current international standard terminology is shown in Table 3. In this report, I use only the new terminology; this frequently necessitated data transformation of units from early published accounts into the currently accepted international terminology.

Table 2. Radiation weighting factors for various types of ionizing radiations (International Commission on Radiological Protection 1991a).

Radiation type and energy range	Radiation weighting factor
X-rays, gamma rays, beta particles, electrons, muons; all energies	1
Neutrons	
10 keV	5
10 keV-100 keV	10
>100 keV-2 MeV	20
>2 MeV-20 MeV	10
>20 MeV	5
Protons	5
Alpha particles, fission fragments, heavy nuclei	20

Table 3. New units[a] for use with radiation and radioactivity (International Commission on Radiological Protection 1977, 1991a; Hobbs and McClellan 1986; United Nations Scientific Committee on the Effects of Atomic Radiation 1988).

Variable	Old unit	New unit	Old unit in terms of new unit
Activity	Curie (Ci) = 3.7 x 10^{10} disintegrations /s (dps)	Becquerel (Bq) = 1 dps	1 Ci = 3.7 x 10^{10} Bq
Exposure	Roentgen (R) = 2.58 x 10^{-4} Coulombs/kg	Coulomb/kg (C/kg)	1 R = 2.58 x 10^{-4} C/kg
Absorbeddose	Rad 100 erg/g	Gray (GY) = 1 J/kg	1 Rad = 0.01 Gy
Dose equivalent	Rem = damage effects of 1 R	Sievert (Sv) = 1 J/kg	1 Rem = 0.01 Sv

[a] See Glossary.

Sources and Uses

General

Most external exposure of living organisms to radiation is from naturally occurring electromagnetic waves, and most internal exposure is from naturally occurring radionuclides such as potassium-40. Natural radiation doses vary significantly with altitude, radionuclide concentrations in the biogeophysical environment, and uptake kinetics. The major source of global anthropogenic radioactivity is fallout from military atmospheric-weapons testing; locally, radiation levels tend to be elevated near nuclear power-production facilities, nuclear-fuel reprocessing plants, and nuclear-waste disposal sites. Dispersion of radioactive materials is governed by a variety of physical chemical, and biological vectors, including winds, water currents, plankton, and avian and terrestrial wildlife.

Natural Radioactivity

Exposure to natural sources of radiation is unavoidable. Externally, individuals receive cosmic rays, terrestrial X-rays, and gamma radiation. Internally, naturally occurring radionuclides of Pb, Po, Bi, Ra, Rn, K, C, H, U, and Th contribute to the natural radiation dose from inhalation and ingestion. Potassium-40 is the most abundant radionuclide in foods and in all tissues. The mean effective human dose equivalent from natural radiations is 2.4 milliSieverts (mSv); this value includes the lung dose from radon-daughter products and is about 20% higher than a 1982 estimate that did not take lung dose into account (Table 4).

Table 4. Annual effective dose equivalent to humans from natural sources of ionizing radiation (Whicker and Schultz 1982a; Hobbs and McClellan 1986; United Nations Scientific Committee on the Effects of Atomic Radiation 1988; Aarkrog 1990).

Source of radiation	Dose equivalents (mSv)
Cosmic rays	
Ionizing component	0.30
Neutron component	0.06
Cosmogenic radionuclides (mainly ^{3}H and ^{14}C)	0.02
Primordial radionuclides	
Potassium-40	0.33
Rubidium-87	0.01
Uranium-238 series	1.34
Thorium-232 series	0.34
Total	2.4

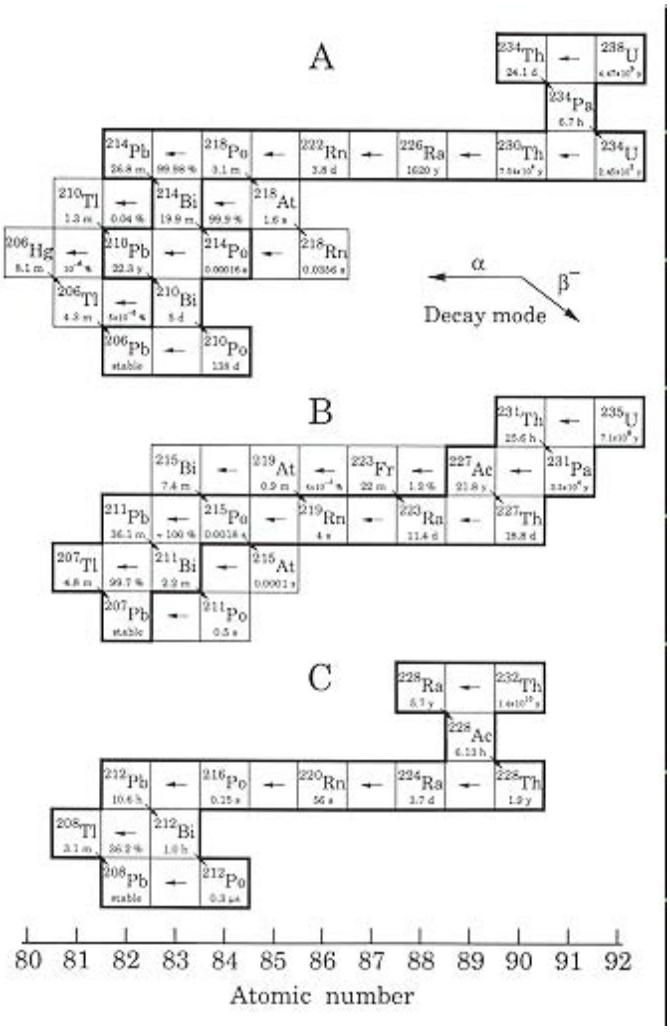

Fig. 3. The three, still-existing natural decay series. A. Uranium-238; B. Uranium-235; and C. Thorium-232 (modified from Holtzman 1969; LWV 1985; UNSCEAR 1988; Kiefer 1990; Rose et al. 1990). Principal decay products occur inside the heavy borders.

Table 5. Annual whole-body radiation doses to humans from various sources (Gray et al. 1989).

Source of radiation	Dose (mSv)
Natural external background	
Denver, Colorado	1.65
Washington State, mean	0.88
United States, average	0.84
Hanford, Washington	0.59
Average medical dose per capita, United States	0.36
Average internal dose from natural radioactivity, United States	0.25
Global weapons fallout	0.05
Consumer product radiation (TV, smoke detector, and other sources)	0.02
Total	1.27-2.33

The dose of natural radiation that an organism receives depends on height above sea level, amount and type of radionuclides in the soil of its neighborhood, and the amount taken up from air, water, and food (ICRP 1977; Whicker and Schultz 1982a; Hobbs and McClellan 1986; UNSCEAR 1988; Aarkrog 1990; Kiefer 1990; Nozaki 1991). Natural radiations in various ecosystems result in radiation dose equivalents that usually are between <0.005 and 2.07 mSv annually (Fig. 4). Radiation doses are substantially higher at atypically elevated local sites (Table 5), such as Denver, and sometimes exceed 17 mSv annually in mountainous regions of Brazil and the former Soviet Union (Whicker and Schultz 1982a).

Anthropogenic Radioactivity

Nuclear explosions and nuclear power production are the major sources of anthropogenic activity in the environment. But radionuclide use in medicine, industry, agriculture, education, production and transport, and disposal from these activities present opportunities for wastes to enter the environment (Whicker and Schultz 1982a; Table 6). Radiation was used as early as 1902 in the treatment of diseases such as an enlarged thymus, tinea capitis, acne, and cancers of childhood and adolescence (Bowden et al. 1990). The use of X-rays by physicians and dentists represents the largest source of annual dose equivalent of the U.S. population to artificial radiation: 0.78-1.01 mSv to bone marrow and 0.016 mSv to the upper GI tract; radiopharmaceuticals contribute an additional 0. 14 mSv or a yearly total mean dose of 0.94-1.17 mSv to bone marrow (Hobbs and McClellan 1986).

Table 6. Sources and applications of atomic energy (Joseph et al. 1971).

Source and output	Application
Nuclear reactor	Electric power (stationary or portable plants), desalination,
Steam, electricity	propulsion of submarines and surface ships
Heat, electricity, neutrons	Spacecraft and satellite power, spacecraft propulsion, research and special materials production
Nuclear explosives, kinetic energy	Military and civilian applications: large-scale earth moving, subsurface excavation, mineral extraction from underground
Encapsulated radioisotopes	Marine navigation aids, unmanned weather stations,
Electricity	spacecraft project power, artificial human organs
Beta and gamma radiation	Food preservation, polymerization, sterilization of medical supplies, thickness gauges
Radionuclides, beta and gamma radiation	Medical uses, tracers in scientific research, measures of manufacturing processes

Atmospheric testing of nuclear weapons is an important human source of environmental radiation (Hobbs and McClellan 1986; UNSCEAR 1988; Aarkrog 1990; Table 7). The first test explosion of a nuclear weapon took place in 1945. Atmospheric tests by the United States, the former Soviet Union, and the United Kingdom continued until they were banned in 1963. France and the People's Republic of China continued to conduct limited atmospheric tests, although no atmospheric nuclear explosions have taken place since 1980. Large nuclear explosions in the atmosphere carry most of the radioactive material into the stratosphere where it remains for 1 to 5 years, depending on the altitude and the latitude; fallout can occur years after an explosion injected material into the atmosphere. Smaller explosions carry the radioactive material only into the troposphere, and fallout occurs within days or weeks. Fallout was highest in the temperate regions and in the northern hemisphere where most of the testing was done. The most abundant radionuclides from atmospheric tests to date are $^{14}C > {}^{137}Cs > {}^{95}Zr > {}^{90}Sr > {}^{106}Ru > {}^{144}Ce > {}^{3}H$. Of the many radionuclides produced in nuclear and thermonuclear explosions, the primary contributors to human radiation exposure include ^{14}C, $^{89+90}Sr$, ^{95}Zr, ^{106}Ru, ^{131}I, ^{137}Cs, ^{141}Ce, and ^{144}Ce; isotopes of plutonium and americium--although present in quantity--are not significant contributors because of their low solubility. The primary dose from fallout radiation is through external gamma radiation, assimilation through the food chain, or beta radiation of the skin.

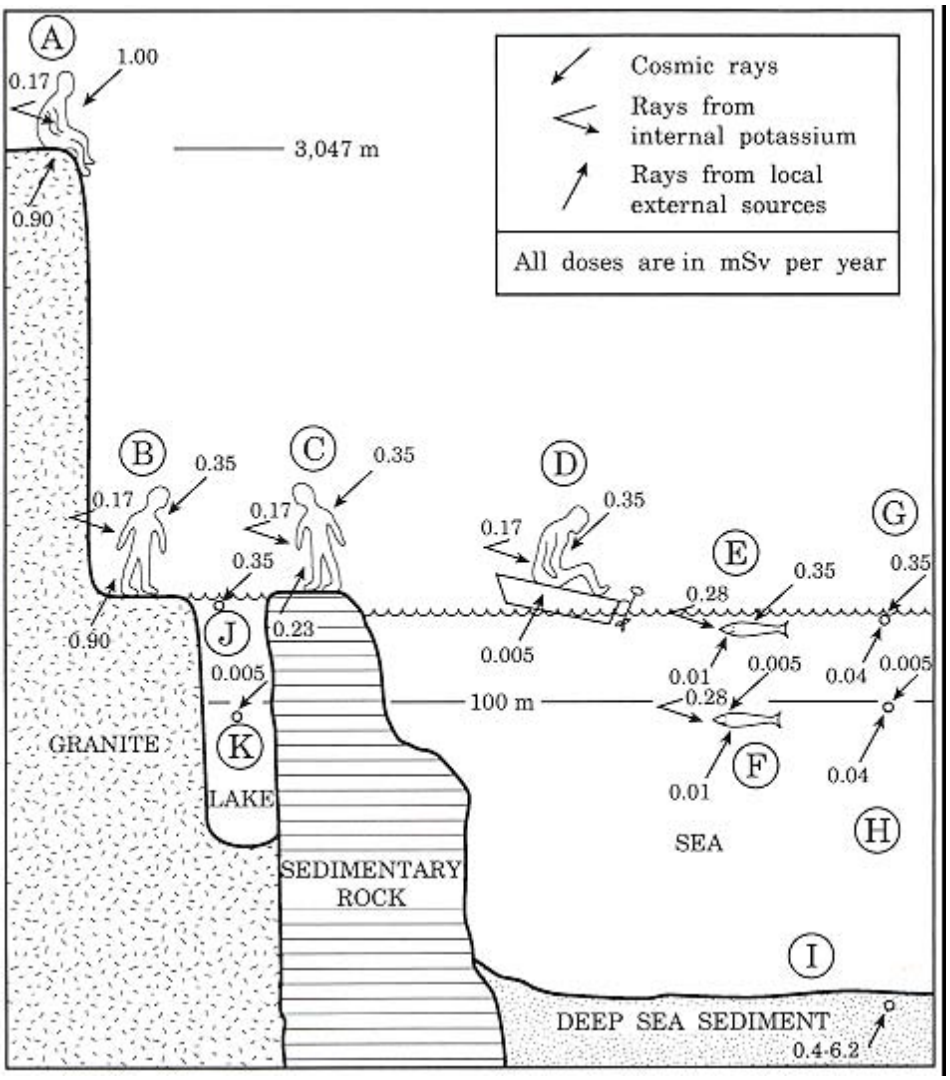

Fig. 4. Natural radiations in selected radiological domains (modified from Folsom and Harley 1957). A. Human over granite at 3,047 m (10,000 feet) elevation above sea level; total annual dose equivalent of 2.07 mSv (cosmic rays 1.00, granite 0.90, internal emitters 0.17); B. Human over granite at sea surface; total annual dose of 1.42 mSv; C. Human over sedimentary rock at sea level; total annual dose of 0.75 mSv; D. Human over sea; total annual dose of 0.525 mSv; E. Large fish in sea near surface; total annual dose of 0.64 mSv; F. Large fish in sea at depth of 100 m; total annual dose of 0.295 mSv; G. Microorganism in water near sea surface; total annual dose of 0.39 mSv, H. Microorganism in water >100 m deep in sea; total annual dose of 0.045 mSv; I. Microorganism buried in deep sea sediments; total annual dose between 0.4 and 6.2 mSv, J. Microorganism near freshwater surface; total annual dose of 0.35 mSv; K. Microorganism 100 m deep in a fresh-water lake; total annual dose of 0.005 mSv.

Radioisotope thermoelectric generators (RTG) are sometimes used as power sources for space systems. In April 1964, a United States RTG navigational satellite, SNAP 9A, reentered the atmosphere and burned up at high altitude over the Mozambique Channel, releasing 629 trillion becquerels (TBq), equivalent to 17,000 Ci of ^{238}Pu and 0.48 TBq of ^{239}Pu (Whicker and Schultz 1982a; Richmond 1989). In January 1978, a Soviet RTG satellite, Kosmos 954, reentered the atmosphere over Canada and spread radiouranium across parts of that country (Richmond 1989). The amount of radioactive materials in space applications is expected to increase (Richmond 1989).

Significant amounts of radioactivity are present in the Great Lakes basin, which has numerous nuclear reactors and uranium-mine waste areas (Joshi 1991). The prevailing low levels of artificially produced radionuclides, arising largely from previous fallout (Table 8), provide small doses of radiation to residents who consume lake water. Radionuclides enter the Great Lakes ecosystem from natural and anthropogenic processes. The main natural processes that introduce radioactivity are the weathering of rocks, which contain uranium-and thorium-series radionuclides, and fallout of cosmic ray-produced radionuclides such as ^{3}H, ^{7}Be, and ^{14}C. Anthropogenic radioactivity is created, for example, by uranium mining, milling, and fuel fabrication; releases of artificially produced radionuclides by nuclear power reactors and nuclear fuel processing plants; medical uses of radioisotopes; and coal-fired electrical generating plants (Joshi 1991).

Table 7. Annual effective dose equivalent from nuclear-weapons testing to humans in the north temperate zone (Aarkrog 1990).

Nuclide	Dose (mSv[a])
^{3}H	0.05
^{14}C	2.6
^{90}Sr	0.18
^{95}Zr	0.29
^{106}Ru	0.14
^{131}I	0.05
^{137}Cs	0.88
^{144}Ce	0.09
Pu and Am nuclides	0.09
Other nuclides	0.08
Total	4.45[b]

[a] External, 24%; inhalation, 5%; ingestion, 71%.

[b] Equivalent to 1.85 times the natural background dose.

Production of power from nuclear reactors involves uranium mining, fuel fabrication, the reactor operations, and storage of wastes. All of these processes may expose humans and the environment to radiation (Hobbs and McClellan 1986). Uranium production in the United States was 12,300 tons of U_3O_8 in 1977, primarily from western states, Texas, and Florida (Whicker and Schultz 1982a). Mining from deep shafts or open pits is the preferred method of uranium extraction, although in Florida it is produced as a byproduct of phosphate mining. Mines disperse radionuclides of uranium, thorium, and radium, which are associated with dust particles, and radon, which emanates from ore as a gas and decays to create a series of radioactive daughters. Groundwater also contains radionuclides of the uranium series. As many as 18 operating uranium mills were close to major mining centers in the western states. Collectively, these mills process or processed about 30,000 tons of ore daily and use acid or alkali leach methods to extract 90% to 95% of the uranium from ore. Uranium is barreled at the mill for shipment as uranium oxide or as salt concentrates (yellowcake) that contain 70-90% U_3O_8 by weight. Residues of the uranium extraction process are usually pumped as a slurry to liquid-retention impoundments; about 0.55 TBq of ^{230}Th and ^{226}Ra enter tailings each day from milling operations. Radium-226 produces gaseous ^{222} Rn; daughters of ^{222} Rn, such as ^{210}Pb, expose the surrounding biota to measurable radiation. Purification of yellowcake to UF_6 (uranium hexafluoride) and its enrichment to ^{235}U causes a loss of about 0.55 TBq annually. Nuclear reactor fuel contains about 3% ^{235}U. A nuclear explosion in a nuclear reactor is highly unlikely because the nuclear fuel that is suitable for weapons must contain more than

90% ^{235}U. Following enrichment, UF_6 is hydrolyzed to uranyl fluoride, converted to ammonium diuranate, and calcined to the dioxide UO_2. Uranium dioxide pellets at one time were prepared by as many as 10 commercial fuel fabrication plants and subsequently transported to nuclear reactors (Whicker and Schultz 1982a). In the current light-water cooled reactors the most abundant radionuclides in the reactor effluents under normal conditions are ^{3}H, ^{58}Co, ^{60}Co, ^{85}Kr, ^{85}Sr, ^{90}Sr, ^{130}I, ^{131}I, ^{131}Xe, ^{133}Xe, ^{134}Cs, ^{137}Cs, and ^{140}Ba (Hobbs and McClellan 1986). Gaseous and volatile radionuclides such as ^{85}Kr, ^{131}Xe, and ^{133}Xe contribute to the external gamma dose, whereas the others contribute to the dose externally by surface deposition and internally by way of the food chain. The mean dose from environmental releases of all radionuclides from nuclear reactors in the United States is less than 0.01 mSv/year (Hobbs and McClellan 1986). Nuclear fission follows the capture of a neutron by an atom of fissionable material such as ^{235}U or ^{239}Pu. The fission releases 1 to 3 neutrons and, if additional fissionable material is present in sufficient quantity and in the right configuration, a chain reaction occurs (Hobbs and McClellan 1986). Radionuclides formed per megaton of fission include fission products (^{89}Sr, ^{90}Sr, ^{95}Zr, ^{103}Ru, ^{106}Ru, ^{131}I, ^{137}Cs, ^{144}Ce), and activation products in air (^{3}H, ^{14}C, ^{39}Ar) and soil (^{24}Na, ^{32}P, ^{42}K ^{45}Ca, ^{55}Fe, ^{59}Fe; Whicker and Schultz 1982a). Fission-product radionuclides of potential biological importance include ^{90}Sr, ^{137}Cs, ^{131}I, ^{129}I, ^{144}Ce, ^{103}Ru, ^{106}Ru, ^{95}Zr, ^{140}Ba, ^{91}Y, ^{143}Ce, ^{147}Nd (Kahn 1971; Whicker and Schultz 1982a), and others (Table 9).

Table 8. Estimated fallout of ^{90}Sr and ^{137}Cs over the Great Lakes, 1954-83, in cumulative millions of Bq/km^2 (Joshi 1991).

Great Lake	Cesium-137	Strontium-90	**Total**
Superior	2,429	1,491	3,920
Michigan	2,738	1,680	4,418
Huron	2,670	1,638	4,308
Erie	2,859	1,754	4,613
Ontario	2,773	1,701	4,474
Total	13,469	8,264	21,733

Table 9. Fission products per kg ^{235}U reactor charge at 100 days cooling (modified from Renn 1957).

		Trillions of becquerels/kg^{235}U	
Product	Grams	Beta	Gamma
Short-lived[a]	15.93	7,217	6,002
Long-lived[b]	16.61	698	755
Inactive fission products	230.00	—	—
Total	262.54	8,045	6,757

a ^{90}Y, ^{106}Rh, ^{144}Ce, ^{95}Zr, ^{95}Nb, ^{91}Y, ^{89}Sr, ^{103}Ru, ^{141}Ce, ^{137}Ba, ^{106}Ru, ^{143}Pr, ^{140}Ba, ^{140}La, ^{131}I.

b ^{137}Cs, ^{90}Sr, ^{144}Pr, ^{129}Te.

Most of the world's supply of uranium consists of about 0.7% ^{235}U and 99% ^{238}U. In theory, about 2.27 kg of ^{235}U can release energy equivalent to 20,000 tons of TNT (Hobbs and McClellan 1986). Uranium-238 and ^{232}Th can be converted into fissionable material after neutron capture. Radionuclides of biological significance that are produced by neutron activation in nuclear reactors include ^{3}H, ^{14}C, ^{24}Na, ^{32}P, ^{35}S, ^{45}Ca, ^{54}Mn, ^{55}Fe, $^{57+58+60}Co$, ^{65}Zn, ^{239}Pu, ^{239}Np, ^{241}Am, and ^{242}Cm (Whicker and Schultz 1982a). Nuclear energy can also be released by fusion of smaller nuclei into larger nuclei that is accompanied by a decrease in mass

(Hobbs and McClellan 1986). Fusion reactors—which do not yet exist—require very high temperatures of several million degrees; no fission products are produced in the fusion process (Whicker and Schultz 1982a).

Table 10. Radioactive waste disposal at sea.

Disposer, and other variables	Quantity, in trillions of becquerels (TBq)	Reference[a]
United States		
Atlantic Ocean, 1951-60 vs. 1961-67	2,939 vs. 2	1
Pacific Ocean, 1951-60 vs. 1961-67	527 vs. 16	1
United Kingdom		
1951-67, alpha vs. beta	123 vs. 1,631	1
Sellafield, alpha (primarily Pu and Am)		
1968-70	50-61	2
1971 vs. 1972	99 vs. 143	2
1973 vs. 1974	181 vs. 17	2
Sellafield reprocessing plant[b]		
1980 vs. 1981	5,145 vs. 4,451	3
1982 vs. 1983	4,005 vs. 3,112	3
1984 vs. 1985	1,835 vs. 646	3
Europe		
Germany, Netherlands, Belgium, France; 1961; alpha vs. beta plus gamma	6 vs. 220	1
France, Cap de la Hague reprocessing plant[c]		
1980 vs. 1981	503 vs. 455	3
1982 vs. 1983	694 vs. 683	3
1984 vs. 1985	670 vs. 674	3

[a] 1, Joseph et al. 1971; 2, Hetherington et al. 1976; 3, UNSCEAR 1988.

[b] Effluent composition primarily ^{137}Cs and ^{241}Pu.

[c] Effluent composition primarily ^{106}Ru and ^{125}Sb.

Radioactive wastes are currently stored in underground tanks or in temporary storage at reactor sites for recycling or disposal (Whicker and Schultz 1982a). For low level wastes, containment and isolation are the preferred disposal options, including burial, hydraulic injection into deep geological strata, and ocean disposal (Table 10). Options for the disposal of high-level wastes include retrievable surface storage and entombment in deep geological strata; many risks are associated with these options, and more suitable alternative disposals are needed. Spent nuclear fuel elements are usually stored for about 3 months to allow the decay of shorter-lived radionuclides before reprocessing or disposal. Reprocessing involves extractions to separate uranium and plutonium from the fission products into UF_6 and plutonium dioxide. Longer-lived fission products such as ^{90}Sr and ^{137}Cs are sometimes chemically separated and encapsulated for storage or disposal. Fuel reprocessing tends to release measurable quantities of various radionuclides that are detected in fishes, wildlife, and food for humans (Whicker and Schultz 1982a). Liquid discharges from the Sellafield reprocessing plant (Table 10) have been reduced by a factor of more than 100 since the mid-1970's (Aarkrog 1990). Human populations that consume higher than average quantities of marine fish and shellfish from the Sellafield area theoretically receive about 3.5 mSv annually from radioactivity associated with nuclear power production. Human populations in the vicinity of nuclear-power production with discharges directly into the marine environment--except Sellafield--generally receive less than 0.05 mSv annually from this source (Aarkrog 1990).

Radioactive transuranic elements with atomic numbers that are greater than 92 have been introduced into the environment since the 1940's from atmospheric testing of nuclear weapons, discharges of nuclear wastes, and nuclear fuel reprocessing (Noshkin et al. 1971; Hetherington et al. 1976; Sibley and Stohr 1990; Morse and Choppin 1991). Transuranic isotopes with half-lives of more than 10,000 years (i.e., ^{247}Cm, ^{248}Cm, ^{239}Pu, ^{242}Pu, ^{244}Pu, ^{237}Np) will persist over geologically significant time periods. Transuranics at detectable but considered nonhazardous levels to biota are now widely dispersed throughout the environment in most waters, soils, sediments, and living organisms including humans. Of current primary concern are ^{244}Cm, ^{241}Am, $^{238+239+240+241}Pu$, and ^{237}Np—especially americium-241, which is increasing globally as a result of ^{242}Pu decay (Sibley and Stohr 1990; Morse and Choppin 1991). However, the estimated peak dose received from Pu and Am radioisotopes seems to be decreasing in the vicinity of the Sellafield nuclear-fuel reprocessor (Table 11). Miscellaneous exposures include radiations from television sets, luminous dial watches, smoke detectors, electron microscopes, building materials, and air travel (Hobbs and McClellan 1986). Most of the exposure in building materials is due to naturally occurring radionuclides; similarly, air travel increases radiation exposure of travellers from increased exposure to cosmic radiations. Dose equivalent rates can be as much as 3 times higher for cigarette smokers than for nonsmokers because of inhalation of ^{210}Po and ^{210}Pb from the cigarette. Some of the lung dose is also received from radionuclides that are released during combustion of fossil fuels, which contain small quantities of naturally occurring radionuclides (Hobbs and McClellan 1986).

Dispersion

Radioactive materials are cycled throughout the environment by a variety of physical, chemical, and biological vectors. Dispersion through the atmosphere is governed by the magnitude, frequency, and direction of the wind; in the hydrosphere, transport is modified by water depth, motion, temperature, winds, tides, and groundwater (Whicker and Schultz 1982b). Deposition from the atmosphere is a function of particle size, precipitation, and dry deposition. Small radioactive particles may be elevated into the airstream from the ground surface; resuspension is a function of disturbances by wind at the soil surface, atmospheric variables (i.e., velocity, turbulence, density, viscosity), and soil-ground variables such as texture, cohesiveness, moisture content, density, vegetation cover, ground surface roughness, and topography (Whicker and Schultz 1982b). Only 1 kg of the original 15 kg of Pu was fissioned from the dropping of the plutonium nuclear bomb on Nagasaki, Japan, on 9 August 1945 (Kudo et al. 1991). The remaining 14 kg of Pu escaped into the environment. Local fallout accounted for about 37 g or 0.26% of the total global fallout; the highest measured $^{239+240}Pu$ concentration was 64 Bq/kg soil about 2.8 km from ground zero (Kudo et al. 1991).

Table 11. Theoretical peak dose, in microsieverts per year, received from plutonium and americium by three human populations (McKay and Pattenden 1990).

	Year		
Population	1973	1987	2000
Average person near Sellafield nuclear fuel reprocessor	24	4	2
Critical group, mainly agricultural workers	35	20	16
Heavy consumers of Irish Sea fish and shellfish in local fishing communities	—	250	55-90

Biological agents can also transport radioactive wastes. Birds, especially waterfowl, disperse accumulated radiocesium and other radionuclides along their migratory flyways (Brisbin 1991). Native mammalian herbivores and their predators that have come in contact with radioactivity in food or soils disperse the material in their feces, urine, or regurgitated pellets (O'Farrell and Gilbert 1975). For example, the black-tailed jackrabbit (*Lepus californicus*) in the vicinity of radioactive waste-disposal trenches dispersed radioactive fecal pellets over an area of 15 km^2; elevated radioactivity readings were recorded in jackrabbits and in their predators, including feces of coyotes (*Canis latrans*) and bones of hawks (O'Farrell and Gilbert 1975).

Biological transport of trace elements and radionuclides in the sea is provided mainly by phytoplankton and zooplankton because of their (1) ability to accumulate these elements to high levels, (2) diurnal vertical migration, and (3) production of detritus in the form of fecal pellets, molts, and carcasses (Lowman et al. 1971). Considerations related to biomass, feeding rates, conversion efficiencies, migratory habits of zooplankton, and the chemical properties of trace elements suggest that the major downward transport of these elements and radionuclides is through gravitational action on fecal pellets, molts, and carcasses; direct biological transport accounts for less than 10% of the total downward movement. In estuarine and near-shore regions, the bottom sediments and their associated epiphyton often significantly influence the distribution of added radionuclides. Large populations of sessile filter feeders may drastically increase the rate of sedimentation of added trace elements and radionuclides (Table 12).

In some coastal areas, some of the radionuclides that are discharged into coastal waters from industrial establishments are recycled by the air-sea interface back onto land (McKay and Pattenden 1990). At the sea surface, aerosol is generated by bubble bursting and wave shearing. The aerosol is advected to land by onshore winds and deposited in coastal regions. Sea-to-land transfer has been documented from the vicinity of nuclear-fuel reprocessing facilities in England, Scotland, and France; however, the sea-to-land transfer pathway was only about 8% of that from the seafood pathway (McKay and Pattenden 1990). The solubility of different radionuclides at the sediment-seawater interface is variable. Plutonium solubility, for example, depends on pH, Eh, ionic strength, complexing ions, organic chelators, living accumulator organisms, and oxidation state (Mo and Lowman 1976). The oceanic distributions of many nuclides are strongly controlled by interactions with particulate matter (Nozaki 1991). Thorium is an extreme case; the high reactivity of this element accounts for its residence of only a few decades in the ocean from where it is removed largely by vertical transport in association with settling particulate matter. Lead-210 and ^{231}Pa are also particle-reactive but to a lesser extent than Th. Their oceanic mean residence time is about 100 years. The mean oceanic residence time of ^{227}Ac and Ra isotopes is about 1,000 years because of particulate scavenging; these nuclides are supplied by insoluble parents in underlying sediments and are released to overlying waters by porewater diffusion. Radium-228 can serve as a novel tracer in ocean circulation for about 30 years; ^{227}Ac can be used for about 100 years. The distribution of ^{226}Ra is largely governed by biogeochemical cycling, much like dissolved silica (Nozaki 1991).

Table 12. Time required to transport selected radionuclides added into marine waters at surface from the upper mixed layer by biological transport. Processes include diurnal vertical migration, fecal pellets, and sinking of dead matter (Lowman et al. 1971).

	Time required to transport radionuclides (in years)		
Radionuclide	Eastern North Pacific	Coastal areas	Upwelling areas
^{54}Mn	74	7	3
$^{55+59}Fe$	7.2	0.7	0.3
$^{57+58+60}Co$	220	20	8.8
^{65}Zn	12	1.1	0.5
^{95}Zr	5.4	0.5	0.2
^{210}Pb	7.3	0.7	0.3

Radionuclide Concentrations in Field Collections

General

The wide dispersion of anthropogenic radiocontaminants has significantly altered natural background levels of radioactivity in many parts of the globe. Radionuclide concentrations in selected abiotic materials and living organisms were usually elevated in samples from the vicinity of human nuclear activities, especially atmospheric military tests. Radionuclide concentrations in organisms were significantly modified by age, sex, diet, metabolism, trophic level, proximity to point source, and many other biological, chemical, and physical variables,

as discussed later. Additional and more detailed data on environmental radionuclide concentrations and isotopic composition and levels of radioactive wastes discharged into the biosphere from nuclear plants and other anthropogenic activities are given in Schultz and Klement (1963), Nelson and Evans (1969), Nelson (1971), IAEA (1976), Whicker and Schultz (1982a, 1982b), and UNSCEAR (1988).

Abiotic Materials

Radionuclide concentrations in selected nonliving materials (Table 13) show that concentrations are elevated in samples from the site of repeated nuclear detonations, near nuclear-fuel reprocessing and waste facilities, and from locations that receive radioactive fallout from atmospheric military tests. Rocks, especially granite, had high levels of naturally occurring radionuclides such as ^{40}K. Concentrations were usually low or negligible in drinking water and in cow's milk for human consumption. Nuclear-weapons testing has released large amounts of radionuclides into the environment. Between 1961 and 1966, for example, the Republic of Korea received fallout from nuclear tests by the former Soviet Union in 1961 and by the United States in 1962 and from 3 explosions by the People's Republic of China (Bai 1969). The highest levels of total combined β and activity in various Korean samples during 1962-64, in Bq/L or Bq/kg, were 0.0002 in air, 133 in water, 1,572 in milk, 2,023 in rain, 16,428 in plants, and 99,345 in soils (Bai 1969).

Table 13. Radionuclide concentrations in field collections of selected materials. Concentrations are in becquerels per kilogram fresh weight (FW) or dry weight (DW).

Table 13 Material, radionuclide, and other variables	Concentration (in Bq/kg or Bq/L)	Reference [a]
Common rock types		
Shale, limestone, sandstone, basalt		
^{40}K	63-518 DW	2
^{232}Th	4-48 DW	2
^{238}U	6-44 DW	2
Granite vs. beach sands		
^{40}K	1,184 DW vs. 100 DW	2
^{232}Th	74 DW vs. 25 DW	2
^{238}U	62 DW vs. 37 DW	2
Drinking water		
Mol, Belgium, 1983, near former nuclear fuel reprocessing plant closed in 1974, ^{129}I	Max. 0.000082 FW	3
United States, nationwide		
1977 vs. 1981		
^{238}Pu	Max. 0.00004 FW vs. max. 0.0004 FW	1,4
^{239}Pu	Max. 0.0004 FW vs. max. 0.0003 FW	1,4
^{234}U	Max. 0.093 FW vs. max. 2.19 FW	1,4
^{235}U	Max. 0.0026 FW vs. max. 0.027 FW	1,4
^{238}U	Max. 0.067 FW vs. max. 0.562 FW	1,4
1988		
^{131}I	Max. 0.011 FW	5
^{238}Pu	Max. 0.002 FW	6

Table 13 Material, radionuclide, and other variables	Concentration (in Bq/kg or Bq/L)	Reference [a]
$^{239+240}Pu$	Max. 0.0003 FW	6
^{226}Ra	Usually <0.007 FW; max. 0.24 FW	6
^{90}Sr	Max. 0.018 FW	6
^{234}U	Max. 0.090 FW	6
^{235}U	Max. 0.007 FW	6
^{238}U	Max. 0.183 FW	6
1989, ^{131}I	Max. 0.022 FW	7
1990, ^{131}I	Max. 0.022 FW	8
Freshwater		
Vicinity of nuclear weapons tests and operation of nuclear reactors, maximum values		
^{141}Ce	0.08 FW	2
^{144}Ce	0.41 FW	2
^{137}Cs	0. 18 FW	2
^{131}I	5.2 FW	2
^{54}Mn	0.05 FW	2
^{103}Ru	0.25 FW	2
^{106}Ru	1.1 FW	2
^{89}Sr	1.9 FW	2
^{90}Sr	0.66 FW	2
$^{95}Zr/^{95}Nb$	2.4 FW	2
Typical maximum concentrations		
^{3}H	0.6 FW	2
^{40}K	0.2 FW	2
^{210}Pb	0.01 FW	2
^{210}Po	0.008 FW	2
^{226}Ra	0.11 FW	2
^{87}Rb	0.00007 FW	2
^{222}Rn	6.7 FW	2
^{232}Th	0.0002 FW	2
^{234}U	0. 12 FW	2
^{235}U	0.002 FW	2
^{238}U	0.06 FW	2
Groundwater		
United States, nationwide, ^{222}Rn, 1981 vs. 1982	Usually <10 FW, max. 388 FW vs. max. 90 FW	1,9
Lakewater		
Canada 1984-87, ^{226}Ra		
Near uranium tailings area, dissolved vs. total	0. 12 FW vs. 0.56 FW	10
Control site, dissolved vs. total	0.012 FW vs. 0.009 FW	10

Table 13 Material, radionuclide, and other variables	Concentration (in Bq/kg or Bq/L)	Reference [a]
Great Lakes, 1973 vs. 1981		
^{137}Cs	0.003 FW vs. 0.0006-0.002 FW	11
^{3}H	12.6 FW vs. 6.7-13.5 FW	11
^{90}Sr	0.019-0.047 FW vs. 0.016-0.024 FW	11
Milk, (cow) pasteurized		
Mol, Belgium, 1983, near former nuclear fuel reprocessing plant, ^{129}I	Max. 0.0005 FW	3
United States, nationwide		
1975 vs. 1977		
^{14}C	17.7-18.8 FW vs.—[b]	12
^{137}Cs	Max. 1.07 FW vs. max. 1.04 FW	4,12
^{129}I	— vs. ND[c]	4
^{131}I	ND vs. max. 0.59 FW	4,12
^{89}Sr	ND vs. max. 0.22 FW	4,12
^{90}Sr	Max. 0.17 FW vs. max. 0.27 FW	4,12
1978 vs. 1981		
^{137}Cs	Max. 0.92 FW vs. max. 0.66 FW	9,13
^{131}I	Max. 0.29 FW vs. max. 0.48 FW	9,13
^{89}Sr	Max. 0.15 FW vs. max. 0.07 FW	9,13
^{90}Sr	Max. 0.32 FW vs. max. 0.14 FW	9,13
1982 vs. 1988		
^{137}Cs	Max. 0.67 FW vs. max. 0.70 FW	1,15,16
^{131}I	Max. 0.25 FW vs. max. 0.48 FW	1,15,16
^{89}Sr	Max. 0.07 FW vs. 0.007-0.09 FW	1,16
^{90}Sr	Max. 0.13 FW vs. max. 0.07 FW	1,16
1983, ^{14}C	16.1-17.5 FW	14
1989 vs. 1990		
^{137}Cs	Max. 0.78 FW vs. max. 0.67 FW	5,7,8, 14,17,18, 19
^{131}I	Max. 0.66 FW vs. max. 0.48 FW	5,7,8, 14,17,18,
^{89}Sr	Max. 0.11 FW vs. —	5,14,17, 18
^{90}Sr	Max. 0.18 FW vs. —	5,14,17, 18
Precipitation		
United States, nationwide		
1978		
^{238}Pu	Max. 0.0004 FW	13
^{239}Pu	Max. 0.0006 FW	13

Table 13 Material, radionuclide, and other variables	Concentration (in Bq/kg or Bq/L)	Reference [a]
^{234}U	Max. 0.004 FW	13
^{235}U	Max. 0.0001 FW	13
^{238}U	Max. 0.003 FW	13
1987 vs. 1988		
^{238}Pu	Max. 0.0007 vs. max. 0.001FW	5,15
$^{239+240}Pu$	Max. 0.0003 FW vs. max. 0.0005 FW	5,15
^{234}U	Max. 0.013 FW vs. max. 0.002 FW	5,15
^{235}U	Max. 0.0004 1FW vs. max. 0.0003 FW	5,15
^{238}U	Max. 0.0026 FW vs. max. 0.002 FW	5,15
Seawater		
Major fallout radionuclides in surface seawater, typical concentrations		
^{14}C	0.0004-0.001 FW	2
^{137}Cs	0.005-0.04 FW	2
^{3}H	0.3-1.8 FW	2
^{90}Sr	0.003-0.026 FW	2
^{239}Pu	0.000004-0.00005 FW	2
Natural radionuclides in surface seawater, typical concentrations		
^{3}H	0.022-0.111 FW	2
^{14}C	0.007 FW	2
^{40}K	11.8 FW	2
^{210}Pb	<0.0003 FW	2
^{210}Po	0.0002-0.001 FW	2
^{226}Ra	0.0016 FW	
^{228}Ra	0.00004-0.004 FW	2
^{87}Rb	0.107 FW	
^{228}Th	0.00007-0.0001 FW	2
^{230}Th	<0.00005 FW	2
^{232}Th	<0.00003 FW	2
^{234}U	0.048 FW	2
^{235}U	<0.002 FW	2
^{238}U	0.044 FW	2
Sediments		
Deep Ocean		
^{232}Th	1-74 DW	2
^{238}U	5-37 DW	2
Hanford, Washington, 1973, plutonium processing waste pond		
^{241}Am	2,627 DW	20
^{238}Pu	4,144 DW	20
$^{239+240}Pu$	4,477 DW	20

Table 13 Material, radionuclide, and other variables	Concentration (in Bq/kg or Bq/L)	Reference [a]
Hudson River estuary, 1970, ^{137}Cs, bottom sediments vs. suspended sediments	75 DW vs. 152 DW	21
Soils		
Belgium, Mol, near former nuclear fuel reprocessing plant, 1983, ^{129}I	Max. 0.2 DW	3
Tennessee, 1974, ^{137}Cs; 12-22 cm depth; accidentally contaminated in 1944 vs. control site	Usually near 185,00 DW, max. 740,000 DW vs. <222 DW	22
Water, various locations		
Hanford, Washington; plutonium processing waste ponds		
^{241}Am	0.04 FW	20
^{238}Pu	0.0003 FW	20
$^{239+240}Pu$	0.00007 FW	20
Hudson River estuary, 1970, ^{137}Cs, dissolved vs. suspended	0.01 FW vs. 0.005 FW	21
Italy, 1971, nuclear power station		
^{60}Co	Max. 0.06 FW	23
^{137}Cs	Max. 0.33 FW	23

[a] 1, U.S. Environmental Protection Agency (EPA) 1982b; 2, International Atomic Energy Agency (IAEA) 1976; 3, Handl et al. 1990; 4, EPA 1977; 5, EPA 1989c; 6, EPA 1990a; 7, EPA 1990c; 8, EPA 1991; 9, EPA 1982a; 10, Clulow et al. 1991; 11, Joshi 1991; 12, EPA 1975; 13, EPA 1979; 14, EPA 1990a; 15, EPA 1989a; 16, EPA 1989b; 17, EPA 1989d; 18, EPA 1990b; 19, EPA 1990d; 20, Emery et al. 1976; 21, Wrenn et al. 1971; 22, Dahlman and Voris 1976; 23, Smedile and Queirazza 1976.

[b] — = no data.

[c] ND = not detectable.

Water in the Great Lakes in 1981 contained measurable concentrations of ^{137}Cs, ^{3}H, and ^{90}Sr and detectable but extremely low concentrations of ^{241}Am, ^{113m}Cd, ^{144}Ce, ^{210}Pb, $^{239+240}Pu$, ^{226}Ra, ^{125}Sb, and ^{228}Th (Joshi 1991). Radiocesium-137 in water from the Hudson River estuary, New York, decreased tenfold between 1964 and 1970, but the ^{137}Cs content in fishes and in sediments remained relatively constant (Wrenn et al. 1971). The effluent from the United Kingdom's Atomic Energy Agency Sellafield facility on the Cumberland Coast of the Irish Sea contained ^{90}Sr and ^{137}Cs, which are soluble in seawater and tend to remain in solution, and ^{106}Ru, ^{144}Ce, and $^{95}Zr/^{95}Nb$, which are relatively insoluble in seawater and coprecipitate or adsorb on free inorganic and organic surfaces (Pentreath et al. 1971).

Soils in the vicinity of an English nuclear-fuel reprocessing facility in 1979-85 contained as much as 42 times more ^{241}Am, 12 times more ^{137}Cs, 13 times more ^{90}Sr, and 87 times more $^{239+240}Pu$ than soils from a reference site (Curtis et al. 1991). In the United States, radiological trends in abiotic materials were difficult to interpret. For example, one nationwide monitoring program for radionuclide concentrations in air, drinking water, milk, groundwater, and precipitation (Table 13) was not consistent in the selection of measured radionuclides, frequency of sampling, and types of analyzed samples.

Aquatic Ecosystems

Field studies indicated that effects of radiation on marine ecosystems cannot be demonstrated at prevailing dose rates (Templeton et al. 1971).Two major periods of worldwide fallout occurred in Arctic ecosystems. The first and most sustained occurred during 1953-59 and the second during 1961-64, reflecting the atmospheric nuclear-weapons test regimes of Great Britain, the former Soviet Union, and the United States (Hanson 1976). Military accidents created localized radiocontamination of the Arctic environment. In one case, a B-52 aircraft from the U.S. Air Force crashed on the ice in northwestern Greenland in January 1968. Plutonium from the nuclear weapons on board contaminated the benthos (Fig. 5). The $^{239+240}Pu$ concentrations in various environmental samples declined at a much faster rate than the physical half-life of ^{239}Pu (24,000 years), suggesting that Pu becomes increasingly unavailable to the benthos over time as a result of dispersion from the epicenter and a dilution effect (Aarkrog 1990).

In marine environments, the major portion of the background dose rate in plankton and fishes arises from the incorporated activity of natural alpha emitters, such as ^{210}Po , and from ^{40}K; in molluscs, crustaceans, and benthos, the gamma radiation from the seabed provides the major background dose (IAEA 1976). The situation is similar in freshwater environments, although water that contains appreciable levels of ^{222}Rn and its daughter radionuclides may exert an additional burden, especially to phytoplankton. Artificial radionucludes that contribute significantly to background concentrations of marine organisms include ^{239}Pu and ^{90}Sr; of freshwater organisms, ^{137}Cs and ^{90}Sr (IAEA 1976). The total natural radiation received by a marine flounder (*Pleuronectes platessa*) in the Irish Sea consisted of 63% from radiations from seabed sediments, 16% from ^{40}K in seawater, 15% from internal ^{40}K and 6% from cosmic radiation (Templeton et al. 1971). The estimated dose rates in aquatic environments from natural background are as high as 3.5 mGy annually and of the same order as those in most terrestrial environments. By 1976, the estimated dose rates from global fallout had declined to the same range as natural dose rates, although environments that receive radioactive wastes had variable responses (IAEA 1976).

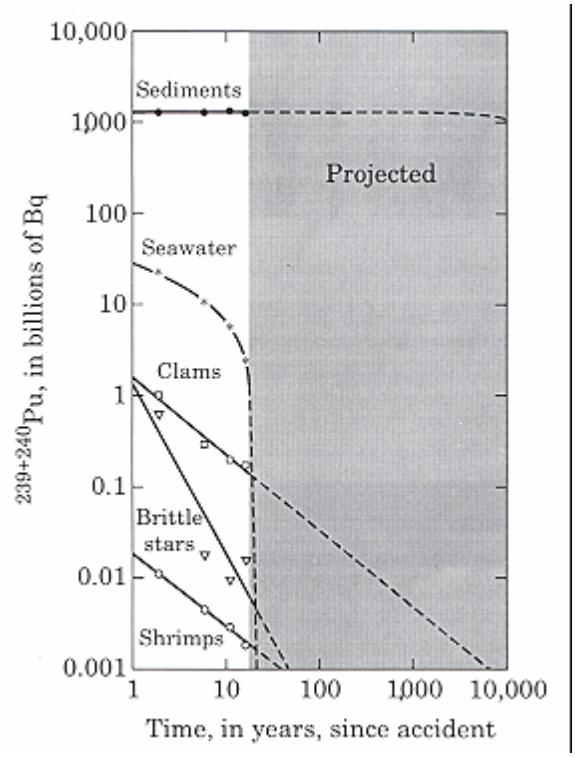

Fig. 5. Plutonium-239 + 240 in environmental samples at Thule, Greenland, between 1970 and 1984, after a military accident in 1968 (modified from Aarkrog 1990). In the contaminated area of 3.2 x 10^9 m^2, the fresh

weight biomass of shrimps was 0.11 x 10^9 kg, of brittle star echinoderms 0.06 x 10^9 kg, and of clam (*Macoma balthica*) soft parts 0.32 x 10^9 kg. The seawater mass was 3 x 10^{14} kg, and the dry weight of the upper 15 cm sediment layer was 3 x 10^{11} kg.

Increasing levels of ^{137}Cs in fish muscles in Minnesota between 1954 and 1966 reflect fallout from atmospheric nuclear testing. The effective half-life of ^{137}Cs in these lakes, as judged from small fishes, is about 30 months (Gustafson 1969). In game fish from Colorado, ^{137}Cs in muscle was as much as 7 times higher in 1968 than in 1965; higher in fishes in mountain lakes than in fishes from reservoirs in the plains, foothills, lakes, and rivers; and highest in trout from alpine lakes and reservoirs (Nelson and Whicker 1969). In 1966, ^{137}Cs levels in trout from Colorado alpine lakes were 8 to 18 times higher than mean levels in muscle of deer from Colorado during the same period and 20 to 300 times higher than in domestic meat products (Nelson and Whicker 1969). Radionuclides in livers of tunas from southern California during 1964-70 originated mainly from weapons tests in 1961-62, although ^{65}Zn may have reached southern California waters from nuclear reactors in Hanford (Washington) and from French or Chinese nuclear tests (Folsom et al. 1971).

Many variables modify radionuclide concentrations in biota. In general, lower trophic levels of aquatic organisms usually have greater concentrations of radionuclides than higher trophic levels (Bowen et al. 1971). However, radionuclide concentrations in biota are modified significantly by the organism's age, size, sex, tissue, season of collection, and other variables--and these have to be acknowledged when integrating radiological analyses. For example, older *Fucus vesiculosus* had higher radioactivity concentrations than younger algae; concentrations of ^{60}Co and ^{54}Mn were highest in older parts of plants during spring and summer; and ^{137}Cs and ^{40}K were highest in receptacles and new vegetative fronds (Carlson and Erlandsson 1991). Changes in concentrations of ^{60}Co and ^{137}Cs in freshwater plankton from the discharge canal of an Italian nuclear power station seem to reflect changes in water concentrations of these isotopes; changes were lowest in winter and highest in summer (Smedile and Queirazza 1976). Marine bivalve molluscs and algae from Connecticut in 1960 and 1961 had the highest levels of gross beta radioactivity in spring and summer and the lowest in winter (Table 14; Hatfield et al. 1963); natural ^{40}K probably accounted for most of the beta radioactivity. Similar seasonal variations in gross beta radioactivity in other species of marine algae and molluscs were documented, suggesting a correspondence with periods of dormancy and activity (Hatfield et al. 1963). Although fat in the livers of crabs accounted for 47% of the fresh weight (74% on a dry weight basis), the gross beta activity of the fat fraction amounted to less than 0.5% of the total radioactivity, suggesting that radiological liver analyses be conducted on the basis of nonfat solids (Chakravarti and Eisler 1961). In some locations of a ^{137}Cs-contaminated reservoir, males of the mosquitofish (*Gambusia holbrooki*) contained higher ^{137}Cs concentrations than females and smaller females contained more ^{137}Cs than larger females (Newman and Brisbin 1990). Strontium-90 concentrations in the carapace bone of turtles from 5 southwestern states in 1970 were used as indicators of ^{90}Sr fallout. However, older turtles tended to have lower concentrations of ^{90}Sr, and concentrations differed geographically; concentrations were highest in Georgia and increasingly lower in Tennessee, Mississippi, Arkansas, and Florida (Table 14; Holcomb et al. 1971).

Table 14. Radionuclide concentrations in field collections of selected living organisms. Concentrations are in becquerels per kilogram fresh weight (FW), dry weight (DW), or ash weight (AW).

Table 14. Ecosystem, organism, radionuclide, and other variables	Concentration (in Bq/kg[a])	Reference[b]
Terrestrial plants		
Sweet potato, *Ipomoea batatas*, Nagasaki, Japan, 1945, postatomic detonation		
^{137}Cs	0.09 DW	1
$^{239+240}Pu$	0.01 DW	1
Lichens, various species, Alaska and Greenland		
^{238}Pu		
1971 vs. 1972	0.4 DW vs. 0.9 DW	2
1973	0.4 DW	2
$^{239+240}Pu$		
1971 vs. 1972	7.4 DW vs. 10.3 DW	2
1973 vs. 1974	5.4 DW vs. 9.6 DW	2
Reed canarygrass, *Phalaris arundinacea*, Columbia River Washington, 1985-87, near reactor, ^{90}Sr	Max. 1,480-1,850 DW	3
Largetooth aspen, *Populus grandidentata*, ^{226}Ra		
Near uranium tailing plant vs. control site		
Leaves	53 DW vs. 4 DW	4
Stems	99 DW vs. 5 DW	4
Elliot Lake, Canada, 1984-87 vs. control site		
Leaves	252 DW vs. 46 DW	5
Stems	223 DW vs. 4 DW	5
Trembling aspen, *Populus tremuloides*, ^{226}Ra		
Near uranium tailings plant vs. control site		
Leaves	42 DW vs. 11-15 DW	4
Stems	69 DW vs. 3-11 DW	4
Vegetation		
Belgium, near former nuclear fuel reprocessing plant, 1983, ^{129}I	Max. 0.09 FW	6
California, deer forage plants, three spp., 1968-69, ^{137}Cs	414-514 DW	7
Colorado, mule deer diet, all plants, ^{90}Sr		
1962-63 vs. 1963-64	2,242 AW vs. 4,499 AW	8
1964-65 vs. 1965-66	3,492 AW vs. 2,257 AW	8

Table 14. Ecosystem, organism, radionuclide, and other variables	Concentration (in Bq/kg[a])	Reference[b]
Colorado, mule deer diet, eight species of forage plants, ^{90}Sr, 1963-64 vs. 1964-65	1,258-17,412 AW vs. 828-16,620 AW	8
Finland, reindeer forage plants, 1961, Lapland		
Lichen, *Cladonia alpestris*		
^{137}Cs	466,200 AW	11
^{90}Sr	53,428 AW	11
Lichen mixture		
^{137}Cs	133,200 AW	11
^{90}Sr	19,980 AW	11
Other forage plants		
^{137}Cs	962-8,800 AW	11
^{90}Sr	266-1,924 AW	11
Florida, April 1969, ^{137}Cs	Max. 0.65 DW	12
Georgia, deer browse, 29 species, 1965-66		
^{144}Ce	Max. 373 DW	13
^{60}Co	Max. 15 DW	13
^{137}Cs	Max. 104 DW	13
^{54}Mn	Max. 118 DW	13
^{106}Ru	Max. 226 DW	13
^{125}Sb	Max. 56 DW	13
^{90}Sr	Max. 377 DW; max. 2,005 AW	13
^{95}Zr	Max. 63 DW	13
^{65}Zn	Max. 22 DW	13
Tennessee, 1974, ^{137}Cs, from soil accidentally contaminated in 1944		
Roots	Usually 3,700 DW; max. 111,000 DW	14
Trees	74-5,920 DW	14
Ground vegetation	592-3,996 DW	14
Ginger, *Zingiber officinale*, root, Nagasaki, Japan, 1945, postatomic detonation		
^{137}Cs	0.07 DW	1
$^{239+240}Pu$	0.04 DW	1
Aquatic plants		
Algae, decomposing; Hanford, Washington, 1973; plutonium processing pond		
^{241}Am	9,472 DW	15
^{238}Pu	36,482 DW	15
$^{239+240}Pu$	22,755 DW	15

Table 14. Ecosystem, organism, radionuclide, and other variables	Concentration (in Bq/kg[a])	Reference[b]
Algae and macrophytes, ^{137}Cs, Hudson River, 1970	1.5-5.6 FW	16
Algae, South Carolina, 1971-72, reactor discharge, ^{137}Cs	12,284 DW	17
Brown algae, *Fucus vesiculosus*	3.2 DW vs. 0.09 DW	18
Ireland, 1985-86, $^{239+240}Pu$, northeast coast vs. western seaboard		
Sweden, 1984, vicinity of nuclear plant		
^{58}Co	20-23 DW	19
^{61}Co	1,700-2,003 DW	19
^{137}Cs	7-16 DW	19
^{40}K	735-966 DW	19
^{54}Mn	36-60 DW	19
^{65}Zn	90-144 DW	19
Seaweed, *Porphyra* sp., 1974, Cumbrian coast UK, <2 km from beach		
^{241}Am	458 FW	20
^{242}Cm	18 FW	20
^{238}Pu	37 FW	20
$^{239+240}Pu$	162 FW	20
Sea lettuce, *Ulva lactuca*, whole, Connecticut, 1960, gross beta activity		
May	5,402-6,253 AW	21
August	5,291-8,066 AW	21
December	2,183-3,700 AW	21
Aquatic invertebrates		
Clams, 15 species, freshwater, 1960, Tennessee River, near Oak Ridge, ^{90}Sr, shell	15-921 AW	23
Connecticut, 1960, gross beta activity		
American oyster, *Crassostrea virginica*, soft parts		
May	2,553-3,589 AW	21
August	2,775-4,551 AW	21
December	851-1,850 AW	21
Mussel, *Mytilus edulis*, soft parts, August vs. December	3,256-4,551 AW vs. 3,034-3,108 AW	21
Crabs, Hudson River, 1970, ^{137}Cs	0.6 FW	16
Crustaceans, marine; fallout radionuclides, typical values		

Table 14. Ecosystem, organism, radionuclide, and other variables	Concentration (in Bq/kg[a])	Reference[b]
^{110m}Ag	37.0 FW	24
^{60}Co	24.1 FW	24
^{54}Mn	2.2 FW	24
^{65}Zn	2.6 FW	24
Crustaceans, marine; natural radionuclides, typical values		
^{14}C	22.2 FW	24
^{3}H	0.1 FW	24
^{40}K	92.5 FW	24
^{210}Pb	2.2 FW	24
^{210}Po	37.0 FW	24
^{87}Rb	1.5 FW	24
Molluscs, bivalves, Hudson River, 1970, ^{137}Cs, soft parts	3 FW	16
Molluscs, freshwater; fallout radionuclides, typical values		
^{14}C	4-11 FW	24
^{3}H	0.1-159 FW	24
^{54}Mn	4-518 FW	24
Molluscs, marine; fallout radionuclides, typical values		
$^{141+144}Ce$	5-1,813 FW	24
^{57}Co	2-16 FW	24
^{60}Co	1-26 FW	24
^{137}Cs	5-25 FW	24
^{55}Fe	14-5,180 FW	24
^{54}Mn	2-222 FW	24
^{63}Ni	1-555 FW	24
^{239}Pu	Max. 0.02 FW	24
$^{103+106}Ru$	1-518 FW	24
^{110m}Ag	0.1-155 FW	24
^{65}Zn	0.7-425 FW	24
$^{95}Zr/^{95}Nb$	3-925 FW	24
Molluscs, marine; natural radionuclides		
^{14}C	18 FW	24
^{3}H	0.1 FW	24
^{40}K	107 FW	24
^{210}Pb	0.3 FW	24
^{210}Po	25 FW	24
^{87}Rb	2 FW	24
Mussel, *Mytilus edulis*, soft parts		

Table 14. Ecosystem, organism, radionuclide, and other variables	Concentration (in Bq/kg[a])	Reference[b]
Irish coastal waters, August 1988		
^{134}Cs	<0.7 DW	18
^{137}Cs	Usually <3 DW; max. 9 DW	18
^{40}K	182-355 DW	18
^{238}Pu	Usually <0.003 DW; max. 0.21 DW	18
$^{239+240}Pu$	Usually <0.035 DW; max. 1 DW	18
England, 1986-87, near nuclear plant		
^{110m}Ag	13 FW	25
^{241}Am	9-15 FW	25
^{144}Ce	6 FW	25
^{60}Co	3-7 FW	25
^{134}Cs	11 FW	25
^{137}Cs	5-31 FW	25
^{40}K	25-188 FW	25
^{95}Nb	3-106 FW	25
^{103}Ru	4-169 FW	25
^{106}Ru	64-151 FW	25
^{95}Zr	3-36 FW	25
Plankton, ^{137}Cs, Hudson River, 1970	2 FW	16
Plankton, Italy, 1971, near nuclear power station		
^{60}Co	Max. 203 FW	26
^{137}Cs	Max. 1,113 FW	26
Plankton, marine; fallout radionuclides, typical values		
$^{141+144}Ce$	14-17,760 FW	24
^{57}Co	85 FW	24
^{60}Co	11-592 FW	24
^{137}Cs	18-1,332 FW	24
^{155}Eu	14 FW	24
^{54}Mn	196 FW	24
^{63}Ni	4-14 FW	24
^{147}Pm	122 FW	24
$^{103+106}Ru$	11-1,110 FW	24
^{125}Sb	33 FW	24
^{90}Sr	0.7-12 FW	24
$^{95}Zr/^{95}Nb$	74-29,600 FW	24
Plankton, marine; natural radionuclides, typical values		
^{14}C	11 FW	24
^{3}H	0.1 FW	24

Table 14. Ecosystem, organism, radionuclide, and other variables	Concentration (in Bq/kg[a])	Reference[b]
^{40}K	92 FW	24
^{210}Pb	9-25 FW	24
^{210}Po	22-62 FW	24
^{226}Ra	0.7 FW	24
^{228}Th	0.4-2 FW	24
^{234}U	0.7-2 FW	24
^{235}U	0.02-0.07 FW	24
^{238}U	0.7-2 FW	24
Polychaete annelid worms, marine; England, 1984-86; near nuclear plant vs. control location		
Arenicola marina		
^{137}Cs	132-321 FW vs. 3 FW	25
^{40}K	162-307 FW vs. 90 FW	25
^{238}Pu	14-16 FW vs. <0.05 FW	25
$^{239+240}Pu$	60-72 FW vs. 0.01 FW	25
Nereis diversicolor		
^{137}Cs	41-358 FW vs. 6 FW	25
^{40}K	23-148 FW vs. 134 FW	25
^{238}Pu	6-11 FW vs. <0.02 FW	25
$^{239+240}Pu$	25-48 FW vs. 0.03 FW	25
Clam, *Rangia cuneata*, Neuse River, North Carolina, 1965-67, soft parts; before Chinese nuclear tests in May and December 1966 vs. posttest		
^{144}Ce	5.3 FW vs. 7.2 FW	27,28
^{137}Cs	1.0 FW vs. 1.6 FW	27,28
^{55}Fe	0.12 FW vs. 0.75 FW	27,28
^{54}Mn	2.5 FW vs. 2.7 FW	27,28
^{106}Ru	2.1 FW vs. 2.7 FW	27,28
^{65}Zn	0.4 FW vs. 0.8 FW	27,28
Sea urchin, *Strongylocentrotus purpuratus*, 1966		
^{210}Pb	Max. 2 AW	29
^{210}Po	Max. 7 AW	29
Fishes		
Goldfish, *Carassius auratus* from plutonium processing waste pond, Hanford, Washington, 1973		
^{241}Am, whole vs. muscle	399 DW vs. 14 DW	15
$^{238+239+240}Pu$, whole vs. muscle	351 DW vs. 10 DW	15

Table 14. Ecosystem, organism, radionuclide, and other variables	Concentration (in Bq/kg[a])	Reference[b]
Colorado, 1965-66, ^{137}Cs, muscle, maximum values		
Cutthroat trout, *Oncorhynchus clarki*	59 FW	30
Rainbow trout, *Oncorhynchus mykiss*	117 FW	30
Sockeye (kokanee) salmon, *Oncorhynchus nerka*	8 FW	30
Brook trout, *Salvelinus fontinalis*	215 FW	30
Lake trout, *Salvelinus namaycusch*	25 FW	30
Brown trout, *Salmo trutta*	121 FW	30
Columbia River, Washington; near nuclear facility, 1961, ^{239}Np, muscle		
Chiselmouth, *Acrocheilus alutaceus*	Max. 14,900 FW	31
Bridgelip sucker, *Catostomus columbianus*	Max. 5,600 FW	31
Largescale sucker, *Catostomus macrocheilus*	Max. 3,600 FW	31
Mountain whitefish, *Prosopium williamsoni*	Max. 18,800 FW	31
Freshwater fish, whole body; fallout radionuclides, typical values		
^{14}C	4-7 FW	24
^{137}Cs	1-973 FW	24
^{55}Fe	1-3 FW	24
^{3}H	0.1-159 FW	24
^{54}Mn	11 FW	24
^{85}Sr	0.04-0.4 FW	24
^{89}Sr	0.2-40 FW	24
^{90}Sr	0.04-177 FW	24
$^{95}Zr/^{95}Nb$	2.2-2.6 FW	24
Freshwater fishes, 1963-64, ^{210}Pb, bone vs. soft tissues	2.5 AW vs. 0.2 AW	29
Freshwater fishes, whole body, ^{137}Cs; Red Lakes, Minnesota		
1954-57	0.7-2.4 FW	32
1959-62	3-12 FW	32
1963-66	8-22 FW	32
Freshwater fishes, typical maximum concentrations, whole body		
^{3}H	0.5 FW	24
^{40}K	130 FW	24

Table 14. Ecosystem, organism, radionuclide, and other variables	Concentration (in Bq/kg[a])	Reference[b]
^{87}Rb	8 FW	24
^{238}U	0.1 FW	24
^{234}U	0.2 FW	24
^{226}Ra	129 FW	24
^{210}Pb (bone)	3 FW	24
^{210}Po (liver)	18 FW	24
^{232}Th	0.05 FW	24
^{235}U	0.004 FW	24
Mosquitofish, *Gambusia holbrooki*, ^{137}Cs, April 1987; from South Carolina reservoir contaminated with ^{137}Cs between 1961 and 1964, whole body	Max. 2,230 FW	33
Hudson River, 1970, ^{137}Cs		
Atlantic sturgeon, *Acipenser oxyrhynchus* muscle	0.6 FW	16
American eel, *Anguilla rostrata*, muscle	1.3 FW	16
Mummichog, *Fundulus heteroclitus*, whole	2.0 FW	16
Catfish, *Ictalurus* sp., muscle	1.9 FW	16
White perch, *Morone americana*, muscle vs. whole body	0.8 FW vs. 0.8 FW	16
Striped bass, *Morone saxatilis*, muscle	0.9 FW	16
Yellow perch, *Perca flavescens*, muscle	1.5 FW	16
Italy, 1971, near nuclear power station, whole fish, various species		
^{60}Co	Max. 9 DW	26
^{137}Cs	Max. 104 DW	26
Lake Ontario, ^{137}Cs, 1981		
Common carp, *Cyprinus carpio*, bone vs. other tissues	5 FW vs. <5 FW	34
Northern pike, *Esox lucius*		
Bone, liver	5 FW	34
Roe	15 FW	34
Other tissues	<5 FW	34
Coho salmon, *Oncorhynchus kisutch*		
GI tract	5 FW	34
Liver	13 FW	34
Other tissues	<5 FW	34
Largemouth bass, *Micropterus*	3,677 DW	17

Table 14. Ecosystem, organism, radionuclide, and other variables	Concentration (in Bq/kg[a])	Reference[b]
salmoides, South Carolina, reactor discharge, 1971-72, ^{137}Cs, whole		
Marine fishes, whole body, fallout radionuclides, typical values		
^{110m}Ag	2-3 FW	24
$^{141+144}Ce$	2-1,036 FW	24
^{60}Co	1-13 FW	24
^{137}Cs	2-3 FW	24
^{55}Fe		
Gonad	8,140-10,360 FW	24
Liver	59,940-68,820 FW	24
Muscle	37-3,922 FW	24
^{54}Mn	0.07-2 FW	24
^{239}Pu	Max. 0.005 FW	24
$^{103+106}Ru$	2-244 FW	24
$^{95}Zr/^{95}Nb$	1-277 FW	24
^{65}Zn	2-7 FW	24
Marine fishes, whole body, natural radionuclides, typical values		
^{14}C	15 FW	24
^{3}H	0.1 FW	24
^{40}K	92 FW	24
^{210}Pb	5 FW	24
^{210}Po	33 FW	24
^{226}Ra	0.2 FW	24
^{87}Rb	1 FW	24
^{234}U	1 FW	24
^{235}U	0.05 FW	24
^{238}U	1 FW	24
Golden shiner, *Notemigonus crysoleucas*, whole, ^{137}Cs, Hudson River estuary		
1966 vs. 1968	0.9 FW vs. 0.8 FW	16
1969 vs. 1970	0.7 FW vs. 0.5 FW	16
Oceanic fishes, 1962-64, bone vs. soft parts		
^{210}Pb	10 AW vs. 0.06 AW	29
^{210}Po	12 AW vs. 0.1 AW	29
^{226}Ra	2 AW vs. 0.06 AW	29
Plaice, *Pleuronectes platessa*, near nuclear facility, England		
1968 vs. 1969, ^{137}Cs		

Table 14. Ecosystem, organism, radionuclide, and other variables	Concentration (in Bq/kg[a])	Reference[b]
Gut contents	44-181 FW vs. 126-266 FW	35
Muscle	26-70 FW vs. 89-152 FW	35
1968, gut contents		
^{144}Ce	880-1,150 FW	35
^{106}Ru	1,343-5,143 FW	35
$^{95}Zr/^{95}Nb$	3,122-5,794 FW	35
Albacore, *Thunnus alalunga* southern California near San Diego, liver		
Summer 1964 vs. summer 1965		
^{110m}Ag	3 FW vs. 4 FW	36
^{60}Co	7 FW vs. 7 FW	36
^{40}K	71 FW vs. 72 FW	36
^{54}Mn	39 FW vs. 22 FW	36
^{65}Zn	46 FW vs. 14 FW	36
Summer 1968 vs. summer 1970		
^{60}Co	2 FW vs. 2 FW	36
^{40}K	81 FW vs. 78 FW	36
^{54}Mn	2 FW vs. 0.6 FW	36
^{65}Zn	25 FW vs. 9 FW	36
Yellowfin tuna, *Thunnus albacares*, 1968, near San Diego, liver		
^{60}Co	1 FW	36
^{40}K	93 FW	36
^{54}Mn	1 FW	36
^{65}Zn	3 FW	36
Tunas, 1970-71, Hawaii, liver		
^{108m}Ag	0.03-2 FW	36
^{110m}Ag	0.01-7 FW	36
^{60}Co	0.9-3 FW	36
^{40}K	68-83 FW	36
^{65}Zn	5-27 FW	36
Reptiles		
Snakes, two species (*Elaphe obsoleta*, *Nerodia taxispilota*), Aiken, South Carolina; whole animal, ^{137}Cs		
Site contaminated with ^{137}Cs between 1961 and 1970, *Elaphe vs*. *Nerodia*		
1972	6,037 FW vs. 7,629 FW	58
1976	592 FW vs. 1,333 FW	58
1980	296 FW vs. 1,037 FW	58
Uncontaminated site, both species, 1972-80	<37 FW	58
Snakes, 19 species, whole,		

Table 14. Ecosystem, organism, radionuclide, and other variables	Concentration (in Bq/kg[a])	Reference[b]
vicinity of Aiken, South Carolina, March 1971-November 1972, $^{134+137}Cs$		
Near reactor effluent stream	4,870 FW, max. 38,200 FW	9
Near reactor cooling reservoir	1,025 FW, max. 5,159 FW	9
Uncontaminated habitats	92 FW	9
Slider turtle, *Trachemys scripta*, from radioactive reservoirs, Aiken, South Carolina, whole body		
High-level waste pond vs. low-level waste pond		
^{137}Cs	3,020 FW vs. 1,002 FW	37
^{90}Sr	94,030 FW vs. 2,236 FW	37
Control sites		
^{137}Cs	0.001 FW	37
^{90}Sr	0.2 FW	37
Turtles, southeastern USA, 1970, ^{90}Sr, exoskeleton		
Snapping turtle, *Chelydra serpentina*	784 (284-1,283) AW	38
Gopher tortoise, *Gopherus polyphemus*	4,765 AW	38
Common mud turtle, *Kinosternon sabrubrum*	1,309 (569-2,904) AW	38
Missouri slider, *Pseudemys floridana hoyi*	1,761 AW	38
Peninsula cooter, *Pseudemys floridana peninsularis*	33 (ND-48) AW	38
Pond slider, *Pseudemys scripta*	777 (188-2,190) AW	38
Loggerhead musk turtle, *Sternotherus minor minor*	24 (ND-48) AW	38
Common musk turtle, *Sternotherus odoratus*	525 (52-999) AW	38
Common box turtle, *Terrapene carolina*	1,087 (48-2,856) AW	38
Birds		
Ruffed grouse, *Bonasa umbellus*; near uranium tailings discharge, Canada, Elliot Lake, 1987-88, ^{226}Ra		
Bone vs. gut contents	10-28 DW vs. 7-22 DW	4
Liver vs. muscle	5-12 DW vs. 1.5-1.9 DW	4
Canada goose, *Branta canadensis moffitti*; Columbia River, Washington, 1985-87, near reactor; eggshell, ^{90}Sr	18-60 DW	39
American coot, *Fulica americana*; Hanford, Washington, June 1974-January 1977		

Table 14. Ecosystem, organism, radionuclide, and other variables	Concentration (in Bq/kg[a])	Reference[b]
^{137}Cs (Hanford vs. control ponds)		
Bone	7,400 vs. 37 DW	40
Gut contents	125,800 vs. 29 DW	40
Liver	16,280 vs. 26 DW	40
Muscle	21,090 vs. 0.7 DW	40
^{90}Sr (Hanford only)		
Bone	96 DW	40
Gut contents	159 DW	40
Liver	18 DW	40
Muscle	10 DW	40
Barn swallow, *Hirundo rustica*; Idaho, 1976-77, nesting near radioactive leaching ponds		
Whole adults		
^{140}Ba	800 FW	41
^{134}Cs	1,300 FW	41
^{137}Cs	6,400 FW	41
^{51}Cr	16,100 FW	41
^{60}Co	1,480 FW	41
^{131}I, whole vs. thyroid	5,500 FW vs. 3,330,000 FW	41
^{24}Na	8,600 FW	41
^{75}Se	5,000 FW	41
^{65}Zn	5,900 FW	41
Nests		
^{140}Ba	1,200 DW	41
^{134}Cs	13,800 DW	41
^{137}Cs	92,000 DW	41
^{141}Ce	1,200 DW	41
^{144}Ce	4,000 DW	41
^{51}Cr	230,000 DW	41
^{131}I	800 DW	41
^{65}Zn	1,800 DW	41
Massachusetts, 1973-75, 15 passerine species, trapped near nuclear power station, whole body		
Northern bobwhite, *Colinus virginianus*		
^{137}Cs	Max. 73 FW	42
^{131}I	Max. 6 FW	42
^{40}K	Max. 131 FW	42
$^{95}Zr/^{95}Nb$	Max. 4 FW	42
Blue jay, *Cyanocitta cristata*		
^{137}Cs	28 FW; max. 65 FW	42
^{131}I	1 FW; max. 9 FW	42

Table 14. Ecosystem, organism, radionuclide, and other variables	Concentration (in Bq/kg[a])	Reference[b]
^{40}K	96 FW; max. 181 FW	42
$^{95}Zr/^{95}Nb$	2 FW; max. 6 FW	42
13 species		
^{137}Cs	Max. 82 FW	42
^{131}I	Max. 18 FW	42
^{40}K	Max. 268 FW	42
$^{95}Zr/^{95}Nb$	Max. 40 FW	42
United Kingdom, Ravenglass estuary, 1980-84, near nuclear plant		
Mallard, *Anas platyrhynchos*		
^{134}Cs, muscle	87 FW	25
^{137}Cs, muscle vs. liver	167 FW vs. 126 FW	25
$^{239+240}Pu$, liver	3.4 FW	25
^{238}Pu, liver	1.1 FW	25
Greylag goose, *Anser anser*		
^{137}Cs, muscle vs. liver	58 FW vs. 28 FW	25
^{238}Pu, muscle vs. liver	0.03 FW vs. 3 FW	25
$^{239+240}Pu$, muscle vs. liver	0.1 FW vs. 13 FW	25
Carrion crow, *Corvus corone*		
^{137}Cs, Ravenglass vs. control location		
Muscle	162 FW vs. 17 FW	25
Liver	131 FW vs. 8 FW	25
Lesser black-backed gull, *Larus marinus*		
^{137}Cs, muscle vs. liver	158 FW vs. 163 FW	25
$^{239+240}Pu$, muscle vs. liver	0.1 FW vs. 5 FW	25
Common black-headed gull, *Larus ridibundus*, whole chick		
^{134}Cs	0.8 FW	25
^{137}Cs	25 FW	25
^{238}Pu	0.1 FW	25
$^{239+240}Pu$	0.5 FW	25
Eurasian oystercatcher, *Haematopus ostralegus*, Ravenglass vs. control location		
^{137}Cs		
Muscle	613 FW vs. 22 FW	25
Liver	463 FW vs. 20 FW	25
^{238}Pu		
Muscle	0.2 FW vs. <0.01 FW	25
Liver	1.8 FW vs. 0.04 FW	25
$^{239+240}Pu$		
Muscle	0.5 FW vs. 0.04 FW	25
Liver	4.1 FW vs. 0.09 FW	25
Bar-tailed godwit, *Limosa lapponica lapponica*		

Table 14. Ecosystem, organism, radionuclide, and other variables	Concentration (in Bq/kg[a])	Reference[b]
^{137}Cs, muscle vs. liver	478 FW vs. 510 FW	25
^{238}Pu, muscle vs. liver	<0.02 FW vs 0.2 FW	25
$^{239+240}Pu$, muscle vs. liver	0.03 FW vs. 0.9 FW	25
Red-breasted merganser, *Mergus serrator*		
^{134}Cs, muscle vs. liver	8 FW vs. 13 FW	25
^{137}Cs, muscle vs. liver	144 FW vs. 251 FW	25
^{238}Pu, muscle vs. liver	<0.01 FW vs. <0.04 FW	25
$^{239+240}Pu$, muscle vs. liver	0.02 FW vs <0.04 FW	25
Eurasian curlew, *Numenius arquata*		
^{137}Cs, Ravenglass vs. control location		
Muscle	140 FW vs. 49 FW	25
Liver	104 FW vs. 99 FW	25
^{238}Pu, Ravenglass vs. control location		
Muscle	0.09 FW vs. <0.02 FW	25
Liver	0.14 FW vs. <0.05 FW	25
$^{239+240}Pu$, Ravenglass vs. control location		
Muscle	0.09 FW vs. <0.02 FW	25
Liver	0.14 FW vs. <0.05 FW	25
Marine mammals		
Bearded seal, *Erignathus barbatus*; Alaska, 1963		
Bone		
^{210}Pb	Max. 2.7 AW	29
^{226}Ra	2.4 AW	29
Soft tissues, ^{210}Pb	Max. 0.2 AW	29
Gray seal, *Halichoerus grypus*, North Sea and northeast Atlantic Ocean, 1987		
Females, milk vs. muscle		
^{241}Am	<0.0002 FW vs. <0.0005 FW	43
^{134}Cs	0.6 (0.4-0.7) FW vs. <0.002 FW	43
^{137}Cs	2.9 (1.1-4.8) FW vs. 14.3 FW	43
^{40}K	107 (67-215) FW vs. 0.2 FW	43
^{238}Pu	<0.0002 FW vs. <0.0005 FW	43
$^{239+240}Pu$	<0.0002 FW vs. <0.0005 FW	43
Pup, muscle vs. liver		
^{241}Am	<0.0003 FW vs. <0.0003 FW	43
^{134}Cs	Max. 0.003 FW vs. max. 0.00 1 FW	43
^{137}Cs	Max. 0.03 FW vs. max. 0.02 FW	43
^{40}K	Max. 0.2 FW vs. max. 0.2 FW	43
^{238}Pu	Max. 0.0005 FW vs. max. 0.001 FW	43
$^{239+240}Pu$	Max. 0.002 FW vs. max. 0.004 FW	43
Spotted seal, *Phoca largha*; Alaska, 1963, bone vs. soft tissues		
^{210}Pb	2 AW vs. 0.1 AW	29

Table 14. Ecosystem, organism, radionuclide, and other variables	Concentration (in Bq/kg[a])	Reference[b]
^{226}Ra	3 AW vs. no data	29
Sperm whale, *Physeter catodon*; Alaska, 1965, bone vs. soft tissue		
^{210}Pb	135 AW vs. 0.37 AW	29
^{210}Po	114 AW vs. 23 AW	29
Terrestrial mammals		
Cattle, *Bos* sp.		
Nevada, 1973, grazing for 3 years in area contaminated in 1957 with transuranic radionuclides		
^{241}Am		
Bone vs. liver	Max. 1 FW vs. max. 0.6 FW	44
Lymph nodes vs. lungs	Max. 24 FW vs. max. 2 FW	44
Other tissues	<0.6 FW	44
^{238}Pu		
Lungs, lymph nodes	Max. 3 FW	44
Testes	Max. 0.8 FW	44
Other tissues	<0.6 FW	44
$^{239+240}Pu$		
Bone vs. liver	Max. 3 FW vs. max. 34 FW	44
Lungs vs. lymph nodes	Max. 34 FW vs. max. 85 FW	44
Muscle vs. other tissues	Max. 7 FW vs. <1.2 FW	44
Europe, ^{129}I, thyroids		
1978		
Belgium vs. Germany	0.017-3.7 FW vs. max. 0.03 FW	6
Italy vs. Netherlands	Max. 0.05 FW vs. max. 0.03 FW	6
1979, Netherlands	Max. 0.07 FW	6
1980, Netherlands	0.07-0.6 FW	6
1981, Germany	Max. 0.02 FW	6
Beaver, *Castor canadensis*; Canada, 1984-87, adults, ^{226}Ra; from watershed containing uranium tailings vs. control site		
Bone	115 DW vs. 20 DW	5
Gut contents	62 DW vs. 9 DW	5
Kidney	9 DW vs. 2DW	5
Liver	2.7 DW vs. 1.4 DW	5
Muscle	2.9 DW vs. 1.0 DW	5
Georgia and South Carolina, 1964-66, ^{137}Cs, whole organism		
Domestic dog, *Canis familiaris*	23 FW	45
Coyote, *Canis latrans*	26 FW	45
Bobcat, *Lynx rufus*	117-561 FW	45
Cotton rat, *Sigmodon hispidus*	16-29 FW	45
Eastern cottontail, *Sylvilagus floridanus*	19-35 FW	45
Gray fox, *Urocyon*	34-169 FW	45

Table 14. Ecosystem, organism, radionuclide, and other variables	Concentration (in Bq/kg[a])	Reference[b]
cinereoargentatus		
Red fox, *Vulpes fulva*	23-60 FW	45
Humans, *Homo sapiens*;		
Denmark, ^{137}Cs, annual dietary loading		
1964	71.9 FW	46
1985	1.4 FW	46
1986-87	12.6 FW	46
Black-tailed jack rabbit, *Lepus californicus*; Nevada test site, bone, ^{90}Sr		
1952-66	74-476 AW	47
1958 (1-year post detonation), ground zero vs. 32-700 km distant	373 AW vs. 88-198 AW	48
1959, ground zero	329 AW	48
1959, 32 km vs. 120-700 km	466 AW vs. 95-222 AW	48
1961, within 160 km of ground zero	143 AW	48
Mule deer, *Odocoileus hemionus*; 1961-65, Colorado, femur, ^{90}Sr		
1961-62 vs. 1962-63	Max. 215 AW vs. max. 528 AW	8
1963-64 vs. 1964-65	Max. 777 AW vs. max. 637 AW	8
Black-tailed deer, *Odocoileus hemionus columbianus*; California		
Muscle vs. rumen contents, 1968-69, ^{137}Cs		
Summer	37 DW vs. 48 DW	7
Fall	33 DW vs. 37 DW	7
Winter	48 DW vs. 67 DW	7
Mendocino County, California, mandible, yearlings, ^{90}Sr		
1952-53 vs. 1954	3-11 AW vs. 26-34 AW	49
1955 vs. 1956	29-124 AW vs. 112 AW	49
1957 vs. 1958	87-239 AW vs. 134-228 AW	49
1959 vs. 1960	243-533 AW vs. 204-332 AW	49
White-tailed deer, *Odocoileus virginianus* Georgia, 1965-66		
^{137}Cs		
Heart vs. kidney	127 FW vs. 149 FW	13
Liver vs. lung	70 FW vs. 73 FW	13
Muscle vs. spleen	126 FW vs. 126 FW	13
Tongue	172 FW	13
^{90}Sr, mandible		
Age 1.5 years	940 AW	13
Age 2.5 years	828 AW	13
Age 3.5 years	799 AW	13

Table 14. Ecosystem, organism, radionuclide, and other variables	Concentration (in Bq/kg[a])	Reference[b]
Southeastern United States		
^{137}Cs, Muscle, 1967-71		
Alluvial region (LA, MS, FL, SC, NC)	85 (9-650) FW	50
Lower Coastal Plain (SC, GA, FL, VA, NC)	1,036 (9-5,658) FW	50
Mountain region (WV, KY, MD, NC, TN, GA)	78 (9-401) FW	50
Piedmont region (GA, SC, AL)	105 (9-383) FW	50
Upper Coastal Plain region (MD, NC, GA, VA, MS, LA, AK)	154 (9-1,752) FW	50
^{90}Sr, bone, 1969		
Lower Coastal Plain	1,172 (376-1,766) FW	50
Mountain region	499 (148-888) FW	50
Piedmont region	471 (263-683) FW	50
Muskrat, *Ondatra zibethicus*; August 1960, Oak Ridge, Tennessee, from settling basin for radioactive wastes, single most radioactive animal		
Brain vs. eyes		
^{60}Co	10,545 DW vs. 39,960 DW	51
^{137}Cs	392,200 DW vs. 640,100 DW	51
^{65}Zn	21,016 DW vs. 36,593 DW	51
Femur		
^{60}Co	5,920 DW	51
^{137}Cs	121,360 DW	51
^{90}Sr	7,030,000 DW	51
^{65}Zn	28,601 DW	51
Kidney vs. spleen		
^{60}Co	279,720 DW vs.47,730 DW	51
^{137}Cs	954,600 DW vs. 799,200 DW	51
Liver		
^{60}Co	156,880 DW	51
^{137}Cs	629,000 DW	51
^{65}Zn	78,440 DW	51
Muscle		
^{60}Co	8,103 DW	51
^{134}Cs	13,949 DW	51
^{137}Cs	1,265,400 DW	51
^{65}Zn	19,610 DW	51
Teeth		
^{137}Cs	64,010 DW	51
^{90}Sr	9,916,000 DW	51
^{65}Zn	25,789 DW	51

Table 14. Ecosystem, organism, radionuclide, and other variables	Concentration (in Bq/kg[a])	Reference[b]
Pelt		
^{60}Co	15,022 DW	51
^{137}Cs	204,980 DW	51
^{90}Sr	37,000 DW	51
^{65}Zn	26,196 DW	51
Domestic sheep, *Ovis aries*		
Near nuclear fuel reprocessing facility vs. control site, England, 1983		
Bone		
^{241}Am	1 FW vs. 0.003 FW	52
$^{239+240}Pu$	0.6 FW vs. 0.002 FW	52
Liver		
^{241}Am	1 FW vs. 0.002 FW	52
^{137}Cs	8 FW vs. 0.2 FW	52
$^{239+240}Pu$	2 FW vs. 0.008 FW	52
Lung		
^{241}Am	0.3 FW vs. 0.003 FW	52
$^{239+240}Pu$	0.4 FW vs. 0.002 FW	52
Muscle		
^{241}Am	0.03 FW vs. 0.0005 FW	52
^{137}Cs	49 FW vs. 0.2 FW	52
$^{239+240}Pu$	0.007 FW vs. 0.0008 FW	52
Near nuclear fuel reprocessing plant, England, winter 1986-87		
^{241}Am		
Bone vs. liver	0.03-0.7 FW vs. 0.03-0.8 FW	53
Lung vs. muscle	0.009-0.1 FW vs. 0.002-0.03 FW	53
Whole sheep	0.27-4 FW	53
^{137}Cs		
Bone vs. liver	1.3-14 FW vs. 1.8-30 FW	53
Lung vs. muscle	1.5-16 FW vs. 4.6-42 FW	53
Whole sheep	159-748 FW	53
$^{239+240}Pu$		
Bone vs. liver	0.024-0.2 FW vs. 0.07-0.9 FW	53
Lung vs. muscle	0.005-0.02 FW vs. 0.0005-0.005 FW	53
Whole sheep	0.02-2 FW	53
Serbia, 1988, wildlife		
Roe deer, *Capreolus sp*.; bone vs. muscle		
^{137}Cs	ND vs. 0.2 AW	54
^{40}K	23 AW vs. 39 AW	54
^{90}Sr	6 AW vs. 0.6 AW	54
Fallow deer, *Dama sp*.; bone vs. muscle		

Table 14. Ecosystem, organism, radionuclide, and other variables	Concentration (in Bq/kg[a])	Reference[b]
^{137}Cs	ND vs. 0.1 AW	54
^{40}K	8 AW vs. 45 AW	54
^{90}Sr	10 AW vs. 0.3 AW	54
Wild hare, *Lepus* sp.; bone vs. muscle		
^{137}Cs	ND vs. 0.1 AW	54
^{40}K	26 AW vs. 52 AW	54
^{90}Sr	18 AW vs. ND	54
Wild boar, *Sus scrofa*; bone vs. muscle		
^{137}Cs	ND vs. 0.4 AW	54
^{40}K	21 AW vs. 56 AW	54
^{90}Sr	34 AW vs. 2 AW	54
Common shrew, *Sorex araneus*; 1988, England, muscle; shrews from mineral soils vs. peaty soils		
^{134}Cs	7 FW vs. 16 FW	55
^{137}Cs	58 FW vs. 161 FW	55
Integrated studies		
Brazil, site of radiological accident in September 1987 at Goiania wherein ^{137}Cs was deposited on soil for 3 weeks before remedial action. Rainwater runoff contaminated the waterways 3 weeks postaccident, up to 12 km from accident area, ^{137}Cs		
Fish muscle	Max. 200 FW	56
Sediments	Max. 1,300 DW	56
Surface waters and suspended particulates	<1 FW	56
10 months postaccident, up to 80 km downstream, ^{137}Cs		
Fish muscle		
Pike, *Hoplias* sp.	14 FW	56
Piranha, *Seerasalmus sp*.	10 FW	56
Sediments	100 DW	56
Water hyacinth, *Eichornia* sp.	Max. 0.4 FW	56
Great Lakes, ^{137}Cs, 1981		
Aquatic plants vs. clams	1.4 FW vs. 0.3 FW	34
Fish vs. plankton	1.5 FW vs. 0.1 FW	34
Sediments vs. water	24 FW vs. 0.0007 FW	34
Irish Sea and North Sea, 1983, invertebrates vs. fish		
^{241}Am	Max. 75 FW vs. max. 0.05 FW	57

Table 14. Ecosystem, organism, radionuclide, and other variables	Concentration (in Bq/kg[a])	Reference[b]
^{242}Cm	Max. 2 FW vs. max. 0.0003 FW	57
$^{243+244}Cm$	Max. 0.5 FW vs. 0.0003 FW	57
^{238}Pu	14 FW vs. 0.01 FW	57
$^{239+240}Pu$	54 FW vs. 0.04 FW	57
^{241}Pu, invertebrates only	Max. 1,000 FW	57
Japan, Nagasaki, 1945 post-atomic detonation		
Fish vs. snail		
^{137}Cs	0.01 DW vs. 0.02 DW	1
$^{239+240}Pu$	0.03 DW vs. 0.03 DW	1
South Carolina, watershed of a former reactor effluent stream, ^{137}Cs, 1971 vs. 1981		
Plants	14,000-19,000 DW vs. 2,600-9,600 DW	10
Arthropods	9,600-16,000 DW vs. 700-3,300 DW	10
South Carolina; reactor cooling impoundment accidentally contaminated in 1961-64 with ^{137}Cs, ^{90}Sr, and various transuranics; samples collected September 1983-February 1984		
^{137}Cs		
Water vs. sediments	0.76 FW vs. max. near 40,000 DW	22
Aquatic macrophytes vs. benthic invertebrates	Max. near 30,000 DW vs. 930-14,000 DW	22
Fish muscle	2,100-8,000 FW; 21,000 DW	22
Turtle muscle	2,100 FW	22
Waterfowl muscle	3,100 FW; 15,000 DW	22
^{90}Sr		
Water vs. sediments	0.14 FW vs. max. near 400 DW	22
Aquatic macrophytes vs. benthic invertebrates	Max. 2,600 DW vs. 42-7,900 DW	22
Fish bone ash vs. fish muscle	12,000-23,000 DW vs. 86-470 DW	22
Turtle shell and bone ash	12,000 DW	22
Waterfowl muscle vs. waterfowl bone ash	14 DW vs. 420 DW	22
^{238}Pu		
Water vs. sediments	0.0000034 FW vs. max. 10 DW	22
Aquatic macrophytes vs. fish muscle	Max. 0.5 DW vs. 0.004 DW	22
Turtle shell ash vs. waterfowl bone ash	0.1 DW vs. 100 DW	22
Waterfowl muscle	0.013 DW	22
$^{239+240}Pu$		
Water	0.0000088 FW	22

Table 14. Ecosystem, organism, radionuclide, and other variables	Concentration (in Bq/kg[a])	Reference[b]
Sediments	Max. near 85 DW	22
Aquatic macrophytes	Max. near 1.2 DW	22
Turtle shell ash	ND	22
Waterfowl muscle	0.008 DW	22
^{241}Am		
Water vs. sediments	0.000023 FW vs. max. 40 DW	22
Turtle shell ash vs. waterfowl muscle	ND vs. 0.015 DW	22
^{244}Cm		
Water vs. sediments	0.00064 FW vs. max. 18 DW	22
Fish liver vs. turtle shell ash	11 DW vs. 0.2 DW	22
Waterfowl muscle	0.071 DW	22

[a]Values originally expressed in strontium units (1 nCi ^{90}Sr/g calcium AW) were transformed to Bq/kg AW by a multiplication factor of 98.4.

[b]1, Kudo, et al. 1991; 2, Hanson 1976; 3, Rickard and Price 1990; 4, Clulow et al. 1992; 5, Clulow et al. 1991; 6, Handl et al. 1990; 7, Book 1969; 8, Farris et al. 1969; 9, Brisbin et al. 1974; 10, Brisbin et al. 1989; 11, Miettinen 1969; 12, Cummings et al. 1971; 13, Plummer et al. 1969; 14, Dahlman and Voris 1976; 15, Emery et al. 1976; 16, Wrenn et al. 1971; 17, Shure and Gottschalk 1976; 18, Crowley et al. 1990; 19, Carlson and Erlandsson 1991; 20, Hetherington et al. 1976; 21, Hatfield et al. 1963; 22, Whicker et al. 1990; 23, Nelson 1963; 24, IAEA 1976; 25, Lowe 1991; 26, Smedile and Queirazza 1976; 27, Wolfe and Schelske 1969; 28, Wolfe and Jennings 1971; 29, Holtzman 1969; 30, Nelson and Whicker 1969; 31, Poston et al. 1990; 32, Gustafson 1969; 33, Newman and Brisbin 1990; 34, Joshi 1991; 35, Pentreath et al. 1971; 36, Folsom et al. 1971; 37, Lamb et al. 1991; 38, Holcomb et al. 1971; 39, Rickard and Price 1990; 40, Cadwell et al. 1979; 41, Millard and Whicker 1990; 42, Levy et al. 1976; 43, S. S. Anderson et al. 1990; 44, Gilbert et al. 1989; 45, Jenkins et al. 1969; 46, Aarkrog 1990; 47, Romney et al. 1971; 48, Neel and Larson 1963; 49, Schultz and Longhurst 1963; 50, Jenkins and Fendley 1971; 51, Kaye and Dunaway 1963; 52, Curtis et al. 1991; 53, Ham et al. 1989; 54, Veskovic and Djuric 1990; 55, Lowe and Horrill 1991; 56, Godoy et al. 1991; 57, United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) 1988; 58, Bagshaw and Brisbin 1984.

[c]ND = non detectable.

Consumption of shellfish represents a negligible radiological risk to humans (Crowley et al. 1990), although bivalve molluscs seem to be effective accumulators of radioisotopes. After the Chinese nuclear tests in May and December 1966, concentrations of ^{144}Ce, ^{103}Ru, ^{95}Zr, ^{95}Nb, ^{140}Ba, and ^{140}La in three species of bivalves in the Neuse River, North Carolina, increased suddenly (Wolfe and Schelske 1969). In 1973, Pacific oysters (*Crassostrea gigas*) from the discharge canal of a nuclear power plant in Humboldt Bay, California, rapidly accumulated ^{54}Mn, ^{60}Co, ^{65}Zn, and ^{137}Cs within 30 min of release. Isotope uptake correlated positively with particulates in the water, including living microorganisms, organic detritus, inorganic materials, and especially resuspended bottom sediments (Harrison et al. 1976). Although concentrations of cesium and plutonium in mussels (*Mytilus edulis*) from most Irish estuaries are essentially the same as global fallout levels, concentrations were elevated in mussels from the northeast coast (Crowley et al. 1990).

Birds

Television and newspaper reporters attributed radionuclides to a decline in bird numbers at the Ravenglass estuary, England, particularly of the common black-headed gull (*Larus ridibundus*), although the concentrations of radionuclides in the avian diet, body tissues, and general environment were at least 1,000 times too low to have had any effect (Table 14; Lowe 1991). Although Eurasian oystercatchers (*Haematopus ostralegus*) and shelducks (*Tadorna tadorna*) had the highest concentrations of ^{137}Cs in their tissues, the breeding success and population sizes of these birds were not affected. Black-headed gulls had less radiocontamination than other

birds at Ravenglass, but their population continued to decline. The most probable cause was a combination of an uncontrolled fox population, a severe outbreak of myomatosis in rabbits (normal fox prey), and a drought--all in the same year (Lowe 1991). Nesting success of birds was unaffected in the vicinity of nuclear power plants. For example, nesting barn swallows (*Hirundo rustica*) near radioactive leaching ponds had normal nesting success despite their consumption of arthropods from the pond and use of contaminated mud for nest construction (Millard and Whicker 1990; Table 14). Adult swallows received a total internal dose rate of 219 uGy/day, mostly (72%) from ^{24}Na; daily dose rates to eggs and nestlings during the nesting season were 840 uGy and 2,200 uGy. The total dose to eggs and nestlings (54 mGy) and adults (450 mGy) had no measurable effect on survival and was below accumulated doses reported to cause death of passerines (Millard and Whicker 1990).

Strontium-90 behaves much like calcium in the biological environment. In birds, ^{90}Sr is expected to occur in bone and in the calcium-rich eggshell. In one case, a positive relation was demonstrated between reactor releases of ^{90}Sr to the Columbia River and ^{90}Sr concentrations in reed canary grass (*Phalaris arundinacea*) and eggshells of the Canada goose (*Branta canadensis moffitti*; Rickard and Price 1990).

No human health problem is anticipated from consumption of ruffed grouse (*Bonasa umbellus*) that are contaminated with ^{226}Ra in Canada or of American coots (*Fulica americana*) contaminated with ^{137}Cs in Washington state. Tissues of ruffed grouse that were collected near discharged uranium tailings in Canada in 1987-88 did not contain grossly elevated levels of ^{226}Ra over controls; consumption of grouse by humans did not present a radiological health problem (Clulow et al. 1992). Based on ^{137}Cs alone, humans who consume a single contaminated American coot captured at Hanford, Washington, would receive about 1.1% of the annual radiation protection dose of 1.70 mSv by individuals and populations in uncontrolled areas (Cadwell et al. 1979).

Mammals

Diets in Denmark contained elevated loadings of ^{137}Cs in 1964 because of the intensive atmospheric nuclear-test series by the United States and the former Soviet Union in 1961 and 1962. Total ^{137}Cs intake declined in the Danish population from 72 Bq/kg BW in 1964 to less than 2 in 1985 but rose to about 13 in 1986 from the effects of debris from Chernobyl on dietary ^{137}Cs during the first year after the accident (Aarkrog 1990). The estimated dose equivalent from ^{137}Cs to human consumers of fish from the Great Lakes is about 0.01 uSv/kg fresh weight (FW) muscle of fish from Lakes Erie and Ontario and 0.06-0.07 uSv/kg from fish in Lakes Superior and Huron (Joshi 1991). The guide for the protection of the general public from radiation is less than 5 mSv annually, and consumption of fish that contains a dose equivalent greater than 0.02 uSv/kg fish flesh is not recommended (Joshi 1991). Some Scandinavians now receive a dose equivalent of about 5 mSv/year from intake of radiocesium in the diet (Johanson 1990). In Finland, uptake of radionuclides by humans in Finnish Lapland and in other areas with an arctic climate is attributed to ecological factors and to a high amount of local fallout. For example, reindeer-herding Finnish Lapps contained about 50 times more ^{137}Cs and 10 times more ^{55}Fe than other Finns during 1961-67. For ^{137}Cs, this disparity is attributed mainly to the reliance by Finns on reindeer meat--which contains high levels of ^{137}Cs as a result of reindeer feeding on lichens--and secondarily, on freshwater fish and cow's milk (Miettinen 1969).

In the United States, the estimated annual whole body human radiation dose equivalent is 1.61 mSv, mostly from natural sources (0.85 mSv) and medical sources (0.70 mSv) but also from fallout (0.03 mSv), miscellaneous sources (0.02 mSv), occupational hazards (0.008 mSv), and nuclear power (0.0001 mSv; League of Women Voters [LWV] 1985). Radiation doses to people who live near the Hanford nuclear industrial and research site in the state of Washington are well below existing regulatory standards. Only trace amounts of radionuclides from Hanford have been detected in the offsite environment (Gray et al. 1989). In December 1984, radon levels as much as 130 times greater than considered safe under the current guideline for underground uranium miners were discovered in human residences in eastern Pennsylvania, New Jersey, and New York. About 25% of all residences in 10 states exceeded the action level for radon of 0.185 Bq/L air (Cross 1990; Oge and Dickson 1990). The significance of this observation to avian and terrestrial wildlife merits investigation.

As a result of nuclear weapons testing, mandibles of Columbian black-tailed deer (*Odocoileus hemionus columbianus*) from California increased from less than 9 Bq ^{90}Sr/kg ash weight (AW) to more than 204 Bq/kg AW between 1952 and 1960 (Table 14; Schultz and Longhurst 1963). Age and season affected strontium kinetics in male mule deer (*Odocoileus hemionus hemionus*) during the period of antler growth; these variables did not affect strontium kinetics in females (Schreckhise and Whicker 1976). The concentrations of ^{90}Sr in forage of mule deer were higher in summer than in winter and the differences were of sufficient magnitude to account for the ^{90}Sr variations in mule deer antlers (Farris et al. 1969); ^{137}Cs concentrations were similar in the forage and flesh of the white-tailed deer (*Odocoileus virginicanus*; Cummings et al. 1971). Levels of iodine-129 in thyroids of mule deer and pronghorns (*Antilocapra americana*) increased with proximity to nuclear-fuel reprocessing plants in Colorado, Idaho, New Mexico, and Wyoming during 1972-76, although levels were considered of no consequence to the health of the animals (Markham et al. 1983).

Radium-226, a bone-seeking α-emitter with a half-life of 1,600 years, may cause tissue damage and possibly subsequent osteosarcoma. Elevated ^{226}Ra concentrations have been reported in tissues of the beaver (*Castor canadensis*) from the Serpent River watershed, Canada, the recipient of uranium tailings during 1984-87 (Table 14). Measurable levels of ^{226}Ra were also found in feces of snowshoe hares (*Lepus americanus*) from this area and in black cutworms (*Agrotis ipsilon*) eaten by herring gulls (*Larus argentatus*) on the tailings (Clulow et al. 1991). Maximum levels in tissues of beavers from this watershed were less than 5 Bq ^{232}Th/kg dry weight (DW) in all tissues, 15 Bq ^{228}Th/kg DW bone, less than 5 Bq ^{228}Th/kg DW muscle and liver, 70-160 Bq ^{210}Po/kg DW bone, 11-75 Bq ^{210}Po/kg DW muscle, and 35-65 Bq ^{210}Po/kg DW liver. Consumption of these beavers would not be hazardous to human health. In the worst case, humans who consume substantial (71 kg) amounts of flesh of beavers from the Serpent River drainage system would receive less than 10% of the annual limits set by Canadian regulatory authorities (Clulow et al. 1991).

Cesium-137 levels in gray seals (*Halichoerus grypus*) in 1987 seem to reflect ^{137}Cs levels in their fish diet, but there is no biomagnification of ^{137}Cs and other radionuclides. An estimated 29% of the ^{137}Cs in the diets of gray seals is from the Chernobyl accident and 71% from the nuclear facility at Sellafield, United Kingdom. The dose to gray seals from their diet is about 36 mSv annually and higher than the permissible dose limit of 5 mSv/year allowed the general public but below the current limit for radiation workers of 50 mSv/year (S.S. Anderson et al. 1990).

The weekly dose rates from internal radionuclides were markedly different in muskrats (*Ondatra zibethicus*) and cotton rats (*Sigmodon hispidus*) collected at Oak Ridge, Tennessee, in August 1960 (20-1,112 mSv in muskrats vs. 3 mSv in cotton rats); the difference is probably due to differences in diets and habitats (Kaye and Dunaway 1963). Foxes and wildcats contain 2 to 16 times more ^{137}Cs than their prey organisms such as rats and rabbits (Jenkins et al. 1969), suggesting food-chain magnification. The biological half-life of ^{137}Cs is about 30 days in foxes, dogs, and pigs but about 60 days in humans (Jenkins et al. 1969). Black-tailed jackrabbits (*Lepus californicus*) in 1958, 1 year after contamination at the Nevada test site, averaged 1,908 Bq ^{90}Sr/kg AW bone within a 160-km radius from ground zero; in 1961, the average in the same population was only 984 Bq ^{90}Sr/kg AW bone, and the few higher values were restricted to older animals (Neel and Larson 1963). The authors concluded that ^{90}Sr from fallout in jackrabbits is at its maximum at an early time after contamination and that biological availability is later reduced by natural (unspecified) mechanisms. Jackrabbits at the Nevada test site also contained certain neutron activation products, including isotopes of Co, Mn, and W (Romney et al. 1971).

Radionuclide concentrations in sheep and cattle that grazed near a nuclear-fuel reprocessing facility amounted to a small fraction of the recommended limits. Americium-241, ^{137}Cs, and $^{239+240}$Pu in bone, liver, lung, and muscle of beef cattle from the vicinity of a nuclear-fuel reprocessing facility in England were quite low between September and December 1986 and practically indistinguishable from control samples. Maximum concentrations, in Bq/kg FW, were 0.0015 $^{239+240}$Pu in lung, 0.019 ^{241}Am in liver, and 3.1 ^{137}Cs in muscle (Curtis et al. 1991). Levels of ^{129}I were elevated in thyroids of cows near Mol, Belgium, in 1978 in the vicinity of a nuclear reprocessing plant closed in 1974 (Table 14; Handl et al. 1990).

Case Histories

Military weapons tests on the Pacific Proving Grounds in the 1940's and 1950's greatly elevated local concentrations of radionuclides, and an accident at the Chernobyl nuclear power plant in the former Soviet Union in 1986 dispersed comparatively low concentrations of radionuclides over a wide geographical area. Both cases are briefly reviewed.

Pacific Proving Grounds

The first artificial, large-scale introduction of radionuclides into a marine environment was at Bikini Atoll in 1946. In succeeding years through 1958, Bikini and Eniwetok became the Pacific Proving Grounds where 59 nuclear and thermonuclear devices were detonated between 1946 and 1958 (Welander 1969; Templeton et al. 1971; Bair et al. 1979; Table 15). Gross radiation injury to marine organisms has not been documented, possibly because seriously injured individuals do not survive and the more subtle injuries are difficult to detect. On land, the roof rat (*Rattus rattus*) survived heavy initial radiation by remaining in deep burrows. Terrestrial vegetation was heavily damaged by heat and blast but generated regrowth in 6 months. The land-dwelling hermit crab (*Coenobita* sp.) and coconut crab (*Birgus latro*) were subjected to higher levels of chronic radiation from internally deposited radionuclides than any other studied Atoll organism; levels remained constant in *Coenobita* at 166,000 Bq of ^{90}Sr/kg skeleton and 16,835 Bq ^{137}Cs/kg muscle for 2 years; *Birgus* contained 25,900 Bq ^{90}Sr/kg skeleton and 3,700 Bq ^{137}Cs/kg muscle for 10 years (Templeton et al. 1971). A survey in August 1964 at Eniwetok and Bikini Atolls (Welander 1969; Table 15) showed that general levels of radioactivity were comparatively elevated and highest in soils and increasingly lower in aquatic invertebrates, groundwater, shorebirds, plants, rats, zooplankton, algae, fishes, sediments, seawater, and seabirds. Cobalt-60 was in all samples of animals, plants, water, sediments, and soils and was the major radionuclide in the marine environment; on land, cesium-137 and ^{90}Sr predominated. All samples contained traces of ^{54}Mn; ^{106}Ru and ^{125}Sb were in groundwater, and soil and trace concentrations were in animals and plants. Trace amounts of ^{207}Bi and ^{144}Ce were usually detected in algae, soils, and land plants. Iron-55 was comparatively high in vertebrates, and ^{239}Pu was found in the soil and in the skin of rats and birds (Welander 1969).

Table 15. Radionuclide concentrations in selected samples from the Pacific Proving Ground. Concentrations are in becquerels/kg fresh weight (FW) or dry weight (DW).

Table 15 Location, sample, radionuclide, and other variables	Concentration, in Bq/kg or Bq/L	Reference[a]
Bikini Atoll		
Samples with highest concentrations, August 1964		
^{207}Bi, sediments	Max. 6,660 DW	1
^{144}Ce, marine algae	Max. 1,739 DW	1
^{137}Cs, land invertebrates	Max. 14,060 DW	1
^{57}Co, sediments	Max. 3,400 DW	1
^{60}Co, marine invertebrates	Max. 35,150 DW	1
^{54}Mn, sediments	Max. 962 DW	1
^{106}Ru, sediments	Max. 10,360 DW	1
^{125}Sb, groundwater	Max. 12,950 DW	1
Seawater, 1972, ^{55}Fe	Max. 0.025 FW	2
Sediments		
1958 vs. 1972, ^{55}Fe	Max. 777,000 DW vs. 11,100 DW	2
August 1964, ground zero		
^{207}Bi	6,660 DW	1
^{57}Co	3,404 DW	1

Table 15 Location, sample, radionuclide, and other variables	Concentration, in Bq/kg or Bq/L	Reference[a]
^{60}Co	9,620 DW	1
^{54}Mn	962 DW	1
^{106}Ru	10,360 DW	1
^{125}Sb	3,663 DW	1
Eniwetok Atoll, August 1964		
Whole marine algae vs. whole marine fishes		
^{207}Bi	181 DW vs. 74 DW	1
^{144}Ce	814 DW vs. nondetectable (ND)	1
^{137}Cs	52 DW vs. 21 DW	1
^{60}Co	355 DW vs. 888 DW	1
^{54}Mn	48 DW vs. 70 DW	1
^{106}Ru	96 DW vs. ND	1
^{125}Sb	34 DW vs. ND	1
Terrestrial invertebrates vs. terrestrial vegetation		
^{207}Bi	6 DW vs. 10 DW	1
^{144}Ce	5 DW vs. 888 DW	1
^{137}Cs	No data vs. 12,580 DW	1
^{60}Co	888 DW vs. 141 DW	1
^{54}Mn	281 DW vs. 296 DW	1
^{106}Ru	15 DW vs. 19 DW	1
^{125}Sb	ND vs. 8 DW	1
Seabirds (whole) vs. shorebirds (whole)		
^{207}Bi	ND vs. ND	1
^{57}Co	12 DW vs. ND	1
^{60}Co	340 DW vs. 4,810 DW	1
^{137}Cs	ND vs. 4,440 DW	1
^{64}Mn	81 DW vs. ND	1
^{106}Ru, ^{125}Sb	ND vs. ND	1
Roof rat, *Rattus rattus*; whole		
^{207}Bi	5 DW	1
^{144}Ce	362 DW	1
^{60}Co	888 DW	
^{137}Cs	19,980 DW	1
^{54}Mn	1 DW	1
^{106}Ru, ^{125}Sb	ND	1
Samples with highest concentrations		
^{207}Bi, marine plankton	Max. 333 DW	1
^{144}Ce, soils	Max. 2,109 DW	1
^{137}Cs, rats	Max. 19,980 DW	1

Table 15 Location, sample, radionuclide, and other variables	Concentration, in Bq/kg or Bq/L	Reference[a]
^{57}Co, sediments	Max. 740 DW	1
^{61}Co, marine invertebrates	Max. 6,290 DW	1
^{14}Mn, land plants	Max. 296 DW	1
^{106}Ru, soils	Max. 4,440 DW	1
^{125}Sb, soils	Max. 703 DW	1
Soils vs. sediments		
^{207}Bi	20 DW vs. 218 DW	1
^{144}Ce	2,109 DW vs. no data	1
^{137}Cs	2,072 DW vs. 814 DW	1
^{57}Co	No data vs. 740 DW	1
^{60}Co	2,849 DW vs. 1,073 DW	1
^{54}Mn	44 DW vs. 148 DW	1
^{106}Ru	4,440 DW vs. 3,700 DW	1
^{125}Sb	703 DW vs. 407 DW	1
Eniwetok Atoll,		
Runit Island		
(8 nuclear detonations between 1948 and 1958)		
Roof rat, whole		
Immediate vicinity of detonations;		
1967 vs. 1973, ^{137}Cs		
Bone	21,978 DW vs. 81,363 DW	3
Intestine	137,344 DW vs. no data	3
Kidney	189,958 DW vs. 126,799 DW	3
Liver	83,657 DW vs. 83,583 DW	3
Muscle	137,122 DW vs. 156,880 DW	3
Skin	13,209 DW vs. 77,256 DW	3
200 m vs. 2,460 m; 1967, ^{60}Co		
Bone	185 DW vs. ND	3
Intestine	8,251 DW vs. no data	3
Kidney	110,223 DW vs. 333 DW	3
Muscle	499 DW vs. 266 DW	3
Skin	259 DW vs. ND	3
Soils		
^{137}Cs, 1967		
Ground zero vs. 200 m	1,258 DW vs. 399 DW	3
1,030 m vs. 2,460 m	88 DW vs. 18 DW	3
^{137}Cs, 1971		
Ground zero vs. 200 m	4,736 DW vs. 403 DW	3
1,030 m	44 DW	3
^{60}Co, 1967		
Ground zero vs. 200 m	1,221 DW vs 66 DW	3
1,030 m	25 DW	3
^{60}Co, 1971		
Ground zero vs. 200 m	1,110 DW vs. 133 DW	3
1,030 m vs. 2460 m	40 DW vs. 4 DW	3
1973, 2,460 m		

Table 15 Location, sample, radionuclide, and other variables	Concentration, in Bq/kg or Bq/L	Reference[a]
^{137}Cs	11 DW	3
^{60}Co	52 DW	3
Terrestrial vegetation		
Ground zero, 1967 vs. 1971		
^{137}Cs	16,199-93,380 DW vs. 34,780-94,239 DW	3
^{60}Co	Max. 1,221 DW vs. max. 2,775 DW	3
1,030 m, 1967 vs. 1971		
^{137}Cs	296-2,035 DW vs. 333-1,961 DW	3
^{60}Co	Max. 14 DW vs. max. 48 DW	3

[a]1, Welander 1969; 2, Schell 1976; 3, Bastian and Jackson 1976.

Chernobyl

General

Several accidents in nuclear facilities have been extensively analyzed and reported. The three most widely publicized accidents were at Windscale (now known as Sellafield), United Kingdom, in 1957; Three Mile Island, Pennsylvania, in 1979; and Chernobyl, Ukraine, in 1986 (UNSCEAR 1988; Severa and Bar 1991). The accident at Windscale released about 750 trillion (T)Bq of ^{131}I, 22 TBq of ^{137}Cs, 3 TBq of ^{89}Sr, 0.33 TBq of ^{90}Sr, and twice the amount of noble gases that were released at Chernobyl but 2,000 times less ^{131}I and ^{137}Cs. The Three Mile Island accident released about 2% as much noble gases and 50,000 times less ^{131}I than the Chernobyl accident. The most abundant released radionuclides at Three Mile Island were ^{133}Xe, ^{135}Xe, and ^{131}I, but the collective dose equivalent to the population during the first post-accident days was less than 1% of the dose accumulated from natural background radiation in 1 year.

The most serious accident of a nuclear reactor occurred on 26 April 1986 at one of the four units at Chernobyl when at least 3,000,000 TBq were released from the fuel during the accident (Table 16). The accident happened while a test was conducted during a normal scheduled shutdown and is attributed mainly to human error: "...the operators deliberately and in violation of rules, withdrew most control rods from the core and switched off some important safety systems..." (UNSCEAR 1988). The first power peak reached 100 times the nominal power within 4 s. Energy released in the fuel by the power excursion suddenly ruptured part of the fuel into minute pieces. Small, hot fuel particles caused a steam explosion. After 2 or 3 s, another explosion occurred, and hot pieces of the reactor were ejected. The damage to the reactor allowed air to enter, causing combustion of the graphite. About 25% of the released radioactive materials escaped during the first day of the accident; the rest, during the next 9 days (UNSCEAR 1988). The initial explosions and heat from the fire carried some of the radioactive materials to an altitude of 1,500 m where they were transported by prevailing winds (Fig. 6) and caused widespread radioactive contamination of Europe and the former Soviet Union, initially with ^{131}I, ^{134}Cs, and ^{137}Cs (Smith and Clark 1986; Anspaugh et al. 1988; Clark and Smith 1988; UNSCEAR 1988; Aarkrog 1990; Johanson 1990; Brittain et al. 1991; Palo et al. 1991). Long-range atmospheric transport spread the radioactive materials through the northern hemisphere where it was first detected in Japan on 2 May, in China on 4 May, in India on 5 May, and in Canada and the United States on 5-6 May 1986 (UNSCEAR 1988). Airborne activity was also detected in Turkey, Kuwait, Monaco, and Israel in early May. No airborne activity from Chernobyl has been reported south of the equator (UNSCEAR 1988). Among the reactors now operating in the former Soviet Union, 13 are identical to the one in Chernobyl, Ukraine, including units in Chernobyl, Leningrad, Kursk, and Smolensk (Mufson 1992).

Table 16. Selected fission products in the Chernobyl reactor core and their estimated escape into the environment (Severa and Bar 1991).

	Trillions of becquerels (TBq)	
Radionuclide	In core	Escaped[a]
^{85}Kr	33,000	33,000
^{133}Xe	1,700,000	1,700,000
^{131}I	1,300,000	260,000
^{132}Te	320,000	48,000
^{134}Cs	190,000	19,000
^{137}Cs	290,000	37,700
^{99}Mo	4,800,000	110,400
^{95}Zr	4,400,000	140,800
^{103}Ru	4,100,000	118,900
^{106}Ru	2,000,000	58,000
^{140}Ba	2,900,000	162,400
^{141}Ce	4,400,000	101,200
^{144}Ce	3,200,000	89,600
^{89}Sr	2,000,000	80,000
^{90}Sr	200,000	8,000
^{239}Np	140,000	4,200
^{238}Pu	1,000	30
^{239}Pu	850	25
^{240}Pu	1,200	36
^{241}Pu	170,000	5,100
^{242}Cm	26,000	780

[a] Aarkrog (1990) estimates escapement of 100,000 TBq of ^{137}Cs; 50,000 TBq of ^{134}Cs; and 35,000 TBq of ^{106}Ru. Aarkrog (1990) also includes the following radionuclides in the Chernobyl escapement: 1,500 TBq of ^{110}Ag, 3,000 TBq of ^{125}Sb, 6 TBq of ^{241}Am, and 6 TBq of $^{243+244}Cm$.

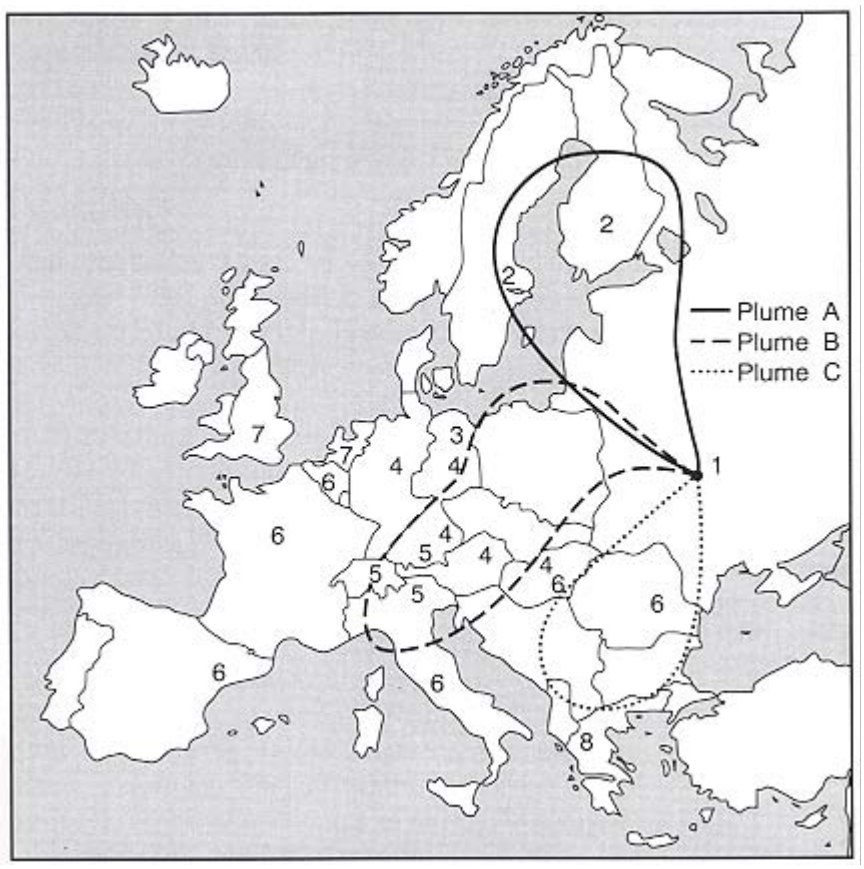

Fig. 6. Chernobyl air plume behavior and reported initial arrival times of detectable radioactivity. Plume A originated from Chernobyl on 26 April 1986, Plume B on 27-28 April, and Plume C on 29-30 April. The *numbers* indicate initial arrival times: 1, 26 April; 2, 27 April; 3, 28 April; 4, 29 April; 5, 30 April; 6, 1 May; 7, 2 May; and 8,3 May (UNSCEAR 1988; country boundaries in 1986).

Effective dose equivalents from the Chernobyl accident in various regions of the world were highest in southeastern Europe (1.2 mSv), northern Europe (0.97 mSv), and Central Europe (0.93 mSv; Table 17). In the first year after the accident, whole-body effective dose equivalents were highest in Bulgaria, Austria, Greece, and Romania (0.5-0.8 mSv); Finland, Yugoslavia, Czechoslovakia, Italy (0.3-0.5 mSv); Switzerland, Poland, USSR, Hungary, Norway, Germany, and Turkey (0.2-0.3 mSv); and elsewhere (<0.2 mSv; UNSCEAR 1988). Thyroid dose equivalents were significantly higher than whole-body effective dose equivalents because of significant amounts of ^{131}I in the released materials. Thyroid dose equivalents were as high as 25 mSv to infants in Bulgaria, 20 mSv in Greece, and 20 mSv in Romania; the adult thyroid dose equivalents were usually 80% lower than the infant dose equivalents (UNSCEAR 1988).

Table 17. Regional total effective human dose-equivalent commitment from the Chernobyl accident (UNSCEAR 1988; Aarkrog 1990).

Region	Effective dose equivalent (mSv)
Southeastern Europe	1.2
Northern Europe	0.97
Central Europe	0.93
Former Soviet Union	0.81
Southwest Asia, West Europe	>0.1-<0.2
North Africa, Greenland, East Africa, Central Africa, South Asia, West Africa	>0.01-<0.l
East Asia, Southwest Europe, Southeast Asia, North America, Caribbean, South America, Central America	<0.01

Local Effects

At Chernobyl, at least 115 humans received acute bone-marrow doses of greater than 1 Gy, as judged by lymphocyte aberrations (UNSCEAR 1988). The death toll within 3 months of the accident was at least 30 individuals, usually from groups that received more than 4 Gy, including the reactor's operating staff and the fire-fighting crew. Local residents were evacuated from a 30-km exclusion zone around the reactor because of increasing radiation levels; more than 115,000 people, including 27,000 children, were evacuated from the Kiev region, Byelorussia, and the Ukraine. Tens of thousands of cattle were also removed from the contaminated area, and consumption of locally produced milk and other foods was banned. Agricultural activities were halted and a large-scale decontamination effort was made (UNSCEAR 1988). The radiological effect of the accident to individual risk was insignificant outside a limited local region, either because contamination levels were generally low or because remedial actions to ban the consumption of highly contaminated foodstuffs prevented high exposures (UNSCEAR 1988).

Biological effects of the Chernobyl accident on local natural resources were documented by Sokolov et al. (1990). They concluded that the most sensitive affected ecosystems at Chernobyl were the soil fauna and pine forest communities and that the bulk of the terrestrial vertebrate community was not adversely affected by released ionizing radiation. Pine forests seemed to be the most sensitive ecosystem. One stand of 400 ha of *Pinus silvestris* died and probably received a dose of 80-100 Gy; other stands experienced heavy mortality of 10-12 year old trees and as much as 95% necrotization of young shoots; these pines received an estimated dose of 8-10 Gy. Abnormal top shoots developed in some *Pinus*, and these probably received 3-4 Gy. In contrast, leafed trees such as birch (*Betula* sp.), oak (*Quercus sp.*), and aspen (*Populus* sp.) in the Chernobyl Atomic Power Station zone survived undamaged, probably because they are about 10 times more radioresistant than pines. There was no increase in the mutation rate of the spiderwort *Arabidopsis thaliana*, a radiosensitive plant, suggesting that the dose rate was less than 0.05 Gy/h in the Chernobyl locale. Populations of soil mites were reduced in the Chernobyl area, but no population showed a catastrophic drop in numbers. By 1987, soil microfauna--even in the most heavily contaminated plots--was comparable to controls. Flies (*Drosophila* spp.) from various distances from the accident site and bred in the laboratory had higher incidences of dominant lethal mutations (14.7%, estimated dose of 0.8 m Gy/h) at sites nearest the accident than controls (4.3%). Fish populations seemed unaffected in July-August 1987, and no grossly-deformed individuals were found; however, $^{134+137}$Cs levels were elevated in young fishes. The most heavily contaminated teleost in May 1987 was the carp (*Carassius carassius*). But carp showed no evidence of mutagenesis, as judged by chromosomal aberrations in cells from the corneal epithelium of some carp as far as 60 km from Chernobyl (Sokolov et al. 1990).

Several rodent species compose the most widely distributed and numerous mammals in the Chernobyl vicinity. It was estimated that about 90% of rodents died in an area that received 60 Gy and 50% in areas that received 6-60 Gy. Rodent populations seemed normal in spring 1987, and this was attributed to migration from adjacent nonpolluted areas. The most sensitive small mammal was the bank vole (*Clethrionomys glareolus*),

which experienced embryonic mortality of 34%. The house mouse (*Mus musculus*) was one of the more radioresistant species. *Mus* from plots receiving 0.6-1 mG/h did not show signs of radiation sickness, were fertile with normal sperm, bred, and produced normal young. Some chromosomal aberrations were evident, namely, an increased frequency of reciprocal translocations (Sokolov et al. 1990). During the early period after the accident, there was no evidence of increasing mortality, decline in fecundity, or migration of vertebrates as a result of the direct action of ionizing radiation. The numbers and distributions of wildlife species were somewhat affected by the death of the pine stand, the evacuation of people, the termination of cultivation of soils (the crop of 1986 remained standing), and the evacuation of domestic livestock. No changes in survival or species composition of game animals and birds were recorded. In fact, because humans had evacuated and hunting pressure was negligible, many game species, including foxes, hares, deer, moose, wolves, and waterfowl moved into the zone in fall 1986-winter 1987 from the adjacent areas in a 50-60 km radius (Sokolov et al. 1990).

Nonlocal Effects

The partial meltdown of the 1,000-megawatt reactor at Chernobyl on 26 April 1986 released large amounts of radionuclides into the environment--especially ^{131}I, ^{137}Cs, and ^{134}Cs--and widely dispersed and deposited radioactive material in Europe and throughout the northern hemisphere (UNSCEAR 1988; Palo et al. 1991; Table 18). Transuranics and to some extent ^{90}Sr were deposited closer to the accident site than more volatile radionuclides such as radiocesium; accordingly, radiological problems changed quantitatively and qualitatively with increasing distance from the accident site (Aarkrog 1990).

Table 18. Radionuclide concentrations in biotic and abiotic materials from various geographic locales before or after the Chernobyl nuclear accident on 26 April 1986. All concentrations are in Bq/kg fresh weight (FW) or dry weight (DW), unless noted otherwise.

Table 18 Locale, radionuclide, sample, and other variables	Concentration	Reference[a]
Alaska and Yukon Territories		
Barren-ground caribou (*Rangifer tarandus granti*); porcupine herd; March-November 1987; ^{137}Cs		
Feces	Max. 802 DW	1
Muscle	133 (26-232) FW	1
Rumen contents	Max. 538 DW	1
Albania		
^{137}Cs; 2 May-19 May 1986		
Air	Max. 1.8 Bq/m^3	2
Milk vs. wheat flour	Max. 380 FW vs. max. 236 FW	2
^{131}I; cow's milk; 2 May-19 May 1986	Max. 3,500 FW	2
Canada		
Caribou, *Rangifer tarandus*; northern Quebec; 1986 (post-Chernobyl)-1987; muscle; ^{137}Cs	166-1,129 FW	3
Czechoslovakia [in 1986]		
$^{134+137}Cs$; 1986 (post-Chernobyl)		
Barley, *Hordeum vulgare*	7 DW	4
Cow, *Bos* sp., milk		
May	42 FW	4

Table 18 Locale, radionuclide, sample, and other variables	Concentration	Reference[a]
July	10 FW	4
December	7 DW	4
Wheat, *Triticum* sp.	16 DW	4
$^{134+137}Cs$; domestic pig, *Sus* sp.; muscle; July 1986 vs. July 1987	15-22 FW vs. 22 FW	4
Danube River, Hungary-Yugoslavia [in 1986]		
Water; 1986; post-Chernobyl		
^{134}Cs	0.015 FW	5
^{137}Cs	0.096 FW	5
^{103}Ru	0.070 FW	5
Fish, various species; 1986 (post-Chernobyl) vs. 1987		
^{134}Cs	8 FW vs. 4 FW	5
^{137}Cs	13 FW vs. 12 FW	5
^{103}Ru	1 FW vs. <1 FW	5
^{106}Ru	4 FW vs. 3 FW	5
Sediments; 1986 (post-Chernobyl) vs. 1988		
^{134}Cs	500 DW vs. 80 DW	5
^{137}Cs	750 DW vs. 200 DW	5
Algae; 1986 (post-Chernobyl) vs. 1988		
^{134}Cs	275 FW vs. 25 FW	5
^{137}Cs	625 FW vs. 100 FW	5
Finland		
Finish Lapland; ^{137}Cs; 1979-84 vs. 1986 (post-Chernobyl)		
Arboreal lichens	120 DW vs. 590 DW	7
Ground lichens	230 DW vs. 900 DW	7
Birch, *Betula* sp.	68 DW vs. 51 DW	7
Horsetails, *Equisetum* sp.	203 DW vs. 280 DW	7
Bilberry, *Vaccinium* sp.	120 DW vs. 590 DW	7
Lichens; ^{137}Cs		
From reindeer herding areas; 1986 (post-Chernobyl) vs. 1987	900 DW vs. 800 DW	8
Isolated areas; 1986 (post-Chernobyl)-1987	3,000-10,000 DW	8
Lake Paijanne (estimated ^{137}Cs Chernobyl loading of 20,000 Bq/m^2); ^{137}Cs; whole fish; three species (northern pike, *Esox lucius*; yellow perch, *Perca flavescens*;		

Table 18 Locale, radionuclide, sample, and other variables	Concentration	Reference[a]
roach, *Rutilus rutilus*)		
1986; pre-Chernobyl vs. post-Chernobyl	580 FW vs. 1,250 FW	6
1987	1,000-2,000 FW	6
1988	160-2,000 FW	6
Reindeer, *Rangifer tarandus*; muscle; ^{137}Cs		
1964-65 (following nuclear tests) vs. 1985-86 (pre-Chernobyl)	Max. 2,500-2,600 FW vs. 300 FW	7,8
1986-87 vs. 1987-88	720 FW, max. 16,000 FW vs. 640 FW, max. 9,000 FW	8
France		
Cows, fed hay (harvested post-Chernobyl) diet containing 5,500 $^{134+137}Cs$/kg for mean daily intake of 15,900 Bq	A plateau was observed in milk after 15 days and in meat after 50-60 days; radiocesium transfer coefficients from diet were 1.1% for milk and 2.0-2.7% for meat	9
Calves fed $^{l34+137}Cs$-contaminated milk from birth to age 80 days	Transfer coefficient from milk to meat was 16%	9
Germany [in 1986]		
Soils; 24 June 1986		
^{134}Cs	Max. 602 Bq/m^2 DW	10
^{137}Cs	Max. 1,545 Bq/m^2 DW	10
^{103}Ru	Max. 808 Bq/m^2 DW	10
Pasture vegetation; May 1986		
^{134}Cs	20 FW	10
^{137}Cs	40 FW	10
^{131}I	75 FW	10
Cow; milk; May 1986		
^{134}Cs	140 FW	10
^{137}Cs	250 FW	10
^{131}I	250 FW	10
^{103}Ru	250 FW	10
Human, *Homo sapiens*		
Intake per person		
^{134}Cs; 1986 vs. 1987	354 Bq vs. 8 Bq	10
^{137}Cs; 1986 vs. 1987	728 Bq vs. 37 Bq	10
Whole body dose (Bonn and vicinity); 1986 vs. 1987	0.0147 mSv (0.008 from ^{137}Cs, 0.0067 from ^{134}Cs) vs. 0.00056 mSv (0.0004 from ^{137}Cs, 0.00016 from ^{134}Cs)	10
Thyroid, ^{129}I	Negligible	11

Table 18 Locale, radionuclide, sample, and other variables	Concentration	Reference[a]
Greece		
Alfalfa, *Medicago sativa*; June 1986		
^{134}Cs	2,303 DW	12
^{137}Cs	4,551 DW	12
^{103}Ru	358 DW	12
^{106}Ru	1,075 DW	12
Lichen, *Ramalina fraxinea* vs. moss, *Homalothecium sericium*; 1986 (post-Chernobyl); after decay of short-lived radionuclides		
^{134}Cs	426 FW vs. 1,121 FW	13
^{137}Cs	951 FW vs. 2,612 FW	13
^{40}K	222 FW vs. 278 FW	13
^{103}Ru	63 FW vs. 115 FW	13
^{106}Ru	436 FW vs. 1,365 FW	13
Rye grass, *Lolium perenne*; June 1986		
^{134}Cs	3,518 DW	12
^{137}Cs	7,090 DW	12
^{103}Ru	708 DW	12
^{106}Ru	1,747 DW	12
Plants, various; measured about 4 months post-Chernobyl; ^{137}Cs; values represent about 9% of initial Chernobyl radioactivity		
Aromatic plants; 11 species	22-11,344 FW; 26-22,000 DW	13
Cereals; four species	11-2,257 FW; 11-2,775 DW	13
Fruit bearing trees; seven species	85-1,572 FW; 122-2,116 DW	13
Fungi; four species	103-5,553 FW; 214-11,418 DW	13
Marine algae; four species	85-139 FW; 529-917 DW	13
Mosses and lichens; six species	1,184-9,413 FW; 1,110-18,847 DW	13
Vegetables; 18 species	18-244 FW; 18-299 DW	13
Northern Greece; May 1986; ^{131}I		
Grasses	Max. 1,500 FW	14
Milk; cow vs. domestic sheep, *Ovis aries*	Max. 300 FW vs. max. 800 FW	14
Domestic sheep; thyroid; ^{131}I; maximum values; 1986		
27 June vs. 2 July	4,000 FW vs. 15,600 FW	15

Table 18 Locale, radionuclide, sample, and other variables	Concentration	Reference[a]
3 July vs. 5 July	618,000 FW vs. 9,000 FW	15
29 July vs. 20 August	8,500 FW vs. 600 FW	15
Italy		
Honey bee, *Apis* spp.; honey; 10 May 1986		
^{134}Cs	Max. 171 FW	16
^{137}Cs	Max. 363 FW	16
^{131}I	Max. 1,051 FW	16
^{103}Ru	Max. 575 FW	16
Cow		
Fed diets contaminated with Chernobyl ^{137}Cs for 8 months before slaughter		
Female vs. fetus		
Amniotic fluid	Max. 82 FW vs. —[b]	17
Blood	Max. 13 FW vs. max. 44 FW	17
Muscle	Max. 179 FW vs. max. 126 FW	17
Kidney	Max. 232 FW vs. max. 139 FW	17
Liver	Max. 163 FW vs. max. 115 FW	17
Placenta	Max. 93 FW vs. —	17
Rodent, *Mus musculus domesticus*; carcass less internal organs; ^{137}Cs		
October-November 1981 vs. May 1986	5 DW vs. 43 DW	18
October-November 1986 vs. May 1987	20 DW vs. 18 DW	18
Northwest Saluggia, May 1986		
^{137}Cs, pasture grass vs. cow's milk	8,000 DW vs. 180 FW	19
^{131}I, pasture grass vs. cow's milk	12,000 DW vs. 870 FW	19
Rabbit, *Oryctolagus* sp.; fed Chernobyl-contaminated alfalfa meal diet containing, in Bq/kg FW, 856 ^{137}Cs, 369 ^{134}Cs, and 540 ^{40}K; or normal diet (112 ^{137}Cs, 41 ^{134}Cs, 503 ^{40}K) for various intervals		
Control diet		
Whole animal	16 ^{137}Cs FW, 7 ^{134}Cs FW, 87 ^{40}K FW	20
Muscle	22 ^{137}Cs FW, 8 ^{134}Cs	20

Table 18 Locale, radionuclide, sample, and other variables	Concentration	Reference[a]
	FW, 117 ^{40}K FW	
21 days on contaminated diet followed by 21 days on control diet		
Whole animal	20 ^{137}Cs FW, 9 ^{134}Cs FW, 79 ^{40}K FW	20
Muscle	31 ^{137}Cs FW,128 ^{134}Cs FW, 117 ^{40}K FW	20
42 days on contaminated diet		
Whole animal	81 ^{137}Cs FW, 32 ^{134}Cs FW, 85 ^{40}K FW	20
Muscle	112 ^{137}Cs FW, 44 ^{134}Cs FW, 124 ^{40}K FW	20
Japan		
^{137}Cs		
Milk; cow; May 1986	Max. 0.6 FW	21
Soil; estimated deposition from Chernobyl	180 Bq/m^2 DW	21
$^{134+137}$Cs; humans, children; estimated internal dose through milk consumption		
1986	0.0006 mSv	21
1987	0.0003 mSv	21
1988	0.0001 mSv	21
^{131}I, grass vs. cow's milk		
10-11 May 1986	65 FW vs. 4.3 FW	22
30 May 1986	14 FW vs. ND[c]	22
Monaco		
Air, Bq/m^3, 26 April 1986; Monaco vs. Chernobyl (Former Soviet Union)		
^{134}Cs	8.2 vs. 53	50
^{137}Cs	1.6 vs. 120	50
^{103}Ru	3.5 vs. 280	50
^{131}I	4.6 vs. 750	50
^{106}Ru	3.0 vs. 110	50
^{140}Ba	9.8 vs. 420	50
^{99}Mo	3.8 vs. 490	50
^{141}Ce	3.7 vs. 190	50
^{144}Ce	2.5 vs. 110	50
^{95}Zr	1.2 vs. 590	50
Marine copepods, 3		

Table 18 Locale, radionuclide, sample, and other variables	Concentration	Reference[a]
species; 6 May 1986; whole organism vs. fecal pellets		
^{103}Ru	280 DW vs. 16,000 DW	49
^{106}Ru	70 DW vs. 5,800 DW	49
^{134}Cs	22 DW vs. 3,400 DW	49
^{137}Cs	34 DW vs. 6,300 DW	49
^{141}Ce	20 DW vs. 900 DW	49
^{144}Ce	100 DW vs. 2,500 DW	49
Mussel, *Mytilus galloprovincialis*; soft parts; 6 May vs. 14 August 1986		
^{103}Ru	480 FW vs. 9.6 FW	51
^{106}Ru	121 FW vs. 11.2 FW	51
^{131}I	84 FW vs. <2 FW	51
^{134}Cs	6 FW vs. 0.1 FW	51
^{137}Cs	5.2 FW vs. 0.3 FW	51
Netherlands		
^{134}Cs; grass silage; 1986 (post-Chernobyl) vs. 1987	Max. 50 DW vs. 2 DW	23
^{137}Cs; grass silage; 1986 (post-Chernobyl) vs. 1987	Max. 172 DW vs. 9 DW	23
^{137}Cs-contaminated roughage fed to lactating cows		
10.3 Bq ^{137}Cs/kg FW; grass	1.0-1.6 FW milk	24
173-180 Bq ^{137}Cs/kg FW; grass silage	12-28 FW milk	24
260-271 Bq ^{137}Cs/kg DW; grass	5.4-6.2 FW milk	24
^{40}K; grass silage; 1986 vs. 1987	910 DW vs. 1,028 DW	23
Norway		
Alpine lake and vicinity; $^{134+137}Cs$		
Dwarf birch, *Betula nana*; leaves; August 1986	4,000 FW	25
Lichens; August 1986	60,000 FW	25
Willow, *Salix spp*.; leaves; September 1980 vs. August 1986	<50 PW vs. 600 FW	25
Lake sediment; upper 10 cm; July-August 1986	1,050 FW	25
Aquatic organisms; July-August 1986		
Cladoceran, *Bosmina longispina*, whole	5,300 FW	25

Table 18 Locale, radionuclide, sample, and other variables	Concentration	Reference[a]
Amphipod, *Gammarus lacustris*, whole	6,700 FW	25
Mayfly, *Siphlonurus lacustris*, whole	2,800 FW	25
Stonefly, 2 spp., whole	1,300-4,120 FW	25
Minnow, *Phoxinus phoxinus*, whole	8,800 FW	25
Brown trout, *Salmo trutta*		
Muscle		
1985 (pre-Chernobyl) vs. June 1986	<100 FW vs. 300 FW	25
August 1986 vs. June 1988	7,000 FW vs. 4,000 FW	25
Eggs vs. milt; July-August 1986	1,740-3,600 FW vs. 1,300 FW	25
Dovrefjell, May 1986 vs. August 1990; ^{137}Cs		
Earthworms (*Lumbricus rubellus*, *Allobophora caliginosa*), whole	121 FW vs. 74 FW	52
Eurasian woodcock, *Scolopax rusticola*, breast muscle	737 FW vs. 53 FW	52
Litter	14,400 DW vs. 2,900 DW	52
Mushroom, *Lactarius* spp.; post-Chernobyl; $^{134+137}Cs$	Max. 445,000 FW	26
Reindeer; muscle; $^{134+137}Cs$		
1986; post-Chernobyl	10,000-50,000 FW	27
January 1987 vs. September 1988	Max. 56,000 FW vs. max. 13,900 FW	28
Reindeer; two groups of adult females were fed lichen diets containing 45,000 Bq $^{134+137}Cs$/kg ration for 35 days; one group received daily oral administration of 250 mg ammonium-ferrohexacyanoferrate (Giese salt)	Both groups accumulated 400 Bq/kg FW daily in muscle. Retention time of Cs isotopes was 25 days without Giese salt and only 7-10 days when treated with Giese salt	29
Poland		
Freshwater fish; four species; muscle; January 1987; $^{134+137}Cs$	4.5-6.1 FW	30
Southern Baltic Sea, $^{134+137}Cs$; pre-Chernobyl (1982-February 1986) vs. post-Chernobyl (June 1986-July 1987)		
Water	(13.8-19.8) Bq/m^3 vs. (59-100) Bq/m^3	30

Table 18 Locale, radionuclide, sample, and other variables	Concentration	Reference[a]
Atlantic cod, *Gadus morhua*; muscle	(1.4-2.3) FW vs. (5.0-7.4) FW	30
Flounder, *Pleuronectes flesus*; muscle	(1.1-4.5) FW vs. (3.4-6.7) FW	30
Spain		
Song thrush, *Turdus philomelos*; edible tissues; November 1986 vs. November 1987		
^{134}Cs	Max. 90 DW for adults and young vs. max. 7 DW for adults and 5 DW for young	31
^{137}Cs	Max. 208 DW vs. max. 27 DW for adults and 22 for young	31
^{90}Sr	Max. 23 DW vs. max. 7 DW	31
Sweden		
Moose, *Alces alces*; central Sweden; muscle; ^{137}Cs		
September 1986; adults vs. calves	300 FW vs. 500 FW	32
1986; all age groups	20-3,000 FW	33
September 1987; adults vs. calves	201 FW vs. 401 FW	32
1987, all age groups	Max. 1,600 FW	34
September 1988, adults vs. calves	640 FW vs. 1,300 FW	32
1988, all age groups	Max. 2,500 FW	34
Moose dietary plants; 1986 (post-Chernobyl)-1988; ^{137}Cs		
Birches, *Betula* spp.; leaves	1,200 DW	34
Heather, *Calluna vulgaris*; whole	13,000-32,000 DW	34,35
Sedges, *Carex* spp.; whole	12,000 DW	35
Hair grass, *Deschampia flexuosa*; whole	1,900 DW	34
Fireweed, *Epilobium angustifolium*; whole	400 DW	34
Grasses, various species; blades	2,500 DW	34
Buckbean, *Menyanthes trifoliata*; whole	3,800 DW	34
Pine, *Pinus sylvestris*; shoots	2,500 DW	34
Aspen, *Populus tremula*; leaves	700 DW	34
Willows, *Salix* spp.; leaves	300 DW	34
Mountain ash, *Sorbus aucuparia*; leaves	1,300 DW	34
Bilberry, *Vaccinium myrtillus*; leaves		
July 1986	2,000 FW; 4,000 DW	32,34

Table 18 Locale, radionuclide, sample, and other variables	Concentration	Reference[a]
July 1987 vs. July 1988	1,138 FW vs. 600 FW	32
Bog whortleberry, *Vaccinium uliginosum*; foliage	5,900 DW	34
Cowberry, *Vaccinium vitis-idaea*; foliage	7,500 DW	34
Cow's milk; ^{137}Cs; July 1986 vs. 1987	Usually <250 FW, max. 375 FW vs. usually <70 FW, max. 120 FW	36
Lichen, *Bryoria fuscescens*; ^{137}Cs; 4 June 1986	34,000-120,000 DW	35
Roe deer, *Capreolus* sp.; muscle; ^{137}Cs; 1986 (post-Chernobyl)	20-12,000 FW	33
Lichens, *Cladina* spp.; ^{137}Cs; 1986 (post-Chernobyl)	Max. 40,000 DW	35
Bank vole, *Clethrionomys glareolus*; collected from soil containing various concentrations of $^{134+137}Cs$; voles analyzed less skull and digestive organs		
1,800 Bq/m^2 soil (control)	Voles had 9 Bq ^{134}Cs/kg FW and 39 of ^{137}Cs; mutation frequency of 1.3; total irradiation of 0.0042 mGy daily	37
22,000 Bq/m^2 soil	In Bq/kg FW, voles had 279 ^{134}Cs and 1,031 ^{137}Cs; mutation frequency was 1.5; daily dose rate of 0.0088 mGy	37
90,000 Bq/m^2 soil	Voles had 1,356 Bq ^{134}Cs/kg FW and 5,119 of ^{137}Cs; mutation frequency 1.9; daily dose 0.0268 mGy	37
145,000 Bq/m^2 soil	Voles had 2,151 Bq ^{134}Cs/kg FW and 7,784 ^{137}Cs; mutation frequency 2.6; daily dose of 0.0394 mGy	37
Buckbean, *Menyanthes trifoliata*; foliage; ^{137}Cs; 1985 vs. 1987	1,800 DW vs. 3,880 DW	35
Reindeer dietary	Usually 40,000-60,000 DW;	38

Table 18 Locale, radionuclide, sample, and other variables	Concentration	Reference[a]
lichens; ^{137}Cs; April 1986	max. 120,000 DW	
Reindeer		
Moved in November 1986	^{137}Cs content in muscle	39
from a highly contaminated area (>20,000 Bq $^{137}Cs/m^2$) to a less-contaminated area (<3,000 Bq/m^2) Of natural pasture	declined from 12,000 FW in November to about 3,000 FW in April	
Muscle; ^{137}Cs; 1986 (post-Chernobyl)	100-40,000 FW	33
Rodents and insectivores;		
July-August 1986; ^{137}Cs		
Control site, soil	1,800 Bq/m^2	40
Bank vole; whole less skull, stomach, viscera	39 FW	40
Common shrew, *Sorex araneus*; whole less skull, stomach, viscera	48 FW	40
Site 2, soil	22,000 Bq/m^2	40
Bank vole vs. common shrew	676 FW vs. 751 FW	40
Site 3, soil	90,000 Bq/m^2	40
Bank vole vs. common shrew	5,119 FW vs. 3,233 FW	40
Site 4, soil	145,000 Bq/m^2	40
Bank vole vs. common shrew	7,993 FW vs. 6,289 FW	40
Syria		
^{137}Cs; Air; 7-10 May 1986	0.12 Bq/m^3	41
^{131}I; 7-10 May 1986; air vs. goat's milk	4 Bq/m^3 vs. 55 FW	41
United Kingdom		
Upland pastures		
Sphagnum moss, *Sphagnum* sp,; September 1986		
^{110m}Ag	202 DW	42
^{144}Ce	202 DW	42
^{134}Cs	8,226 DW	42
^{137}Cs	17,315 DW	42
^{106}Ru	1,893 DW	42
^{125}Sb	294 DW	42
Vegetation; $^{134+137}Cs$; June 1986 vs. January 1989	about 6,000 DW vs. 1,000 DW	43
Marine molluscs; 7 species; near nuclear plant; 1984 (pre-Chernobyl) vs. 1986 (post-Chernobyl)		

Table 18 Locale, radionuclide, sample, and other variables	Concentration	Reference[a]
^{110m}Ag	<77 FW vs. 13-77 FW	44
^{60}Co	<29 FW vs. 16-32 FW	44
^{134}Cs	<14 FW vs. 37-388 FW	44
^{137}Cs	<139 FW vs. 31-836 FW	44
^{40}K	<59 FW vs. 57-61 FW	44
^{238}Pu	<27 FW vs. 11-22 FW	44
$^{239+240}Pu$	<107 FW vs. 19-89 FW	44
^{106}Ru	<632 FW vs. 124-1,648 FW	44
^{125}Sb	ND vs. 29 FW	44
Eurasian oystereatcher, *Haematopus ostralegas*; near nuclear reactor; June 1986; egg contents vs. egg shells		
^{134}Cs	4 FW vs. — [b]	44
^{137}Cs	18 FW vs. 6 FW	44
^{238}Pu	0.2 FW vs. 1.1 FW	44
$^{239+240}Pu$	0.05 FW vs. 4.6 FW	44
Red grouse, *Lagopus lagopus*; muscle; November 1986-February 1987		
^{134}Cs; cock vs. hen	325 FW vs. 602 FW	45
^{137}Cs; cock vs. hen	962 FW vs. 1,684 FW	45
Common black-headed gull, *Larus ridibundus ridibundus*; near nuclear reactor; 1980 vs. June 1986		
Egg contents		
^{134}Cs	ND vs. 22 FW	44
^{137}Cs	10 FW vs. 43 FW	44
^{238}Pu	0.02 FW vs. 0.01 FW	44
$^{239+240}Pu$	0.05 FW vs. 0.04 FW	44
Egg shells		
^{134}Cs	— vs. 7 FW	44
^{137}Cs	— vs. 16 FW	44
^{238}Pu	<0. 17 FW vs. 0.4 FW	44
$^{239+240}Pu$	0.6 FW vs. 1.6 FW	44
Eurasian woodcock, *Scolopax rusticola*; muscle; November 1986-February 1987		
^{134}Cs	13 FW	45
^{137}Cs	42 FW	45
Black grouse, *Tetrao tetrix*; ^{137}Cs , November 1986-February 1987; diet vs. muscle	167 FW vs. 270 FW	45
Cow's milk; 5-8 May 1986		

Table 18 Locale, radionuclide, sample, and other variables	Concentration	Reference[a]
^{137}Cs	Max. 150 FW	46
^{131}I	Max. 127 FW	46
Roe deer, *Capreolus capreolus*; ^{137}Cs; muscle; November 1986-February 1987		
Calves	711 FW	45
Hinds	375-586 FW	45
Stags	1,564 FW	45
Red deer, *Cervus elephus*; muscle; November 1986-February 1987		
^{134}Cs; calf vs. hind	186 FW vs. 112 FW	45
^{137}Cs; calf vs. hind	535 FW vs. 311 FW	45
Brown hare, *Lepus capensis*; ^{137}Cs; female; November 1986-February 1987; diet vs. muscle	198 FW vs. 656 FW	45
Blue hare, *Lepus timidus*; ^{137}Cs; November 1986-February 1987		
Males; diet vs. muscle	808 FW vs. 1,677 FW	45
Females; diet vs. muscle	577 FW vs. 1,440 FW	45
Rabbit, *Oryctolagus* sp.; muscle; male; November 1986-February 1987		
^{134}Cs	6 FW	45
^{137}Cs	15 FW	45
Domestic sheep		
Muscle; ^{137}Cs; September 1986 vs. July 1987	1,500 FW vs. 1,170 FW	42
Liver; ^{110m}Ag; ewes vs. lambs		
September 1986	34 FW vs. 17 FW	47
July 1987 55	FW vs. <8 FW	47
Diet (rye grass and vegetation); ^{110m}Ag; 1986 vs. 1987	32 DW vs. 10-30 DW	47
Lambs fed a milk replacement diet containing 950 Bq ^{137}Cs/kg ration for 21 days. After weaning, lambs were fed silage contaminated with fallout radiocesium plus ionic $^{134}CsCl$	Absorption during the first 21 days was about 90%, equivalent to 975 Bq ^{137}Cs/kg BW. During the silage feeding period, uptake of ionic ^{134}Cs was about twice that of fallout ^{134}Cs	48

Table 18 Locale, radionuclide, sample, and other variables	Concentration	Reference[a]
for 3 weeks	present in silage	
Red fox, *Vulpes vulpes*; muscle; November 1986-February 1987; vixen		
^{134}Cs	176 FW	45
^{137}Cs	461-643 FW	45

[a]1, Allaye-Chan et al. 1990; 2, Kedhi 1990; 3, Crete et al. 1990; 4, Kliment 1991; 5, Conkic et al. 1990; 6, Korhonen 1990; 7, Rissanen and Rahola 1989; 8, Rissanen and Rahola 1990; 9, Daburon et al. 1989; 10, Clooth and Aumann 1990; 11, Handl et al. 1990; 12, Douka and Xenoulis 1991; 13, Sawidis 1988; 14, Assimakopoulos et al. 1989; 15, Ionannides and Pakou 1991; 16, Tonelli et al. 1990; 17, Calamosca et al. 1990; 18, Cristaldi et al. 1990; 19, Spezzano and Giacomelli 1991; 20, Battiston et al. 1991; 21, Imanaka and Koide 1990; 22, Aii et al. 1990; 23, Voors and Van Weers 1991; 24, Vreman et al. 1989; 25, Brittain et al. 1991; 26, Hove et al. 1990a; 27, Skogland and Espelien 1990; 28, Eikelmann et al. 1990; 29, Mathiesen et al. 1990; 30, Grzybowska 1989; 31, Baeza et al. 1991; 32, Palo et al. 1991; 33, Johanson 1990; 34, Bothmer et al. 1990; 35, Eriksson 1990; 36, Johanson et al. 1989; 37, Cristaldi et al. 1991; 38, Jones 1990; 39, Jones et al. 1989; 40, Mascanzoni et al. 1990; 41, Othman 1990; 42, Coughtrey et al. 1989; 43, Crout et al. 1991; 44, Lowe 1991; 45, Lowe and Horrill 1991; 46, Clark and Smith 1988; 47, Beresford 1989; 48, Moss et al. 1989; 49, Fowler et al. 1987; 50, Whitehead et al. 1988b; 51, Whitehead et al. 1988a; 52, Kalas et al. 1994.

[b] — = no data.

[c] ND = not detectable.

Soil and Vegetation. The radiocesium fallout in Sweden was among the highest in western Europe--exceeding 60,000 Bq/m^2 on Sweden's Baltic coast--and involved mainly upland pastures and forests (Johanson 1990; Brittain et al. 1991; Palo et al. 1991). In Norway, radiocesium deposition from the Chernobyl accident ranged from less than 5,000 to more than 200,000 Bq/m^2 and greatly exceeded the deposition from prior nuclear weapons tests (Hove et al. 1990a). In Italy, heavy rainfall coincident with the passage of the Chernobyl radioactive cloud caused high local deposition of radionuclides in soil, grass, and plants (Battiston et al. 1991). The Chernobyl plume reached Greece on 1 May 1986. A total of 14 gamma emitters were identified in the soil and vegetation in May 1986, and three (^{134}Cs, ^{137}Cs, ^{131}I) were also detected in the milk of free-grazing animals in the area (Assimakopoulos et al. 1989). Radiocesium-134 and ^{137}Cs intake by humans in Germany during 1986-87 was mainly from rye, wheat, milk, and beef (Clooth and Aumann 1990). In the United Kingdom, elevated concentrations of radionuclides of iodine, cesium, ruthenium, and others were measured in the air and in rainwater during 2-5 May 1986 (Smith and Clark 1986). The background-activity concentrations were about three times normal levels in early May, and those of ^{131}I approached the derived emergency reference level (DERL) of drinking water of 5 mSv ^{131}I (equivalent to a thyroid dose of 50 mSv); however, ^{131}I levels were not elevated in foodstuffs or cow's milk (Smith and Clark 1986). Syria--1,800 km from Chernobyl--had measurable atmospheric concentrations of ^{137}Cs and ^{131}I and near-detection limit concentrations of ^{144}Ce, ^{134}Cs, ^{140}La, and ^{106}Ru (Othman 1990). The maximum ^{131}I thyroid dose equivalent received by Syrians was 116 uSv in adults and ^{210} uSv in children; 1 year later, these values were 25 uSv in adults and 70 uSv in a 10-year-old.

The amount of fallout radioactivity deposited on plant surfaces depends on the exposed surface area, the developmental season of the plants, and the external morphology. Mosses, which have a relatively large surface area, showed the highest concentrations of radiocesium. (Table 18). In northern Sweden, most of the radiocesium fallout was deposited on plant surfaces in the forest ecosystem and was readily incorporated into living systems because of browsing by herbivores and cesium's chemical similarity to potassium (Palo et al. 1991). Forest plants seemed to show less decrease than agricultural crops in ^{137}Cs activity over time (Bothmer et al. 1990). For example, the effective retention half-life of ^{137}Cs from Chernobyl was 10-20 days in

herbaceous plants and 180 days in chestnuts, *Castanea* spp. (Tonelli et al. 1990). The radioactive fallout from the Chernobyl accident also resulted in high ^{137}Cs levels in Swedish pasture grass and other forage, although levels in grain were relatively low (Andersson et al. 1990). Radiocesium isotopes were still easily measurable in grass silage that was harvested in June 1986 and used as fodder for dairy cows in 1988 (Voors and Van Weers 1991). The rejection of the first harvests of radiocesium-contaminated perennial pasture and in particular of rye grass (*Lolium perenne*) does not constitute a safe practice because later harvests--even 1 year after the contamination of the field--may contain very high values as in Greece (Douka and Xenoulis 1991).

Aquatic Life. After Chernobyl, the consumption of freshwater fishes by Europeans declined, fish-license sales dropped by 25%, and the sale of fish from radiocesium-contaminated lakes was prohibited (Brittain et al. 1991). Many remedial measures have been attempted to reduce radiocesium loadings in fishes, but none has been effective to date (Hakanson and Andersson 1992). Radiocesium concentrations in muscle of fishes from the southern Baltic Sea increased 3 to 4 times after Chernobyl (Grzybowska 1989), and $^{134+137}Cs$ and ^{106}Ru in fishes from the Danube River increased by a factor of 5; however, these levels posed negligible risk to human consumers (Conkic et al. 1990). Chernobyl radioactivity, in particular ^{141}Ce and ^{144}Ce, that entered the Mediterranean as a single pulse, was rapidly removed from surface waters and transported to 200 m in a few days primarily in fecal pellets of grazing zooplankton (Fowler et al. 1987). Bioconcentration factors (BCF) of ^{137}Cs in fishes from Lake Paijanne, Finland--a comparatively contaminated area--ranged from 1,250 to 3,800; the highest BCF values were measured in the predatory northern pike (*Esox lucius*) 3 years after the Chernobyl accident; consumption of these fishes was prohibited (Korhonen 1990).

After the Chernobyl accident, radiocesium. isotopes were also elevated in trees and lichens that bordered an alpine lake in Scandinavia and in lake sediments, invertebrates, and fishes (Table 18). Radiocesium levels in muscle of resident brown trout (*Salmo trutta*) remained elevated for at least 2 years (Brittain et al. 1991). People who consumed food near this alpine lake derived about 90% of their effective dose equivalent from the consumption of freshwater fish, reindeer meat, and milk. The average effective dose equivalent of this group during the next 50 years is estimated at 6-9 mSv with a changed diet and 8-12 mSv without dietary changes (Brittain et al. 1991).

Wildlife. Reindeer (*Rangifer tarandus*)--*also* known as caribou in North America--are recognized as a key species in the transfer of radioactivity from the environment to humans because (1) the transfer factor of radioactivity from reindeer feed to reindeer muscle is high, (2) lichens--which constitute a substantial portion of the reindeer diet--are efficient accumulators of Sr, Cs, and actinide radioisotopes and, (3) reindeer feed is not significantly supplemented with grain or other feeds of low contamination (Jones et al. 1989; Rissanen and Rahola 1989; 1990; Eikelman et al. 1990; Skogland and Espelien 1990). During 1986-87, about 75% of all reindeer meat from Sweden was unfit for human consumption because ^{137}Cs exceeded 300 Bq/kg FW. In May 1987, the maximum permissible level of ^{137}Cs in Swedish reindeer, game, and freshwater fish was raised to 1,500 Bq/kg FW; however, about 25% of slaughtered reindeer in 1987-89 still exceeded this limit (Ahman et al. 1990b). Concentrations in excess of 100,000 Bq $^{134+137}Cs$/kg FW lichens have been recorded in the most contaminated areas and in the 1986-87 season was reflected in reindeer muscle concentrations of greater than 50,000 Bq/kg FW from the most contaminated areas of central Norway (Roed et al. 1991). Norwegian reindeer with 60,000-70,000 Bq ^{137}Cs/kg FW in muscle receive an estimated yearly dose of 500 mSv (Jones 1990). The maximum radiation dose to reindeer in Sweden after the Chernobyl accident was about 200 mSv/year with a daily dose rate of about 1 mSv during the winter period of maximum tissue concentrations (Jones et al. 1989). In general, reindeer calves had higher ^{131}Cs levels in muscle than adult females (4,700 vs. 2,700 Bq/kg FW) during September 1988, suggesting translocation to the fetus (Eikelmann et al. 1990). Two reindeer herds in Norway that were heavily contaminated with radiocesium had a 25% decline in survival of calves; survival was normal in a herd with low exposure (Skogland and Espelien 1990). Several compounds inhibit uptake and reduce retention of ^{137}Cs in reindeer muscle from contaminated diets, but the mechanisms of the action are largely unknown. These compounds include zeolite--a group of tectosilicate minerals--when fed at 25-50 g daily (Ahman et al. 1990a); ammonium hexacyanoferrate--also known as Prussian Blue or Giese salt--at 0.3-1.5 g daily (Hove et al. 1990b; Mathiesen et al. 1990; Staaland et al. 1990); bentonite--a montmorillonite clay--when fed at 2% of diet (Ahman et al. 1990a); and high intakes of potassium (Ahman et al. 1990a). Much additional work seems needed on chemical and other processes that hasten excretion and prevent uptake and

accumulation of radionuclides in livestock and wildlife. Reindeer herding is the most important occupation in Finnish Lapland and in portions of Sweden (Rissanen and Rahola 1989). Swedish Lapland reindeer herders have experienced a variety of sociocultural problems as a result of the Chernobyl accident. The variability of contamination has been compounded by the variability of expert statements about risk, the change in national limits of Bq concentrations set for meat marketability, and the variability of the compensation policy for slaughtered reindeer. These concerns may result in fewer Lapps becoming herders and a general decline in reindeer husbandry (Beach 1990).

Caribou in northern Quebec contained as much as 1,129 Bq ^{137}Cs/kg muscle FW in 1986-87, but only 10-15% of this amount originated from Chernobyl; the remainder is attributed to fallout from earlier atmospheric nuclear tests (Crete et al. 1990). The maximum concentration of ^{137}Cs in meat of caribou (*Rangifer tarandus granti*) from the Alaskan Porcupine herd after the Chernobyl accident did not exceed 232 Bq/kg FW, and this is substantially below the recommended level of 2,260 Bq ^{137}Cs/kg FW (Allaye-Chan et al. 1990). Radiocesium transfer in an Alaskan lichen-reindeer-wolf (*Canis 1upus*) food chain has been estimated. If reindeer forage contained 100 Bq/kg DW in lichens and 5 Bq/kg DW in vascular plants, the maximum winter concentrations--at an effective half-life of 8.2 years in lichens and 2.0 years in vascular plants--was an estimated 20 Bq/kg FW in reindeer-caribou skeletal muscle and 24 Bq/kg FW in wolf muscle (Holleman et al. 1990).

The radioactive body burden of exposed reindeer and the character of chromosomal aberrations--which was different in exposed and nonexposed reindeer--indicated a genetic effect of radiation from the Chernobyl accident (Roed et al. 1991). Chromosomal aberrations in Norwegian female reindeer positively correlated with increasing radiocesium concentrations in flesh (Skogland and Espelien 1990). The frequency of chromosomal aberrations in reindeer calves from central Norway were greatest in those born in 1987 when tissue loadings were equivalent to fetal doses of 70-80 mSv and lower in 1988 (50-60 mSv) and 1989 (40-50 mSv), strongly suggesting a dose-dependent induction (Roed et al. 1991). Mutagenicity tests with feral rodents have also been used successfully to evaluate the biological effects of the radiation exposure from the Chernobyl accident. Increased mutagenicity in mice (*Mus musculus domesticus*) was evident as judged by tests of the bone-marrow micronucleus at 6 months and 1 year after the accident. Rodents with increased chromosomal aberrations also had ^{137}Cs burdens that were 70% higher 6 months after the accident and 55% higher after 1 year, but elevated radiocesium body burdens alone were not sufficient to account for the increase in mutagenicity (Cristaldi et al. 1990). In bank voles, however, mutagenicity (micronucleated polychromatic erythrocytes) correlated well with the ^{137}Cs content in muscle and in the soil of the collection locale (Cristaldi et al. 1991). The estimated daily absorbed doses (4.2-39.4 uGy) were far lower than those required to produce the same effect in the laboratory (Cristaldi et al. 1991).

For many households in Sweden, moose (*Alces alces*) are an important source of meat (Palo et al. 1991). Radiocesium concentrations in the foreleg muscle of moose in Sweden during 1987-88 were highest in fall when the daily dietary intake of the animals was about 25,000 Bq ^{137}Cs and lowest during the rest of the year when the mean daily intake was about 800 Bq (Bothmer et al. 1990). Cesium-137 levels in moose flesh did not decrease significantly for about 2 years after the Chernobyl accident (Johanson 1990). The selection of food by moose is paramount to the uptake of environmental contaminants and the changes in tissue levels over time. Increased foraging on highly contaminated plant species, such as bilberry (*Vaccinium myrtillus*), aquatic plants, and mushrooms, may account for the increased ^{137}Cs radioactivity in moose (Palo et al. 1991). For reasons that are not yet clear, transfer coefficients of ^{137}Cs from diet to muscle were about the same in moose (0.03) and beef cattle (0.02) but were significantly higher in sheep (0.24) (Bothmer et al. 1990).

The song thrush (*Turdus philomelas*) collected in Spain in November 1986 had elevated concentrations of ^{134}Cs, ^{137}Cs , and ^{90}Sr; the contamination probably occurred in central and northern Europe before the bird's migration to Spain (Baeza et al. 1991). Spaniards who ate songthrushes that were contaminated with radiocesium isotopes usually received about 58 uSv/year, which is well below the current international guidelines (Baeza et al. 1991). Consumption of game or wildlife in Great Britain after the Chernobyl accident probably also do not exceed the annual limits of intake (ALI) based on $^{134+137}$Cs concentrations in game and the numbers of animals that can be eaten in 1 year before ALI is exceeded (Lowe and Horrill 1991). For example, a person that eats hares with 3,114 Bq $^{134+137}$Cs/kg FW in muscle would have to consume 99 hares

before exceeding the ALI; for the consumption of red grouse (3,022 Bq/kg), this number is 441 grouse; and for the consumption of woodcock (55 Bq/kg), it is 45,455 woodcocks (Lowe and Horrill 1991). Rabbits (*Oryctolagus* sp.) from northeastern Italy that were fed Chernobyl-contaminated alfalfa meal (1,215 Bq $^{134+137}$Cs/kg diet) had a maximum of 156 Bq/kg muscle FW of $^{134+137}$Cs, a value much lower than the current Italian guideline of 370 Bq/kg FW for milk and children's food and 600 Bq/kg FW for other food (Battiston et al. 1991). More than 85% of the ingested radiocesium was excreted by rabbits in feces and urine; about 3% was retained (Battiston et al. 1991).

Cesium radioactivity in tissues and organs of the wolverine (*Gulo gulo*), lynx (*Felis lynx*), and Arctic fox (*Alopex logopus*) in central Norway after the Chernobyl accident was highly variable. In general, cesium-137 levels were substantially lower in these carnivores than in lower trophic levels (Ekker et al. 1990), suggesting little or no food-chain biomagnification, and at variance with results of studies of the omnivore and herbivore food chain.

Domestic Animals. Radiocesium isotopes from the Chernobyl accident transferred easily to grazing farm animals (Hove et al. 1990a). Both ^{134}Cs and ^{137}Cs were rapidly distributed throughout the soft tissues after dietary ingestion and were most highly concentrated in muscle (Book 1969; Van Den Hoek 1989). Radiocesium activity in milk and flesh of Norwegian sheep and goats increased 3 to 5 fold 2 years after the accident and coincided with an abundant growth and availability of fungal fruit bodies in which $^{134+137}$Cs levels were as much as 100 times greater than green vegetation (Hove et al. 1990a). In cattle, coefficients of radiocesium transfer from diet to muscle were about 2.5% in adults and 16% in calves; the higher value in calves was probably due to a high availability of cesium from the gastrointestinal tract and to daily uptake of potassium in growing animal muscle (Daburon et al. 1989). There was no correlation between the retention of ^{137}Cs and the pregnancy stage in cattle (Calamosca et al. 1990). Radiocesium concentrations in pork in Czechoslovakia did not decline between 1986 and 1987 because the feed of pigs during this period contained milk byproducts contaminated with $^{134+137}$Cs (Kliment 1991).

Sheep farming is the main form of husbandry in the uplands of western Cumbria and northern Wales, a region that received high levels of radiocesium fallout during the Chernobyl accident. Afterwards, typical vegetation activity concentrations were about 6,000 Bq/kg (down to about 1,000 Bq/kg in January 1989). But concentrations in sheep muscle exceeded 1,000 Bq ^{137}Cs/kg FW, which is the United Kingdom's dietary limit for human-health protection (Crout et al. 1991). Contaminated lambs--which usually had higher concentrations of ^{137}Cs than ewes--that were removed to lowland pastures (<50 Bq/kg vegetation) rapidly excreted radiocesium in feces and urine, and cesium body burdens had an effective half-life of 11 days. This practice should not significantly increase radiocesium levels in the soil and vegetation of lowland pastures (Crout et al. 1991). The absorption and retention of radiocesium by suckling lambs is highly efficient, about 66%. Fecal excretion was an important pathway after the termination of ^{137}Cs ingestion. In weaned animals, the absorption of added ionic cesium was about twice that of cesium fallout after the accident at Chernobyl (Moss et al. 1989). Silver-110m was also detected in the brains and livers of ewes and lambs in the United Kingdom. The transfer of ^{110m}Ag was associated with perennial rye grass that was harvested soon after deposition in 1986. Silver-110m was taken up to a greater extent than ^{137}Cs in liver, but unlike ^{137}Cs, ^{110m}Ag was not readily translocated to other tissues. Other than cesium isotopes and ^{131}I, ^{110m}Ag was the only detected nuclide in sheep tissues (Beresford 1989).

Atmospheric deposition of ^{137}Cs from Chernobyl to vegetation and eventually to the milk of sheep, cows, and goats on contaminated silage was reported in Italy, the Netherlands, Japan, and the United Kingdom (Book 1969; Belli et al. 1989; Pearce et al. 1989; Voors and Van Weers 1989; Aii et al. 1990; Monte 1990). The effective half-life of ^{137}Cs was 6.7 days in pasture grass and 13.6 days in milk (Spezzano and Giacomelli 1991). The average transfer coefficient of $^{134+137}$Cs from Chernobyl from a 70% grass-silage diet to milk of Dutch dairy cows was about 0.250%/liter/day (Voors and Van Weers 1991). In goats (*Capra* sp.), about 12% of orally administered ^{137}Cs was collected in milk within 7 days after dosing (Book 1969).

Iodine-131 was one of the most hazardous radionuclides released in the Chernobyl accident because it is easily transferred through the pasture-animal-milk pathway and rapidly concentrated in the thyroid gland to an extent unparalleled in any other organ. Because of its high specific activity, ^{131}I can transmit a high dose of radiation to the thyroid (Ionannides and Pakou 1991). Iodine-131 levels of 618,000 Bq/kg FW in sheep thyroids from northwestern Greece on 3 July 1986 are similar to maximal ^{131}I concentrations in sheep thyroids in Tennessee in 1957 after global atmospheric fallout from military weapons tests and in London after the Windscale accident (Ionannides and Pakou 1991). Iodine-131 has an effective whole-body half-life of about 24 h and is rapidly excreted from sheep and cows (Assimakopoulos et al. 1989). The effective half-life of ^{131}I was 3.9 days in pasture grass and 5 days in cow's milk (Spezzano and Giacomelli 1991). The transfer coefficients of ^{131}I from vegetation to cow's milk was 0.007% day/L milk; this value was 57 times higher (0.4) in sheep (Monte 1990), but the mechanism to account for this large interspecies difference is not clear.

Effects: Nonionizing Radiations

Living organisms are constantly exposed to nonionizing electromagnetic radiations, including ultraviolet, visible, infrared, radio, and other low energy radiations that form an integral part of the biosphere. Emissions from anthropogenic sources such as radios, microwave ovens, television communications, and radar significantly altered the character of our natural electromagnetic field (Garaj-Vrhovac et al. 1990). Although the primary focus of my review is on ionizing radiations, an assumption that low energy electromagnetic waves cannot elicit significant biological responses would be misleading. For example, behavioral and biochemical changes are reported in rats, monkeys, rabbits, and other laboratory animals after exposure to nonionizing electromagnetic radiations; the severity of the effect is associated with the type and duration of the radiation and with various physicochemical variables (Ghandi 1990). Selected examples follow.

Ultraviolet radiation in mammals causes the aging of skin, making it wrinkled and leathery (Kligman and Kligman 1990). Dermatologists of the late nineteenth century described the devastating effects of sunlight on the skin of farmers and sailors when compared with indoor workers. Photoaged skin has a variety of neoplasms, deep furrows, extensive sagging, and profound structural alterations that are quite different from those in protected, intrinsically aged skin (Kligman and Kligman 1990). Similar results were documented of skin of guinea pigs (Davidson et al. 1991) and rodents (Ananthaswamy and Pierceall 1990; Ronai et al. 1990) after exposure to ultraviolet radiation. Ultraviolet radiation causes eye cancer in cattle (Anderson and Badzioch 1991), interferes with wound healing in guinea pig skin (Davidson et al. 1991), is a potent damaging agent of DNA and a known inducer of skin cancer in experimental animals (Ronai et al. 1990), and interferes with an immune defense mechanism that normally protects against skin cancer (Ananthaswamy and Pierceall 1990). Aquatic organisms exposed to ultraviolet radiation show disrupted orientation, decreased motility, and reduced pigmentation in *Peridinium gatunense*, a freshwater alga (Hader et al. 1990); effects were similar in several species of marine algae (Lesser and Shick 1990; Hader and Hader 1991; Shick et al. 1991). Increased lipid peroxidation rates and a shortening of the life span after ultraviolet exposure were reported in the rotifer *Asplanchna brightwelli* (Sawada et al. 1990). Cells of the goldfish (*Carassius auratus*) were damaged, presumably by DNA impairment, from UV exposure (Yasuhira et al. 1991).

Visible radiation adversely affected survival and growth of embryos of the chinook salmon (*Oncorhynchus tshawytscha*; Eisler 1961), chloroplast structure in the symbiotic marine dinoflagellate *Symiodinium sp*. (Lesser and Shick 1990), and in-vitro growth of cultured mammalian cells (Karu 1990). Infrared radiation contributes significantly to skin photoaging, producing severe elastosis; the epidermis and the dermis were capable of self-restoration when the exogenous injury ceased (Kligman and Kligman 1990).

Investigations of the cellular effects of radiofrequency radiation provide evidence of damage to various types of avian and mammalian cells. These effects involve radiofrequency interactions with cell membranes, especially the plasma membrane. Effects include alterations in membrane cation transport, Na^+/K^+-ATPase activity, protein kinase activity, neutrophil precursor membrane receptors, firing rates and resting potentials of neurons, brain cell metabolism, DNA and RNA syntheses in glioma cells, and mitogenic effects on human lymphocytes (Cleary 1990). Microwaves inhibit thymidine incorporation by DNA blockage in cultured cells of the Chinese hamster; irradiated cells had a higher frequency of chromosome lesions (Garaj-Vrhovac et al. 1990). Microwaves induce teratogenic effects in mice when the intensity of exposure places a thermal burden on the dams and fetuses, reducing fetal body mass and increasing number of resorptions (O'Connor 1990).

Extremely low frequency (ELF) electromagnetic fields--similar to fields that emanate from electrical appliances and the electrical power distribution network, usually less than 300 Hz--are used therapeutically in the healing of human nonunion bone fractures, in the promotion of nerve regeneration, and in acceleration of wound healing (Anderson 1990). ELF electric and magnetic fields produce biological effects, usually subtle, and of low hazard in short-term exposure. These effects include altered neuronal excitability, neurochemical changes, altered hormone levels, and changes in behavioral responses. For example, electric-field perception has been reported in humans, mice, pigs, monkeys, pigeons, chickens, and insects; altered cardiovascular responses in dogs and chickens; and altered growth rate of chicks. No deleterious effects of ELF fields on mammalian reproduction and development or on carcinogenesis and mutagenesis have been documented (Anderson 1990). ELF fields had no effect on the growth of bone in chicks (Coulton and Barker 1991). However, adult eastern newts (*Notophthalmus viridescens*), regenerating amputated forelimbs, had grossly abnormal forelimbs 12% of the time when exposed for 30 days to ELF fields of the type reported to facilitate healing of human bone fractures (Landesman and Douglas 1990). Additional studies are recommended on the biological effects of nonionizing radiations on fishes and wildlife, especially ELF radiations.

Effects: Ionizing Radiations

General

High acute doses of ionizing radiation produce adverse biological effects at every organizational level: molecule, cell, tissue-organ, whole animal, population, community, and ecosystem (ICRP 1977; Whicker and Schultz 1982b; LWV 1985; Hobbs and McClellan 1986; UNSCEAR 1988; Kiefer 1990; Severa and Bar 1991). Typical adverse effects of ionizing radiation include cell death (McLean 1973; LWV 1985; Kiefer 1990), decreased life expectancy (Lorenz et al. 1954; Brown 1966; Hobbs and McClellan 1986; Kiefer 1990; Rose 1992), increased frequency of malignant tumors (Lorenz et al. 1954; ICRP 1977; Hobbs and McClellan 1986; UNSCEAR 1988; Hopewell 1990; Kim et al. 1990; Little 1990; Nagasawa et al. 1990; Raabe et al. 1990; Fry 1991), inhibited reproduction (ICRP 1977; Barendson 1990; Kiefer 1990; Rose 1992), increased frequency of gene mutations ICRP 1977; Whicker and Schultz 1982b; Hobbs and McClellan 1986; Abrahamson 1990; Evans 1990; Kiefer 1990; Thacker 1990; Sankaranarayanan 1991a; 1991b; Rose 1992), leukemia (ICRP 1977; Kiefer 1990), altered blood-brain barrier function (Trnovec et al. 1990), and reduced growth and altered behavior (Rose 1992). Species in kingdoms have a wide variation in sensitivity, and sometimes at low radiation exposures the response is considered beneficial (Luckey 1980; Rose 1992). Overall, the lowest dose rate at which harmful effects of chronic irradiation have been reliably observed in sensitive species is about 1 Gy/year; this value for acute radiation exposures is about 0.01 Gy (Rose 1992).

In general, the primitive organisms are the most radioresistant taxonomic groups and the more advanced complex organisms--such as mammals--are the most radiosensitive (Fig. 7). The early effects of exposure to ionizing radiation result primarily from cell death; cells that frequently undergo mitosis are the most radiosensitive, and cells that do not divide are the most radioresistant. Thus, embryos and fetuses are particularly susceptible to ionizing radiation, and very young animals are consistently more radiosensitive than adults (McLean 1973; Hobbs and McClellan 1986). In addition to the evolutionary position and cell mitotic index, many extrinsic and intrinsic factors modify the response of a living organism to a given dose of radiation. Abiotic variables include the type and energy of radiation, exposure rate, length of exposure, total exposure and absorbed dose, dose rate, spatial distribution of dose, season, temperature, day length, and environmental chemicals; biotic variables include the species, type of cell or tissue, metabolism, sex, nutritional status, sensitizing or protective substances, competition, parasitism, and predation (Whicker and Schultz 1982b; Hobbs and McClellan 1986; UNSCEAR 1988; Kiefer 1990).

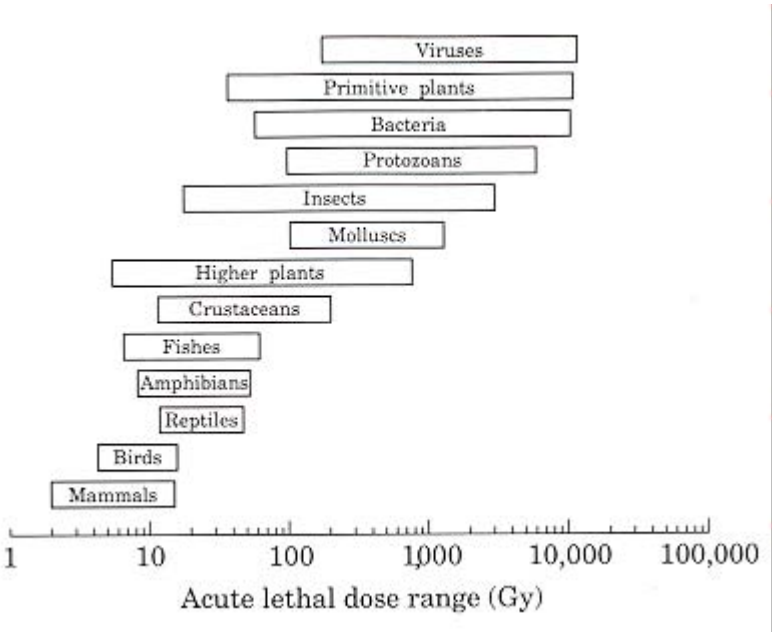

Fig. 7. Acute radiation dose range fatal to 50% (30 days postexposure) of various taxonomic groups (modified from Whicker and Schultz 1982b; Hinton and Scott 1990).

Radiosensitivity of cells is related directly to their reproductive capacity and indirectly to their degree of differentiation (Hobbs and McClellan 1986). Early adverse effects of exposure to ionizing radiation are due mainly to the killing of cells. Cell death may result from the loss of reproductive integrity, that is, when after irradiation a cell fails to pass through more than one or two mitoses. Reproductive death is important in rapidly dividing tissues such as bone marrow, skin, gut lining, and germinal epithelium. When the whole animal is exposed to a large dose of ionizing radiation, some tissues are more prone to damage than others. Death rates of mammalian reproductive cells from ionizing radiations is modified by variations in the linear energy transfer of the radiation, the stage in the cell cycle, cell culture conditions, and sensitizing and protecting compounds (Barendsen 1990). The chemical form of the main stage of the acute radiation syndrome depends on the size and distribution of the absorbed dose. It is determined mainly by damage to blood platelets and other blood-forming organs at 4-5 Gy, to epithelial cells lining the small intestine at 5-30 Gy, and to brain damage at more than 30 Gy; death usually occurs within 48 h at more than 30 Gy (McLean 1973).

Cellular DNA is extremely sensitive to ionizing radiation, although other cell constituents may approach DNA in sensitivity (IAEA 1976; Billen 1990; Kiefer 1990; Lett 1990; Lucke-Huhle et al. 1990; Woloschak et al. 1990a; Shadley et al. 1991). Radiation-induced mutations are explainable on the basis of chromatin and DNA organization in cells and the biophysical properties of ionizing radiation (Sankaranarayanan 1991b). Based on studies of spontaneous and radiation-induced mutations in the mouse (Sankaranarayanan 1991a), more than 67% of the ionizing radiation-induced mutations are lethal and almost all mutations, including enzyme activity variants, dominant visibles, and dominant skeletal mutations, are lethal. These findings are consistent with the view that most radiation-induced mutations in germ cells of mice are due to DNA deletions (Sankaranarayanan 1991a).

Experimental animal data clearly demonstrate that ionizing radiation at relatively high doses and delivered at high dose rates is mutagenic (Hobbs and McClellan 1986). However, radiation-induced genetic damage in the offspring of exposed parents has not been credibly established in any study with humans (Abrahamson 1990). In one human population--the ethnically isolated Swedish reindeer breeding Lapps--elevated concentrations of fallout products have been ingested via the lichen-reindeer-human food chain since the 1950's. However, during 1961-84, incidences of genetic damage did not increase in Lapps (Wiklund et al. 1990).

Radiation is carcinogenic. The frequency of death from cancer of the thyroid, breast, lung, esophagus, stomach, and bladder was higher in Japanese survivors of the atomic bomb than in nonexposed individuals, and carcinogenesis seems to be the primary latent effect of ionizing radiation. The minimal latent period of most

cancers was less than 15 years and depended on an individual's age at exposure and site of cancer. The relation of radiation-induced cancers to low doses and the shape of the dose-response curve (linear or non-linear), the existence of a threshold, and the influence of dose rate and exposure period must be determined (Hobbs and McClellan 1986).

Radioactive materials that gain entry to the body, typically through ingestion or inhalation, exert effects that are governed by their physical and chemical characteristics which, in turn, influence their distributions and retention inside the body. The effective half-life includes physical and biologic half-times. In addition, the type of radiation (i.e., αβ) and its retention and distribution kinetics govern the radiation-dose pattern. In general, the radiation dose from internal emitters is a function of the effective half-life, energy released in the tissue, initial amount of introduced radioactivity, and mass of the organ (Hobbs and McClellan 1986). Retention of radionuclides by living organisms is quite variable and modified by numerous biologic and abiotic variables. For example, ^{137}Cs retention in selected animals varies significantly with the body weight, diet, and metabolism of an organism (Fig. 8). The time for 50% persistence of ^{137}Cs is between 30 and 430 days in ectotherms and was longer at lower temperatures and shortest in summer and under conditions of inadequate nutrition (Hinton and Scott 1990). In mammals, the ^{137}Cs biological half-life was between 6 and 43 days in rodents, dogs, mule deer, reindeer, and monkeys; in humans, this value ranged from 60 to 160 days. The biological half-life of ^{90}Sr ranges from 122 to 6,000 days in ectotherms and is longer at colder temperatures and under laboratory conditions. In mammals and under conditions of chronic intake, the ^{90}Sr biological half-life was 533 days in rats, 750 days in humans, and at least 848 days in beagles (Hinton and Scott 1990).

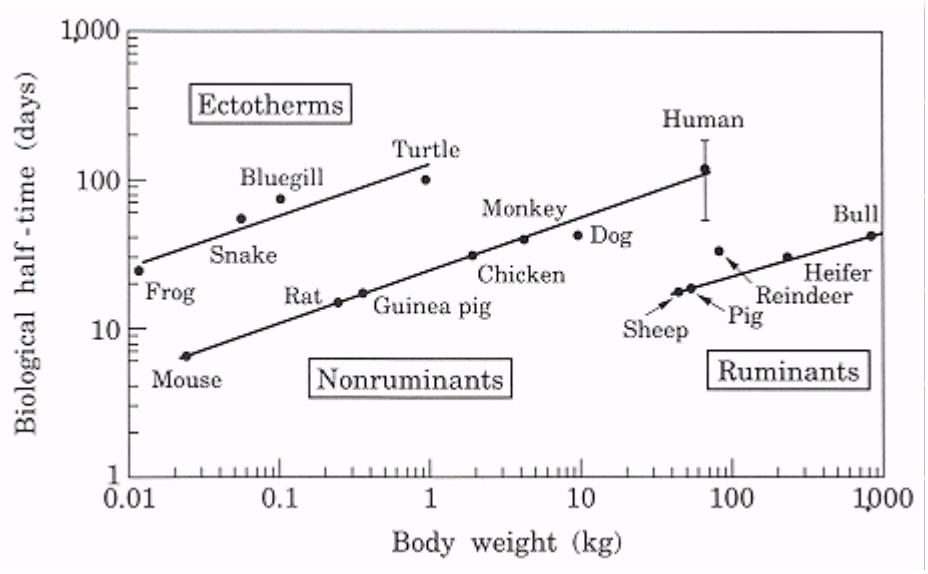

Fig. 8. Relation between diet, metabolism, and body weight with half-life retention of longest-lived component of cesium-137. Data are from selected ruminant and nonruminant mammals (Richmond 1989) and ectotherms (Hinton and Scott 1990).

Terrestrial Plants and Invertebrates

Radiosensitive terrestrial plants exposed to single doses of ionizing radiation had reduced growth at 0.5-1.0 Gy and reduced survival at 3.0-4.1 Gy (Table 19); chronic exposures of 0.2-0.65 Gy/day adversely affected sensitive forest ecosystems (Table 19). Chronic gamma irradiation of 131 Gy/year and higher of mixed forest ecosystems caused the disappearance of trees and shrubs and subsequent erosion of the soil (Poinsot-Balaguer et al. 1991). The radiation sensitivity of five plant communities suggested that pine (*Pinus* spp.) forests were the most sensitive and that deciduous evergreen forests, tropical rain forests, herbaceous rock-outcrop communities, and abandoned cropland were increasingly less sensitive (McCormick 1969). Neutrons were 3 to 4 times more effective than gamma rays in root growth inhibition (Witherspoon 1969). Altitude affects the response of vegetation to ionizing radiation. Peas (*Pisum sativum*) in gardens at 2,225-3,750 m above mean

sea level and exposed to 0, 5, 10, or 50 Gy had reduced growth from all treatments at increasing altitudes; however, a dose-response growth curve was evident only at less than 3,049-m altitude (Osburn 1963). Seeds of tobacco (*Nicotiana tabacum*) exposed to cosmic rays aboard a spacecraft had a higher mutation rate than controls; effects occurred at total doses as low as 0.1-0.2 Gy (Gaubin et al. 1990), but this needs verification.

Table 19. Radiation effects on selected terrestrial plants.

Table 19. Species, dose, and other variables	Effect	Reference[a]
Tropical rainforest tree, *Dacryodes excelsa*, 4 to 280 Gy/year	Growth stimulation	1
Deciduous evergreen forest		
40 Gy yearly	Minor effects	2
100 Gy yearly	Severe sublethal effects	2
350 Gy yearly	Lethal	2
Deciduous plants, 13 species		
4 to 15 Gy, single fast neutron doses	Shoot growth inhibited by >85%	3
60 to 85 Gy, single gamma radiation dose	Shoot growth inhibited by >85%	3
Forest ecosystem, northern Wisconsin, experimentally exposed to a ^{137}Cs point source for 5 months during a growing season. Distance from source (meters) and daily exposure (Gy)		
5 m, 15 Gy	No vegetation	4
5-10 m, 5-15 Gy	Lower plants present	4
10-15 m, 1.5 Gy	Resistant trees and shrubs present	4
10-15 m, 2.5-5.0 Gy	Some growth	4
20-30 m, 0.65-1.5 Gy	Resistant angiosperm trees	4
30-50 m, 0.2-0.65 Gy	Angiosperm trees present	4
50 m, 0.2 Gy	Original northern forest	4
Herbaceous rock outcrop community		
90 Gy yearly	Minor effects	2
400 Gy yearly	Severe sublethal effects	2
1,000 Gy yearly	Lethal	2
Mango, Queensland *Mangifera indica,* fruit irradiated postharvest, single dose, 250 or 750 Gy	At 250 Gy, skin and pulp color inhibited 50% due to irradiation-induced suppression of chlorphyll breakdown and reduction in carotenoid production. At 750 Gy, fruit respiration increased for 3-5 days, but no effect on fruit firmness	5
Mixed oak forest, southern France, experimentally irradiated for 18 years by a ^{137}Cs source at dose rates between 0.3 and 116 mGy/h, equivalent to a yearly rate between 2.6 and 1,016 Gy	At 60-100 mGy/h (525-876 Gy yearly), all trees, shrubs, and litter were absent; low overall insect density; soil deficient in carbon, nitrogen, and water. At 15 mGy/h (131 Gy yearly), woody	6

Table 19. Species, dose, and other variables	Effect	Reference[a]
	plants were present, but visibly abnormal	
Tobacco, *Nicotiana tabacum,* 55 Gy/year	Growth stimulation	1
Pine forest		
1-10 Gy yearly	Minor effects	2
20 Gy yearly	Severe sublethal effects	2
30 Gy yearly	Lethal	2
Slash pine, *Pinus elliottii,* acute single exposure of 3 Gy	50% dead 1-4 months after exposure; no other deaths in 2 years	2
Sugar pine, *Pinus lambertiana,* acute single exposure of 4.1 Gy	LD50 (30 days postexposure)	1
Longleaf pine, *Pinus palustris*		
0.5 Gy, single dose	Growth inhibition	1
8 Gy, acute single exposure	50% of trees <5 years old died in 1-4 months; others survived for at least 2 years	2
>28 Gy, acute single exposure	Fatal to 50% of trees >5 years old in 1-4 months; no other deaths in 2 years	2
Winter wheat, *Triticum aestivum,* acute single exposure of 1.0 Gy	Growth inhibition	1
Tropical rainforest		
70 Gy yearly	Minor effects	2
350 Gy yearly	Severe sublethal effects	2
400 Gy yearly	Lethal	2
Vegetation, abandoned crop land		
50 Gy yearly	Minor effects	2
450 Gy yearly	Severe sublethal effects	2
1,500 Gy yearly	Lethal	2
Bean, *Vicia faba,* 58-100 Gy/year	Growth stimulation	1

[a]1, Rose 1992; 2, McCormick 1969; 3, Witherspoon 1969; 4, Zavitokovski and Rudolph 1971; 5, Boag et al. 1990; 6, Poinsat-Balaguer et al. 1991.

Sometimes, irradiation prevents the usual colonizing vegetation from becoming established (Poinsot-Balaguer et al. 1991). Germination and survival of shrub seedlings have been much slower in nuclear test sites than in undisturbed sites (Romney et al. 1971). The return to its original state of the perennial shrub vegetation takes decades on a radiation-disturbed site, although native annual species and grasses have grown abundantly within 12 months. Transplanting of shrubs into radiation-disturbed areas has been largely unsuccessful because of intense browsing by rabbits and other small mammals (Romney et al. 1971). A nuclear detonation damages terrestrial vegetation by heat, blast, or radiation. Plant injury from thermal or ionizing radiation at an above-ground detonation site varied with stem rigidity and stability of the substratum, although radiation effects are ordinarily masked by damage from blasts. A typical nuclear detonation at the Nevada test site--an airburst of a 20- to 40-kiloton yield--denuded a zone of desert within a 0.8-km radius of shrub vegetation. Recovery at the Nevada site seemed complete within 4 years, suggesting little relation between fatal

injury, morphological aberration in vegetation, and ionizing radiation from nuclear detonations (Shields and Wells 1963). A northern Wisconsin forest experimentally subjected to a ^{137}Cs radiation source for 5 months showed several trends: (1) herbaceous and shrub species with a spreading form of growth are more radioresistant than upright forms; (2) larger pines and oaks are more radioresistant than smaller trees; (3) perennial plants with shielded buds and vigorous asexual reproduction are relatively radioresistant; (4) plants that are adapted to extreme habitats such as old fields and granite rock outcrops and plants that are typical of early successional stages are relatively radioresistant; (5) all plants are more radiosensitive during the growing season than during the dormant season; and (6) reproductive stages are always more radiosensitive than vegetative stages (Zavitokovski and Rudolph 1971). The recovery of vegetation in a tropical rain forest in Puerto Rico--after plants were deliberately subjected to lethal doses of gamma radiation--closely resembled secondary succession after other types of disturbances such as mechanical stripping and treatment with the Picloran herbicide (Jordan 1969).

Hormesis--the beneficial physiological stimulation by low doses of a potentially harmful agent--is documented of ionizing radiation and many species of terrestrial plants and invertebrates (Luckey 1980). Radiation hormesis in plants includes increased germination, growth, survival, and yield. Some species of terrestrial invertebrates had increased fecundity, growth, survival, disease resistance, and longevity after exposure to low sublethal doses of ionizing radiation (Luckey 1980).The growth and development of some terrestrial invertebrates are stimulated at comparatively high sublethal acute doses (i.e., 2 Gy in silkworm [*Bombyx mori*]), but survival is reduced at 10 Gy; in all cases, younger stages were the most sensitive (Table 20). Cockroaches (*Blaberus giganteus*) adapted to the dark reportedly can visually detect radiation sources as low as 0.001 mGy (Rose 1992), however, the mechanisms are not understood.

Table 20. Radiation effects on selected terrestrial invertebrates.

Species, dose, and other variables	Effect	Reference[a]
Caribbean fruit fly, *Anastrepha suspensa*; larvae, heavily parasitized by the hymenopteron *Diachasmimorpha longicaudata*, exposed to single acute exposures of 10 to 70 Gy	50% of control flies developed into adults vs. 25% at 10 Gy, and <1% at 30 Gy. At 40 Gy and higher, no adults were recovered but parasite development was the same at all doses	1
Silkworm, *Bombyx mori*; eggs, acute single exposure of 2, 5, or 10 Gy	At 2 Gy, an average increase of 23% in larval mass, cocoon shell weight, and silk production; no stimulatory effect at 5 Gy; at 10 Gy, larval development inhibited	2
Mediterranean fruit fly, *Ceratitis capitata*; females, acute single exposure of 150-155 Gy	Inhibited oviposition	3
Moth, *Ectomyelois ceratoniae*; male pupae, age 3 or 5 days, acute single exposure of 50 to 500 Gy	Younger pupae were more sensitive than older pupae. Only 3% of pupae developed into adults at 500 Gy. At >250 Gy, progeny development reduced 50%. Normal fecundity at 100-250 Gy when mated with control females	4
Leafmining fly, *Liriomyza trifolii*; immature stages, on artificially infested bean	All dead at >750 Gy; 80% dead at 250 Gy; eggs and prepupae were the most sensitive	5

Species, dose, and other variables	Effect	Reference[a]
seedlings; acute single exposure of 25 to 2,000 Gy	stage; no phytotoxic effects	

[a] 1, Sivinski and Smittle 1990; 2, Yusifov et al. 1990; 3, McInnes and Wong 1990; 4, Al-Izzi et al. 1990; 5, Yathom et al. 1990.

Following the successful application of radiation to sterilize male screw-worm flies (*Cochliomyia hominivorax*), various insect pests became the target of similar techniques throughout the world (Al-Izzi et al. 1990). The technique has suppressed populations of the Mediterranean fruit fly (*Ceratitis capitata*), a major pest of fruits, although results have not been as spectacular as with the screw-worm fly (McInnes and Wong 1990). The pestiferous Caribbean fruit fly (*Anastrepha suspensa*), heavily parasitized by a beetle, became sterile after acute exposures to ionizing radiation, although beetles remained fecund. Mass rearing and inundative release of the radioresistant beetle parasite is now considered an option for control of the Caribbean fruit fly (Table 20).

Aquatic Organisms

Among aquatic organisms, it is generally acknowledged that primitive forms are more radioresistant than complex vertebrates and that older organisms are more resistant than the young (Donaldson and Foster 1957; Bonham and Welander 1963; Templeton et al. 1971; Table 21). Developing eggs and young of some species of freshwater fishes are among the most sensitive tested aquatic organisms; death was observed at acute doses of 0.3-0.6 Gy, and minor effects on physiology or metabolism were observed at chronic daily dose rates of 0.01 Gy (Bonham and Welander 1963; Templeton et al. 1971; IAEA 1976; Table 21). Radiosensitivity correlated positively with the metabolic rate of the dividing cell, which accounts for the radioresistance of dormant eggs of aquatic invertebrates and the general sensitivity of early embryonic stages of all aquatic species (Donaldson and Foster 1957; Table 21).

Table 21. Radiation effects on selected aquatic organisms.

Table 21. Taxonomic group, organism, dose, and other variables	Effect	Reference[a]
Algae		
Diatom, *Nitzchia closterium*; acute single exposure of 100 Gy	Lethal	1
Euglena, *Euglena gracilis*; acute single exposure of 550 Gy	Tolerated	1
Freshwater algae, seven species, held in water containing 1,110 Bq ^{226}Ra/L for as long as 14 days	After 24 h, four species (*Ankistrodesmus falcatus*, *Chlorella vulgaris*, *Coelastrium cambricum*, *Scenedesmus obliquus*) had decreased oxygen production by 22-37%; after 14 days, no effect on growth or protein content	2
Various species, single acute exposure		
80-1,000 Gy	LD50, 45 days, postexposure	3
250-6,000 Gy	LD100, 45 days after single exposure	3
Protozoans, various species, acute single exposure		

Table 21. Taxonomic group, organism, dose, and other variables	Effect	Reference[a]
100-300 Gy	LD50, up to 40 days postexposure	3
180-12,500 Gy	LD100, up to 40 days postexposure	3
Coelenterates		
Sea anemone, *Anthopleura xanthogrammica*; 0.2 Gy, acute single exposure	Tentacles withdrawn	1
Jellyfish, *Aurelia aurita*; acute single exposure		
50-150 Gy	No deaths in 60 days	4
50-400 Gy	Dose-dependent increase in developmental abnormalities and abnormal budding rates and patterns	4
100 Gy	Metamorphosis and budding inhibited; reduction in pulsation rate	4
150 Gy	Inhibited reproduction	4
200 Gy	60% died in 60 days	4
400 Gy	90% died in 30 days	4
Molluscs		
Water snail, *Physa heterostropha*; exposure of 2.4-5.5 Gy daily for 1 year	Increased growth rate	1
Various species, acute single exposure		
50-200 Gy	LD50, up to 2 years postexposure	3
100-500 Gy	LD100, up to 2 years postexposure	3
Crustaceans		
Brine Shrimp, *Artemia salina*; acute single exposure		
0.004 Gy	No adverse effects on development of cysts	5
0. 1-9 Gy	Decreased development when exposed as cysts	5
4.5-9 Gy	LD50, nauplii, 20-25 days postexposure	6
130 Gy	LD50, adults, 25 days after exposure	6
486-2,084 Gy	Dose-dependent delay in development of eggs	7
3,000 Gy	LD50, cysts	5
Blue crab, *Callinectes sapidus*; continuous exposure to 0.76 Gy daily for 1 year	Increased growth rate	1
Shore crab, *Carcinus maenus*		
Americium-241, dose unknown	After 8 days, bioconcentration factors (BCF) were 145 in whole crab, 960 in gills, and 240 in exoskeleton; 50% elimination in 45 days	8
Plutonium-237, dose unknown	After 8 days, BCF values were	8

Table 21. Taxonomic group, organism, dose, and other variables	Effect	Reference[a]
	75 in whole crab, 340 in gills, and 70 in exoskeleton; 50% elimination in 55 days	
Daphnid, *Daphnia pulex*; daily exposure to 8.2-17.8 Gy for 1 year	Increased growth rate	1
Various species, single acute exposure		
5-900 Gy	LD50, up to 80 days postexposure	3
50-800 Gy	LD100, up to 80 days postexposure	3
Annelids		
Polychaete, *Neanthes arenaceodentata*; acute single exposure		
1-4 Gy	Adverse effects on reproduction	9
2-100 Gy	Significant increase in frequency of chromosomal aberrations	9
>100 Gy	Decreased life span	9
>500 Gy	Death	9
Fishes		
Common carp, *Cyprinus carpio*		
Adults, 3 Gy, acute single exposure	No effect on growth	1
Fertilized eggs, exposed through hatch		
144 million Bq ^{238}Pu/L	Increased abnormalities	10
277 million Bq ^{238}Pu/L	Decreased hatch	10
44 million Bq ^{232}U/L	Increased abnormalities	10
815 million Bq ^{232}U/L	Decreased hatch	10
Anchovy, *Engraulis* sp.; fertilized eggs, ^{90}Sr-^{90}Y, continuous exposure		
7.4 Bq/L	Increased developmental abnormalities	10
740 Bq/L	Decreased hatch, retarded growth rate	10
Fishes, various species, acute single exposure		
6-30 Gy	LD50, up to 460 days postexposure	3
3.7-200 Gy	LD100, up to 460 days postexposure	3
Pinfish, *Lagodon rhomboides*; exposure to 0.197 Gy daily for 1 year	Increased growth rate	1
Bluegill, *Lepomis macrochirus*; acute single exposure of 10, 20, or 30 Gy	At 20 and 30 Gy, serum proteins were reduced more than 50% within 24 h; damage to the GI capillary system and injury to the gastroepithelium accounted for the excessive protein loss	12
Marine teleosts, 6 species,	LD50	11

Table 21. Taxonomic group, organism, dose, and other variables	Effect	Reference[a]
10-55 Gy, acute single exposure		
Silver salmon, *Oncorhynchus kisutch*; acute single exposure		
Early embryonic stages, 0.3-0.6 Gy	LD50 at hatch	13
Later embryonic stages, 9.2-18.7 Gy	LD50 at hatch	13
Rainbow trout, *Oncorhynchus mykiss*		
Embryos, acute single exposure		
0.6 Gy, 1-cell stage	LD50 by end of yolk resorption	10
0.8 Gy, 1-cell stage	LD50 at hatch	3
3.1 Gy, 32-cell stage	LD50 by end of yolk resorption	10
4.1 Gy, early eyed stage	LD50 by end of yolk resorption	3
4.6 Gy, 32-cell stage	LD50 by hatch	3
9.0 Gy, late eyed stage	LD50 by end of yolk resorption	3
Embryos held in water containing 370 million Bq/L of ^{3}H from immediately after fertilization through hatching	No effect on hatching abnormalities	10
Embryos held in water containing 37 million Bq/L of ^{3}H from 6 h after fertilization through hatch	Suppressed immune response of fry	10
Immatures, single acute exposure of 0.2 Gy	Growth stimulation	1
Juveniles exposed for 27 days to radioneptunium isotopes and analyzed 2-15 days postexposure	Maximum BCF values were 8.7 for whole fish, 1.1 for skin, and 0.34 for muscle	14
Yearlings, force-fed 185,000, 1.85 million, or 18.5 million Bq ^{90}Sr-^{90}Y/kg BW daily for 21 weeks	At highest dose, adverse effects on growth (week 12) and survival (week 15); survivors had leucopenia and gut histopathology, and concentrations of 9.2 billion Bq/kg FW in bone and 9.99 million Bq/kg FW in muscle. Residues in the 1.85 million group were 1.04 billion Bq/kg in bone and 2.96 million Bq/kg in muscle. For the 185,000 group, these values were 77.7 million Bq/kg in bone and 74,000 Bq/kg in muscle	11,15
Yearlings force-fed 370,000, 3.7 million, or 37 million Bq ^{65}Zn/kg BW daily for 17 weeks, or 370 million Bq ^{65}Zn/kg BW daily	Adverse effects on growth, survival, or gut histology at any dose; leucopenia evident at week 10 at the highest dose. Residues, in Bq/kg FW,	11,15

Table 21. Taxonomic group, organism, dose, and other variables	Effect	Reference[a]
for 10 weeks	in the 37 million group at 17 weeks were 148 million in bone and 12.9 million in muscle	
Yearlings force-fed 222,000, 2.2 million, or 22.2 million Bq ^{32}P/kg BW daily for as long as 25 weeks	At highest dose tested, adverse effects on growth, survival, and gut histology between day 17 and 77. In the intermediate 2.2 million group, adverse effects on growth at 17 weeks; residues (Bq/kg FW) were 66.6 million in bone and 8.5 million in muscle. The 220,000 group had no adverse effects in 25 weeks on growth, survival, or tissue alterations	11
Gametes of adults, single acute exposure of 0.5-1.0 Gy	50% reduction in fecundity	3
Adults, single acute exposure of 15 Gy	LD50	3
Chinook salmon, *Oncorhynchus tshawytscha*		
0.0004 Gy/h, eggs, 81-day exposure, total dose of 0.78 Gy	Significant adverse effects on survival and development	21
0.005 Gy daily, continuous exposure from egg fertilization through yolk sac absorption; total dose of 0.35 Gy	No adverse effects on growth and survival or on numbers of returning adults after seaward migration	22
0.028 Gy daily, continuous exposure from egg fertilization through yolk sac absorption; total dose of 1.99 Gy	No adverse effects observed prior to seaward migration	22
0.2 Gy, single acute exposure	Growth increase	1
10 Gy, eyed eggs, single acute exposure	LD50	3
12.5-25 Gy, fingerlings, single acute exposure	LD50	3
Medaka, *Oryzias latipes*; adult males receiving single acute exposure of 0.64,4.75, or 9.5 Gy	Dose-dependent increase in total mutations in sperm, spermatids, and spermatogonia	16
Sea lamprey, *Petromyzon marinus*; males captured during spawning run, single acute exposure		
20 Gy	LD50, 45 days postexposure; survivors sterile	17

Table 21. Taxonomic group, organism, dose, and other variables	Effect	Reference[a]
30 Gy	All died before spawning,	17
Fathead minnow, *Pimephales promelas*; developing eggs, continuous exposure		
4,440 Bq ^{144}Ce-^{144}Pr/L	No effect on embryonic development or hatch	10
9.6 million Bq ^{238}Pu/L	Increased abnormalities	10
48.1 million Bq ^{238}Pu/L	Decreased hatch	10
7.4 million Bq ^{232}U/L	Increased abnormalities	10
18.5 million Bq ^{232}U/L	Decreased hatch	10
Atlantic salmon, *Salmo salar*; fertilized eggs, continuous immersion in 92.5 Bq ^{137}Cs/L or 185 Bq ^{90}Sr/L	Increased mortality of embryos and fry	10
Brown trout, *Salmo trutta*		
Fertilized eggs continuously immersed in water containing 3.7 million Bq/L of ^{90}Sr-^{90}Y through hatch	No effect on hatch or developmental abnormalities	10
Juveniles held in water containing 30,000 Bq ^{110m}Ag/L for 57 days, then transferred to uncontaminated media for 28 days	At day 57, whole trout contained 105,000 Bq ^{110m}Ag/kg FW; about 70% was in liver. No detectable radioactivity after depuration for 28 days	18
Juveniles fed diet containing 3,343,000 Bq ^{110m}Ag/kg for 1 week (5 times weekly), then 269,000-296,000 Bq ^{110m}Ag/kg diet between weeks 2 and 5. Depuration for 28 days	At end of exposure, whole trout contained 27,400 Bq ^{110m}Ag/kg, equivalent to 11.7% of ingested radioactivity; liver accounted for 63%. No detectable radioactivity after depuration for 28 days	19
Integrated study		
Artificial stream simulating outfall from Czechoslovakian nuclear power plant, 28-day exposure, strontium-90		
Water	894 Bq/L	20
Sediments	1,589-2,288 Bq/kg FW	20
Alga, *Cladophora glomerate*	Max. 22,106 Bq/kg FW	20
Snail, *Planorbis corneus*, shell vs. soft parts	760,588 Bq/kg FW vs. 27,468 Bq/kg FW	20
Common carp, *Cyprinus carpio*		
Bone	29,144 Bq/kg FW	20
Muscle	580 Bq/kg FW	20

Table 21. Taxonomic group, organism, dose, and other variables	Effect	Reference[a]
Scales	13,101 Bq/kg FW	20
Uncontaminated site		
Water	0.002-0.005 Bq/L	20
Common carp, internal organs vs. scales	0.1-0.5 Bq/kg FW vs. 1.5-9.3 Bq/kg FW	20

[a] l, Rose 1992; 2, Havlik and Robertson 1971; 3, Donaldson and Foster 1957; 4, Prokopchak et al. 1990; 5, Gaubin et al. 1990; 6, Engel and Davis 1976; 7, Su et al. 1990; 8, Guary and Fowler 1990; 9, S. L. Anderson et al. 1990; 10, Whicker and Schultz 1982b; 11, Templeton et al. 1971; 12, Ulrickson 1971; 13, Bonham and Welander 1963; 14, Poston et al. 1990; 15, IAEA 1976; 16, Shima and Shimada 1991; 17, Hanson 1990; 18, Garnier et al. 1990; 19, Garnier and Baudin 1990; 20, Stanek et al. 1990; 21, National Council on Radiation Protection and Measurements (NCRP) 1991; 22, Donaldson and Bonham 1970.

Adverse effects on the fecundity of sensitive aquatic vertebrates were detected at dose rates as low as 0.4 mGy/h; adverse effects on fecundity were measured only at dose rates of greater than 1.0 mGy/h. Thus, deleterious effects in populations of aquatic vertebrates are probably not detected until the 0.4-1.0 mGy/h dose rate is exceeded (NCRP 1991). Organisms such as estuarine organisms that are exposed to variable physicochemical conditions are more radioresistant than those in buffered environments, and this may be due to a higher degree of genetic polymorphism in species of fluctuating environments (IAEA 1976). Estimated dose effects from the induction of chromosomal aberrations in polychaete annelid worms were dependent on cell stage at time of irradiation (S.L. Anderson et al. 1990). For reproduction, the estimated dose effects were dependent on the potential regeneration of gonadal tissue (S.L. Anderson et al. 1990). Radiation causes dominant lethal mutations in the medaka (*Oryzias latipes*; Shima and Shimada 1991). Ionizing radiation at low-level chronic exposure reportedly has no deleterious genetic effects on aquatic populations because exposure is compensated by density-dependent responses in fecundity (IAEA 1976); however, this needs verification.

Accumulation of radionuclides from water by aquatic organisms varies substantially with ecosystem, radionuclide, and trophic level (Tables 22-24); with numerous biological, chemical, and physical variables; and with proximity to sources of radiation (Bowen et al. 1971; Lowman et al. 1971; Templeton et al. 1971; Mo and Lowman 1976; Shure and Gottschalk 1976; Whicker and Schultz 1982b; Becker 1990; Poston et al. 1990; Joshi 1991). Accumulated radionuclides inside embryos of the scorpionfish (*Scorpaena porcus*) and the turbot (*Scopthalmus maeoticus maeoticus*) increased the frequency of nuclear disruptions in these species; ^{90}Sr-^{90}Y and ^{91}Y had greater cytogenetic effects than other tested radionuclides (Polikarpov 1973). In the absence of site-specific data, the U.S. Nuclear Regulatory Commission recommends the use of listed concentration ratios--the concentration of the element in the organism (in mg/kg FW) divided by the concentration in the medium (in mg/L)—for various elements in marine and freshwater fishes and invertebrates (Whicker and Schultz 1982b). However, the commission clearly indicates that these values are only approximations.

Table 22. Concentration factors[a] for cesium-137 and strontium-90 in aquatic organisms (Whicker and Schultz 1982a).

Radionuclide and ecosystem	Molluscs, whole	Crustaceans, whole	Fishes, muscle
Cesium-137			
Freshwater	600	4,000	3,000
Marine	8	23	15
Strontium-90			
Freshwater	600	200	200
Marine	1	3	0.1

[a] Bq per gram fresh weight sample/Bq per mL medium.

Table 23. Maximum concentration factors[a] reported for selected elements in marine organisms at various trophic levels (Bowen et al. 1971).

Element	Algae	Grazers	Predators
Ag	1,000	20,000	3,000
Cd	6,000	2,000,000	10,000
Ce	4,500	300	12
Co	1,000	10,000	50,000
Cr	600	300,000	3,900
Cs	50	15	10
Fe	70,000	300,000	30,000
I	7,000	70	10
Mo	200	175	200
Mn	20,000	60,000	100,000
Ni	1,000	10,000	80
Pb	3,000,000	2,000,000	200,000
Ru	1,000	16	10
Sr	90	85	5
Ti	30,000	20,000	3,000
Zn	3,000	100,000	20,000
Zr	20,000	30,000	40,000

[a] Bq per gram fresh weight tissue/ Bq per mL seawater.

Table 24. Approximate maximum concentration factors[a] for selected transuranics in marine sediments, macroalgae, and fishes (Morse and Choppin 1991).

	Concentration factor		
Transuranic nuclide	Sediments	Macroalgae	Fish
Neptunium	1,000	5,000	10
Plutonium	100,000	2,000	40
Americium, curium, berkelium, californium	2,000,000	8,000	50

[a] Bq per gram fresh weight sample/Bq per mL water.

After more than 400 atmospheric nuclear test explosions and the fallout from Chernobyl, ^{137}Cs became the most frequently released nuclear fission product throughout central Europe (Jandl et al. 1991). Cesium behaves like potassium; it has a ubiquitous distribution inside the body, especially in soft tissues. In the common snail *Helix pomatia*, the biological half-time after a single 24-h dietary dose was 2.5 days by the short-lived component and 28.5 days by the long-lived component (Jandl et al. 1991). Concentration factors (CF) of ^{137}Cs in muscle (ratio of Bq/kg FW muscle:Bq/L filtered seawater) of marine fishes from the North Sea between 1978 and 1985 ranged from a low of 39 in the plaice (*Pleuronectes platessa*) to a high of 150 in the whiting (*Merlangius merlangius*); CF values were intermediate in the haddock (*Merlanogrammus aeglifinus*; CF of 58) and in the Atlantic cod (*Gadus morhua*; CF of 92). These data seem to support the use of a CF of 100 for ^{137}Cs in muscle of marine fishes in generalized assessments, although some adjustment is necessary when particular species, such as whiting, form the bulk of a consumer's diet (Steele 1990). In the Great Lakes, the maximum CF values of ^{137}Cs range from 1,000 to 10,000 in algae, amphipods, and fishes and from 100 to 1,000 in zooplankton (Joshi 1991). Maximum concentration factors of ^{137}Cs in a contaminated creek in South Carolina were 4,243 in suspended particulates, 938 in detritus, 4,496 in algae and macrophytes, 997 in omnivores, 1,292 in primary carnivores, and 1,334 to 2,595 in top carnivores such as the redbreast sunfish (*Lepomis auritis*), largemouth bass (Mi*cropterus salmoides*), and water snakes (*Natrix* spp.; Shure and Gottschalk 1976). Cesium

uptake by oligochaete worms (*Limnodrilus hoffmeisteri*) *is* inhibited by low temperatures, potassium concentrations of greater than 1 mg/L and the presence of bacteria (*Escherichia coli*) that compete with the worms for ^{137}Cs (Steger and Goodnight 1976).

Atmospheric fallout from nuclear testing is the main pathway by which transuranic nuclides such as Np, Pu, Cm, and Am enter the aquatic environment (Guary and Fowler 1990). In general, transuranics are strongly partitioned onto particulates. Living organisms are less enriched than particulate matter by as much as 1,000 times, and concentration factors by marine biota are similar for transuranics beyond neptunium (Morse and Choppin 1991). The uptake of ^{241}Am and ^{244}Cm from contaminated sediments by a freshwater amphipod (*Hyalella* sp.) and oligochaete (*Tubifex* sp.) is reported, presumably by way of adsorption, and this is considered the principal uptake pathway by benthic organisms in freshwater and in marine ecosystems (Sibley and Stohr 1990). Transuranics that were ingested with food by various crabs were initially excreted with feces; the remaining transuranics entered a soluble radionuclide pool inside the animal that was slowly excreted. Decapod crustaceans assimilate and retain 10-40% of the tranuranic nuclides in their diets. Initially, absorbed radionuclides accumulate in the hepatopancreas but are then translocated to other tissues, particularly to tissues of the exoskeleton; accordingly, molting strongly influences elimination in crustaceans (Guary and Fowler 1990). Neptunium isotopes have a higher potential for environmental transport in aquatic systems and in groundwater than other tested actinides. Laboratory studies with ^{235}Np and ^{237}Np, for example, showed concentration factors between 275 and 973 in a green alga (*Selenastrum capricornutum*), 32 and 72 in a daphnid (*Daphnia magna*), 2 in an amphipod (*Gammarus* sp.) and in juvenile rainbow trout, 8.7 in carcass, 1.1 in skin, and 0.3 in muscle during a 96-h period (Poston et al. 1990). When the much higher biological effectiveness of alpha than of beta or gamma radiation is considered, plutonium isotopes may contribute more artificial radiation dose equivalent to marine invertebrates than either ^{90}Sr or ^{137}Cs. Concentration factors of ^{239}Pu and marine organisms ranged from 300 to 100,000 in seaweeds, 250 to 690 in molluscs, 760 to 1,020 in echinoderms, 2,100 in sponges, and as much as 4,100 in worms (Noshkin et al. 1971). Concentration factors of $^{239+240}Pu$ in Lake Michigan were between 1 and 10 in predatory salmonids, 10 and 300 in nonpredatory fishes, 900 and 1,200 in amphipods and shrimp, about 200 in zooplankton, and about 6,000 in algae (Joshi 1991).

Iodine-131 (half-life of 8 days) may cause deleterious effects in marine teleosts--although ^{131}I concentrations in tissues were not detectable. In one case, coral reef fishes from Eniwetok Atoll collected as long as 8 months after a nuclear explosion had thyroid necroalteration, suggesting a thryotoxic level of ^{131}I in the environment. Laboratory studies with teleosts injected with ^{131}I showed similar signs of histopathology. Herbivorous fishes and species that habitually consumed bivalve molluscs were most severely affected (Gorbman and James 1963).

Strontium-90 is an anthropogenic radionuclide in liquid effluents from some European nuclear power plants. Algae and sediments are the most important accumulators of ^{90}Sr, although levels in gastropods and fish bone and scales are also elevated, suggesting piscine uptake through gills and skin (Stanek et al. 1990). Fishes tend to accumulate calcium more than strontium, even when Ca levels in food and water were low. Gill tissue was the most and gut the least discriminatory against Sr. Strontium assimilation was linked to the Sr:Ca ratio in food and water, amounts of Ca derived from each source, and biological discrimination against Sr relative to Ca (Ophel and Judd 1976). The ability of organisms to discriminate between strontium radioisotopes is also documented. In one case, ^{85}Sr was taken up rapidly in bluegill (*Lepomis macrochirus*) muscle and blood and quickly exchanged with stable strontium; however, ^{90}Sr was retained longer than 35 days in these tissues (Reed and Nelson 1969).

Ruthenium-106 appeared in clams from North Carolina within 2 weeks after the third and fifth Chinese nuclear tests in 1965-67. Its retention was resolved into two rate functions with apparent effective half-lives of 40 days and 7 days (Wolfe and Jennings 1971). Iron-55 is a neutron-activation product produced in large quantities from ferrous materials in the immediate vicinity of a nuclear detonation. Concentration factors of ^{55}Fe and plankton in Bikini Atoll ranged from 15,000 to 25,000 (Schell 1976). Silver-110m has been detected in marine organisms after atmospheric weapon tests in the Pacific, in fishes from the Rhone River after the Chernobyl

accident, and in fishes near reactor-waste outfalls (Gamier et al. 1990). Silver-110m is depurated rapidly by brown trout (*Salmo trutta*) after high intake exposures in the water or diet (Gamier et al. 1990; Gamier and Baudin 1990). Radiotungsten is produced in quantity by certain types of nuclear devices. In one case tungsten was the most abundant radionuclide in the environment, accounting for about 90% of the total fallout activity 167 days after the detonation (Reed and Martinedes 1971). Tungsten-181 tended to concentrate in the hepatopancreas and gut of the crayfish (*Cambarus longulus longerostris*); whole body elimination consisted of two components: a rapid 1-day component and a second slower component with a biological half-time of 12.2 days (Reed and Martinedes 1971). Benthic organisms take up limited amounts of heavy metals and radionuclides from bottom sediments and recycle them to benthic and pelagic food webs. For example, polychaete worms (*Nereis diversicolor*) in contact with ^{65}Zn-contaminated sediments for 5 days lost 50% of accumulated ^{65}Zn in about 19 days on transfer to uncontaminated sediments (Renfro and Benayoun 1976).

Amphibians and Reptiles

Radiation adversely affects limb regeneration of amphibians, alters DNA metabolism, and increases the frequency of chromosomal aberrations and liver lesions (Table 25). In some species of amphibians and reptiles as in many mammals, mortality rates after acute exposure to radiation do not stabilize within 30 days--effectively invalidating the conventional LD50 (30-day postexposure) value. In the rough-skinned newt (*Taricha granulosa*), for example, the minimal LD50 dose was 2.5 Gy at 200 days after irradiation but 350 Gy at 30 days (Willis and Lappenbusch 1976). Low temperatures seem to prolong the survival of amphibians that are exposed to ionizing radiation. The survival was greater of leopard frogs (*Rana pipiens*) held at low temperatures (5-6° C) after total-body exposure to lethal doses of X-rays than of frogs held at higher temperatures. Prolonged survival at low temperatures was due to a prolongation of the latent period rather than to appreciable recovery (Patt and Swift 1948).

Table 25. Radiation effects on selected amphibians and reptiles.

Table 25. Species, dose, and other variables	Effect	Reference[a]
Leopard lizard, *Crotaphytus wislizenii*; chronic field exposure of 0.04-0.06 Gy daily (4-5 Sv yearly) for 3 to 6 years	No female reproduction in years 3 and 4. In year 5, males were sterile and females had complete regression of ovaries, undeveloped oviducal walls, and hypertrophied fat bodies. In year 6, 75% of females lacked ovaries and 25% had normal ovaries with signs of recent egg deposition; males appeared normal	1
Mud puppy, *Necturus maculosus*; 1.1 Gy, single acute exposure	LD50, 30 days postexposure	2
Salamander, *Necturus sp*.; 0.8 Gy, single acute exposure	LD50, 200 days postexposure	3
Eastern newt, *Notophthalmus viridescens*; adults, single acute exposure of 20Gy, one limb shielded; or 22 Gy, whole body, no limbs shielded	Forelimb regeneration completely suppressed when limbs to be amputated were irradiated directly. Irradiated limbs had severe and protracted inflammation, with total resorption of the affected limbs in 85% of the cases. Shielded limbs subsequently amputated had	4

Table 25. Species, dose, and other variables	Effect	Reference[a]
	delays-but not suppression-in rate of forelimb regeneration and skin graft rejection	
Frog, *Rana sp*., single acute exposure		
7.0-7.2 Gy	LD50, 730 days after exposure	3
7.8 Gy	LD50, 150 days after exposure	3
Snakes, 2 species, 3 to 4 Gy, single acute exposure	LD50, 90 days after exposure	3
Rough-skinned newt, *Taricha granulosa*; single acute exposure		
2.5 Gy	LD50, 200 days after exposure; skin lesions and depigmentation	5
>6.5 Gy	Progressive anemia over 6-week postirradiation period; reduction in erythrocyte numbers and weight of spleen	5
80 Gy	LD50, 100 days after exposure	5
350 Gy	LD50, 30 days after exposure	5
Turtles, 4 species, <8 to 15 Sv, single acute exposure	LD50, 120 days postexposure	3
Common slider, *Trachemys scripta*; inhabiting a radioactive reservoir, in Aiken, South Carolina. Radionuclide concentrations, in Bq/kg, whole body fresh weight, (FW) were 1,002 for ^{137}Cs and 550 for ^{90}Sr. For controls, these values were 2 for ^{137}Cs and 260 for ^{90}Sr	Contaminated turtles, when compared with controls, had greater variation in DNA content of red blood cell nuclei, suggesting genetic damage. The biological half-life of ^{137}Cs in soft tissues was 64 days; for ^{90}Sr in shell and bone, it was 364 days	6
Spiny tailed lizard, *Urornastix hardwickii*; single acute exposure, held for up to 14 days after irradiation		
2.25 Gy	No lesions in liver	7
4.5 Gy	No liver lesions, but swollen hepatocytes, increases in bile pigmentation, and altered cytoplasmic degranulation; normal after 14 days	7
9.0 Gy	Some liver lesions, but all livers normal 14 days postexposure	7
Side-blotched lizard, *Uta stansburiana*; 10-22 Gy, single acute exposure	LD50, 30 days postexposure	3

[a] 1,Turner et al. 1971; 2, Rose 1992; 3, Hinton and Scott 1990; 4, Sicard and Lombard 1990; 5, Willis and Lappenbusch 1976; 6, Lamb et al. 1991; 7, Gupta and Umadevi 1990.

The South African clawed frog (*Xenopus laevis*) was a suggested bioindicator of radioactive contamination because of the greater radiosensitivity of amphibians than fishes, the ease of maintaining *Xenopus* in the laboratory, and the sensitivity of the *Xenopus* liver to radioactive contamination--including ^{45}Ca, which does not accumulate in the liver (Giannetti et al. 1990). *Xenopus* oocytes exposed to X-rays showed single- and double-strand breaks in DNA and oxidative-type base lesions at a frequency between 85% and 95%. *Xenopus* oocytes repaired X-ray induced damage in plasmid DNA; however, some X-ray lesions can stimulate homologous recombination in these cells (Sweigert and Carroll 1990). Common slider turtles (*Trachemys scripta*) in a radioactive reservoir show evidence of genetic damage, and this was attributed to long-term exposure of low concentrations of long-lived radionuclides, including ^{137}Cs and ^{90}Sr (Lamb et al. 1991). Natural populations of gulf coast toads (*Bufo valliceps*) reportedly can survive genetically damaging doses of ionizing radiation without impairment of population integrity (Whicker and Schultz 1982b). Toads and many other species share a high attrition on the large numbers of young produced each generation, and this provides an agency for intensive selection. Also under this regime, recessive mutants are eliminated as they are exposed through inbreeding in future generations (Whicker and Schultz 1982b). Sterility in field collections of the leopard lizard (*Crotaphytus wislizenii*) and the western whiptail lizard (*Cnemidophorus tigris*) was reported after long-term exposure of 3 to 5 years to various doses of gamma radiation, that is, 4-5 Sv annually in *Crotaphytus* and 2.0-2.5 Sv annually in *Cnemidophorus* (Turner et al. 1971). However, a third species of lizard in the study area (side-blotched lizard, *Uta stansburiana*) reproduced normally (Turner et al. 1971).

The retention of selected isotopes by amphibians and reptiles is quite variable. For example, whole-body retention of ^{131}I after intraperitoneal injection in the rough-skinned newt showed 2 distinct loss components with biological half-lives of 2 and 210 days; the slower component accounted for 26% of the administered activity; thyroid contained 78% of the total ^{131}I and clearly accounted for the long-term component (Willis and Valett 1971). However, similar studies with ^{131}I and the leopard frog showed three distinct loss components (0.1 day; 1.4-2.9 days; 44.3-69.4 days); loss of each component was greater at 25° C than at 10° C; also, the fast component probably represented plasma clearance through urinary excretion (Willis and Valett 1971).

Birds

Among birds, as in most other tested species, there is a direct relation between dose and mortality at single high doses of ionizing radiations (Whicker and Schultz 1982b; Table 26). For any given total dose, the survival of a bird is higher if the dose is delivered at a lower rate or over a longer period of time and suggests that biological repair processes compensate for radiation-induced cellular and tissue damage over a prolonged period or at a comparatively low dose rate (Brisbin 1991). Nestling bluebirds (*Sialia sialis*) were more resistant to gamma radiation than young domestic chickens (*Gallus* sp.), and nestling greatcrested flycatchers (*Myiarchus crinitus*) were more sensitive than bluebirds (Willard 1963). Passerine nestlings are more resistant to radiation stress than adults of larger-bodied precocial species (Brisbin 1991). But the comparatively resistant passerine nestlings frequently show a disproportional disturbance in radiation-induced growth, resulting in a reduction of overall survival. For example, if feather growth is stunted, death results from the inability to escape predators because of impaired flight (Brisbin 1991).

Table 26. Radiation effects on selected birds.

Table 26. Species, dose, and other variables	Effect	Reference[a]
Green-winged teal, *Anas carolinensis*; 4.8 Gy, single acute exposure	LD50, 30 days postexposure	1
Northern shoveler, *Anas clypeata*; 8.9 Gy, single acute exposure	LD50, 30 days postexposure	1
Blue-winged teal, *Anas discors*; 7.2 Gy, single acute exposure	LD50, 30 days postexposure	1

Table 26. Species, dose, and other variables	Effect	Reference[a]
Birds		
Eggs, passerine species, single acute exposure, 5-10 Gy	LD100	2
Nestlings, various species, 1 Gy daily	Growth retardation	3
Common quail, *Coturnix coturnix*; fertilized eggs, exposed first 9 days of incubation, single acute exposure		
5 Gy	Negligible effect on survival	4
7 Gy	Mortality >50%	4
9 Gy	All dead before hatch	4
Domestic chicken, *Gallus sp*.		
Single acute exposure		
Eggs of broilers exposed to 0.05-2.1 Gy before incubation	No adverse effects on embryonic development at 1.6 Gy and lower; at 2.1 Gy, adverse effects on development, survival, and body weight of hatched chicks	5
Chicks, age 15 days		
2.1 Gy	Reversible changes in blood chemistry within 60 days; no deaths	6
6.6 Gy	Irreversible and permanent damage to red blood cells, hemoglobin, and hematocrit; all dead within 7 days	6
Dietary exposure		
Laying hens fed diet containing 400 Bq ^{137}Cs/kg ration for 4 weeks	Of total ^{137}Cs ingested, 3% was distributed in egg contents (29-33 Bq/kg egg; 2 Bq egg); 9% in muscle (171 Bq/kg FW); and 81% in excreta	7
Broiler chickens fed diets containing 400 Bq ^{137}Cs/kg ration for 40 days; some diets contained up to 5% bentonite	Feeding with bentonite reduced ^{137}Cs concentration in muscle by 32% from 155 to 105 Bq/kg FW	7
Common black-headed gull, *Larus ridibundus ridibundus*; eggs, 9.6 Gy over 20 days	LD50	3
Great crested flycatcher, *Myiarchus crinitus*; nestlings, single acute exposure >8 Gy	All dead by fledging	8
Eastern bluebird, *Sialia sialis*, single acute exposure		
Nestlings, age 2 days		
3 Gy	Reduced growth after 16 days	8
3-5 Gy	Reduced growth and shorter primary feathers at fledging	8
4-12 Gy	Developed normally and	2

Table 26. Species, dose, and other variables	Effect	Reference[a]
	fledged successfully	
5-6 Gy	LD50, nestling to fledgling	8
25 Gy	LD50, 16 days postexposure	8
30 Gy	All dead 4 days postexposure	2,8
Fertilized eggs, 6 Gy	All dead before hatch	2
European starling, *Sturnus vulgaris*; >2 Gy, single exposure	Fatal	9
Tree swallow, *Tachycineta bicolor*		
0.006 mGy/h during breeding season, equivalent to annual dose of about 50 mSv	No adverse effects on breeding performance of adults or growth performance of nestlings	10
0.9-4.5 Gy, single acute exposure, nestlings	Adverse effects on growth, survival or both	2
1.0 Gy daily, chronic	Reduced hatch, depressed growth	2
House wren, *Troglodytes aedon*; fledglings, 0.9 Gy, single acute exposure	Growth reduction	9

a 1, Hinton and Scott 1990; 2, Millard and Whicker 1990; 3, Lowe 1991; 4, Wetherbee 1966; 5, Zakaria 1991; 6, Malhotra et al. 1990; 7, Andersson et al. 1990; 8, Willard 1963; 9, Rose 1992; 10, Zach et al. 1993.

Free-living, resident bird populations in the vicinity of sites contaminated with low levels of ionizing radiations generally have negligible genotoxic effects (George et al. 1991). However, 14% of mallards (*Anas platyrhynchos*) from an abandoned South Carolina reactor cooling reservoir heavily contaminated with ^{137}Cs (mallards contained an average of 2,520 Bq ^{137}Cs/kg whole body FW) had abnormal chromosome numbers and unusual variability in the concentration of erythrocyte DNA (George et al. 1991). Contaminated waterfowl rapidly eliminate accumulated radionuclides, suggesting inconsequential long-term damage to the birds and little hazard to human consumers of waterfowl flesh (Halford et al. 1983). This conclusion was from a study of mallards that were held for 68 to 145 days on liquid radioactive waste ponds in southeastern Idaho before they were transferred to an uncontaminated environment for 51 days. The biological half-life in mallards under these conditions was 10 days at ^{131}I and ^{134}Cs, 11 days at ^{137}Cs, 22 days at ^{140}Ba, 26 days at ^{75}Se, 32 days at ^{58}Co, 67 days at ^{6O}Co and ^{65}Zn, and 86 days at ^{51}Cr. At the time of removal from the waste ponds, radionuclide concentrations were highest in gut, then in feather, liver, and muscle, in that order. After 51 days in a radionuclide-free environment, the decreasing order of radionuclide concentrations was feather, liver, muscle, and gut (Halford et al. 1983).

Zinc-65 in trace amounts is accumulated by migratory waterfowl in the Pacific Flyway of North America from ^{65}Zn discharged into the Columbia River from water-cooled reactors at Hanford, Washington (Curnow 1971). The retention of ^{65}Zn in mallards was affected by sex and season but not by the age of the duck. Biological retention of ^{65}Zn was greater in males (Tb½ of 34.7 days) than in females (29.8 days) and greater in October (38 days) than in spring (32 days). Egg production accounted for the elimination of 25% of the ^{65}Zn and feather molt of 2-8% (Curnow 1971). Retention of ^{60}Co and ^{137}Cs--but not ^{109}Cd--in the Northern bobwhite (*Colinus virginianus*) after either acute or chronic exposure to contaminated food is similar. The biological half-life in bobwhites during exposure for 21 days was 8 days at ^{109}Cd, 11 days at ^{137}Cs, and 13 days at ^{60}Co. When radioisotopes were administered during a single 4-h feeding, Tb½ values were 3 days at ^{109}Cd, 10 days at ^{137}Cs, and 15 days at ^{60}Co (Anderson et al. 1976). The biological half-life of ^{137}Cs in avian tissues is about 6.0 days in the domestic chicken (Andersson et al. 1990), 6.7 days in the blue jay (*Cyanocitta cristata*); Levy et al. 1976), 5.6 days in the American wood duck (*Aix sponsa*) and 11.7 days in the mallard (Cadwell et al. 1979). Domestic poultry seem to accumulate a higher fraction of the daily ingested ^{137}Cs/kg muscle than mammals, but levels were effectively reduced by feeding an uncontaminated ration for at least 10 days prior to slaughter

(Andersson et al. 1990).

Mammals

The mammalian sensitivity to acute and chronic exposures of ionizing radiation, ability to retain selected radionuclides, and effect of biological and abiotic variables on these parameters are briefly summarized in Table 27. These data clearly indicate a dose-dependent effect of radiation on growth, survival, organ development, mutagenicity, fatal neoplasms, kidney failure, skeletal development, behavior, and all other investigated parameters. In general, fetuses and embryos were most sensitive to ionizing radiation, and acute or chronic exposures between 0.011 and 0.022 Gy were demonstrably harmful to mice, rats, and guinea pigs.

Table 27. Radiation effects on selected mammals.

Table 27. Species, dose, and other variables	Effect	Reference[a]
Short-tailed shrew, *Blarina brevicauda*; 7.8 Gy, single acute exposure	LD50, 30 days postexposure	1
Cow, *Bos sp*.		
Oral intake of 0.89 Bq ^{129}I, whole animal	Thyroid contained 0.97 Bq ^{129}I/kg fresh weight (FW) vs. <0.0012 Bq/kg FW for all other tissues	2
Fed 6.4 Bq ^{129}I daily for 8 days	After 8 days, 22% of total dose of 51.2 Bq was in thyroid; after 63 days, thyroid contained 1 Bq/kg FW and other tissues <0.01 Bq/kg FW	3
Dog, *Canis familiaris*		
Beagle embryos age 55 days, or pups 2 days old, given single acute exposure of 0.16, 0.83, or 1.25 Gy	Dose-dependent increase in immature dysplastic glomeruli and other signs of progressive renal failure	4
Beagles, prenatal and early neonatal stages, given single acute dose of 0.2-1.0 Gy, then observed over 11-year life span	Irradiation at all ages was associated with increased risk: decreased fertility; inhibited growth and development; lower brain weight; and increase in fatal neoplasms	5
Beagle embryos or pups As above, 2.24-3.57 Gy	Reduction in total number of nephrons and progressive renal failure	4
Beagles, 17-20 months old, single intravenous injection of 200-440,000 Bq ^{226}Ra/kg body weight (BW)	Dose-dependent increase in skeletal malignancies in 36% of dogs during lifetime	6
Beagles, age 5 years, given single injection of ^{226}Ra, in Bq/kg BW, of 39,000, 116,000, or 329,000. Injected ^{226}Ra solutions also contained ^{210}Po, ^{210}Pb, and ^{210}Bi	At lowest dose of 39 kBq/kg BW, kidney was normal, death after 2,032 days. At intermediate dose, death in 1, 210 days; at high dose, death in 581 days. Tubular degeneration and necrosis of kidney at 116 and 329 kBq	7
Beagles, age 7 years, single	No kidney damage with ^{226}Ra, but	7

Table 27. Species, dose, and other variables	Effect	Reference[a]
injection of ^{226}Ra (no contaminating ^{210}Po, ^{210}Pb, or ^{210}Bi) at 45,000 Bq/kg BW, or 122,000 Bq ^{210}Po/kg BW	kidney damage with ^{210}Po	
Beagles, age 7 years given single injection of 1,629,000 Bq ^{226}Ra/kg BW, equivalent to 1.89 Gy (from ^{210}Po contaminants), or 4,831,000 Bq ^{226}Ra/kg BW = 5.15 Gy from ^{210}Po	At low dose, all dead after 516 days; at high dose, all dead in 266 days. Death was from renal failure	7
Guinea pig, *Cavia* sp.; chronically irradiated daily during 8-h exposure. Daily dose, in Gy		
0.000	Mean survival time of 1,372 days	8
0.001	50% dead in 1,457 days	8
0.011	50% dead in 1,224 days	8
0.022	50% dead in 978 days; anemia	8
0.044	50% dead in 653 days; anemia	8
0.088	50% dead in 187 days; anemia	8
Monkey, *Cebus apella*; 1 Gy, whole body, single acute exposure	Leucocyte reduction in 6 days; blood chemistry normal after 90 days	9
Chinese hamster, *Cricetus* sp.; ovary cells, single acute dose ranging between 0.005 and 0.06 Gy	Increased frequency of sister chromatid exchange at 0.005 Gy; increased numbers of chromosomal aberrations at >0.02 Gy; no significant increase in cell death	10
Syrian hamster, *Cricetus* sp.; 0.12-2 Gy, single acute exposure	Genes modifying cytoskeletal development adversely affected at all doses within 3 h by both high LET (neutrons) and low LET (gamma rays, X-rays) radiations	11
Human, *Homo sapiens*		
Developing forebrain, 0.18-0.55 Gy (estimated dose to prenatally exposed Japanese atomic bomb survivors)	Seizures in childhood; reduced school performance at least through age 11 years; some cases of severe mental retardation by age 17 years	12
Fetus, 1 Sv, 8-15 weeks of gestation	40% probability of severe mental retardation; IQ score lowered 30 points	13
Sperm chromosomes, 0.23, 0.45, 0.91, or 1.82 Gy, single acute exposure	Chromosomal aberrations increased linearly from 6.1% at 0.23 Gy to 62% at 1.82 Gy	14
Thyroid, single acute exposure 0.065 Gy	Minimum dose for induction of thyroid carcinoma	15

Table 27. Species, dose, and other variables	Effect	Reference[a]
3-5 Gy	5% increase in thyroid malignancies 20 years after exposure, with tumors appearing 4-5 years after exposure	16
7-10 Gy	Linear dose relation to thyroid cancer, and pathology in adjacent parathyroid and salivary glands	16
Whole body		
Single brief exposure		
0.05-0.11 Sv	Doubles rate of cancers	17
0.15 Sv	Temporary sterility, males	13
0.18-0.29 Sv	Doubles rate of pregnancy complications	17
0.5-2.0 Sv	Opacity of lens; depression of hematopoiesis	13
0.68-1.10 Sv	Doubles rate of F1 generation mortality	17
1 Sv, adults	1% probability of hereditary effects; 4% probability of fatal cancer in occupational workers	13
2.5-6.0 Sv	Sterility, females	13
3.5-6.0 Sv	Permanent sterility, males	13
5.0 Sv	Cataracts	13
<1 Gy	Survival almost certain	18
1-2 Gy	Survival probable	18
1-2 Gy	About 5% mortality in several months from infection and hemorrhage	19
2-5 Gy	Survival possible	18
2-7.5 Gy	Hematopoietic syndrome characterized by bone marrow damage, anemia, lowered immune response, hemorrhage, and sometimes death	20
3-5 Gy	Death in 30-60 days, bone marrow damage	13
5-10 Gy	100% adversely affected within weeks with bone marrow abnormalities; about 45% mortality	19
5-15 Gy	Death in 10-20 days; GI tract and lung damage	13
5-20 Gy	Survival improbable	18
7.5-30 Gy	Gastrointestinal damage: nausea, vomiting, anorexia, diarrhea, lethargy, weight loss, dehydration, exhaustion, and death	20
10-15 Gy	All adversely affected with intestinal problems within 30 min; 95% dead in 2 weeks from enterocolitis shock	19
>15 Gy	Death in 1-5 days, nervous	13

Table 27. Species, dose, and other variables	Effect	Reference[a]
	system damage	
>50 Gy	All dead in 48 h, usually from cerebral edema	19
Annual dose rate or protracted annual exposure for many years		
>0.1 Sv	Lens opacity	13
>0.15 Sv	Cataracts	13
>0.2 Sv	Sterility, females	13
0.4 Sv	Temporary sterility, males; hematopoiesis depression	13
2.0 Sv	Permanent sterility, males	13
Rhesus monkey, *Macaca mulatta*		
Females, single acute dose of 0.25 to 6.5 Gy, observed over a 17-year postexposure period	At doses >2 Gy, 53% developed endometriosis (abnormal uterine growth) vs. 26% in controls; irradiated monkeys weighed 43% less than controls, 35% were anorexic, 89% had abnormal uterine anatomy, and histopathology in most tissues exceed 50% frequency	21
Exposed to single brief whole body proton irradiation (protons in the energy range encountered by astronauts) ranging between 0.25 and 12 Gy and observed for 24 years until death	Dose-dependent life shortening of at least 40 months at doses >4.5 Gy; mean life shortening was 200-500 monkey days per Gy, equivalent to 500-1,250 human days. Brain cancer first observed in 8 Gy group after 13 months. Monkeys receiving between 3 and 8 Gy had a significantly higher proportion of cancer deaths than those receiving 0.25-2.8 Gy. Latent period for cancer in animals receiving 4-8 Gy ranged between 13 months and 20 years	22
Mammals		
10 species, 2.8-8.05 Gy, single acute exposure	LD50, 30 days after exposure	23
Various species, bioconcentration factors (BCF) of selected radionuclides		
^{60}Co		
Herbivores	Whole body BCF of 0.3	24
Caribou, *Rangifer tarandus*		
Bone vs. kidney	BCF of 0.5 vs. 0.4	24
Liver vs. muscle	BCF of 0.9 vs. 0.02	24
$^{134+137}Cs$		
Herbivores	Whole body BCF of 0.3-2.0	24
Omnivores	Whole body BCF of 1.2-2.0	24

Table 27. Species, dose, and other variables	Effect	Reference[a]
Carnivores	Whole body BCF of 3.8-7.0	24
^{131}I		
Herbivores	Whole body BCF of 0.05	24
Omnivores	Whole body BCF of 0.2	24
Carnivores	Whole body BCF of 0.1	24
^{90}Sr, caribou, muscle vs. bone	BCF of 0.02 vs. 7.0	24
Various species, biological half-life of selected radionuclides		
^{241}Am		
Bone	27.4 years	25
Gonads	>27.4 years	25
Kidney, liver	11 years	25
Muscle	4 years	25
Serum	5 days	25
^{137}Cs, kidney, liver, and muscle	30 to 50 days	25
$^{238+239+240}Pu$		
Bone	49 years	25
Gonads	>49 years	25
Kidney, liver	19 years	25
Muscle	5.5 years	25
Serum	5 days	25
Singing vole, *Microtus miurus*; 8.46 Gy, single acute dose	LD50, 30 days after exposure	26
Creeping vole, *Microtus oregoni*; 6.51 Gy, single acute dose	LD50, 30 days after exposure; sensitivity may be associated with low chromosome complement	26
Meadow vole, *Microtus pennsylvanicus*; single brief exposure		
7.04 (6.35-7.98) Gy, irradiated in November	LD50, 30 days after exposure; irradiated voles released into environment	27
7.67 (7.01-8.39) Gy, irradiated in May	LD50, 30 days after exposure; irradiated voles released into environment	27
8.44 (8.17-8.77) Gy	LD50, 30 days after exposure; irradiated voles held in laboratory	7
Pine vole, *Microtus pinetorum*; single brief exposure		
7 Gy	None dead 30 days after exposure; weight normal	28
8.8 Gy, males	LD50, 30 days after exposure; weight loss in survivors	28
10.0 Gy, females	LD50, 30 days postexposure; weight loss in survivors	28
House mouse, *Mus museulus*;		

Table 27. Species, dose, and other variables	Effect	Reference[a]
single brief exposure		
7.5-8.8 Gy	LD50, 30 days postexposure	26,29
7.8, 8.1, 8.3, and 9.8 Gy	LD50, 30 days after exposure; four different strains	1
Mouse, *Mus* sp.		
Intraperitoneal injection		
Single injection of 850 Bq ^{227}Ac/kg BW alone or in combination with ^{227}Th at 18,500, 74,000, or 185,000 Bq/kg BW	The highest bone cancer incidence was observed at the highest doses of ^{227}Th. The addition of ^{227}Ac resulted in an additional osteosarcoma incidence only at 18,500 Bq ^{227} Th/kg BW	30
Single injection of ^{241}Am at concentrations-in Bq/kg BW-of 0.02, 0.06, 0.19, 0.37, or 1.2	Survival time decreased from 594 days for controls and 0.02 group to 135 days in the 1.2 group; increased frequency of tumors in bone, liver and lymph	31
Adult males, 84 days old, given 2 to 64 Bq ^{224}Ra/mouse, either as single injection, or 8 injections at 3.5 day intervals over 4 weeks; observed for 24 months	No difference in single or multiple injection effects. No effect at 16 Bq and lower, At 32 and 64 Bq, reduction in bone growth and osteonecrosis of mandible ("radium jaw")	32
Oocytes given single exposures of 0.1, 0.15, or 0.25 Gy; immature oocytes examined 8-12 weeks later	Controls had 100% survival and zero chromosome aberrations; the 0.1 Gy group had 30% survival and 2% chromosome aberrations; 0.15 Gy had 17% survival and 6% chromosome aberrations; 0.25 Gy had 5% survival and 23% chromosomal aberrations	33
Whole body, single brief exposure		
1 Gy	Acute exposure may extend life span	34
1.35 Gy	Doubles mutation rate of spermatogonia	17
7.6 Gy	LD50, 30 days postexposure	35
9.5 Gy	LD100, 30 days postexposure	35
10.0 Gy, adult males, 16-20 weeks old, observed for 7 days	At 90 min after irradiation, locomotor activity was suppressed and remained depressed; at 4 days, body weight decreased; at day 7, offensive aggressive behavior	20
12.5 Gy	LD100, days 3-7 postexposure	35
0.12-2.5 Sv	Doubles rate of heritable translocations in males	17
0.25-2.5 Sv	Doubles rate of congenital malformations in females	17
0.4-1.0 Sv	Doubles frequency of dominant	17

Table 27. Species, dose, and other variables	Effect	Reference[a]
	lethal mutations	
0.5-1.0 Sv	Doubles rate of heritable translocations in females	17
0.8-2.5 Sv	Doubles rate of congenital malformations in males	17
1.5-3.0 Sv	Doubles rate of recessive lethal mutations	17
Chronic exposure, daily dose over 8-h period, in Gy		
0.0	Mean survival time of 703 days	8
0.001	Mean survival of 761 days	8
0.011	Mean survival time of 684 days; 50% weight gain over controls	8
0.022	50% dead in 630 days	8
0.044	50% dead in 591 days	8
0.088	50% dead in 488 days	8
Total yearly dose, chronic exposure, 16 Gy (about 0.044 Gy daily)	Tolerated	34
Domestic ferret, *Mustela putorius*; 2, 4, or 6 Gy; single brief exposure; adult males	All doses depressed locomotion; vomiting in 22 min at 2 Gy, 13 min at 4 Gy, and 11 min at 6 Gy. Various substituted benzamides reduced vomiting	36
Marsh rice rat, *Oryzomus palustris*; 5.25 Gy, single acute exposure	LD50, 30 days after exposure; this species was the most sensitive of 10 rodent species tested	1
Domestic sheep, *Ovis aries*		
Ewes fed hay containing 9,000 Bq ^{137}Cs/kg DW for 50-60 days, then 40 days on uncontaminated hay; some diets contained 30 or 60 g of vermiculite daily, or 2 g of ammonium ferricyanoferrate (AFCF) daily	Maximum levels of ^{137}Cs were reached in 10 days in milk and 35-40 days in muscle. Radionuclide transfer to milk and meat was reduced 2.5 times at daily intakes of 30 g vermiculite, and 8 times at 60 g vermiculite or 2 g AFCF	37
Ewes given oral dose of 74,000 Bq ^{137}Cs, observed for 76 days	At 76 days, only 26% of ^{137}Cs remained; tissue concentrations, in Bq ^{137}Cs/kg FW, were 77,000 in salivary gland; 42,000 in muscle; 24,000-36,000 in pancreas, liver, and kidney; 14,000-17,000 in spleen and lung; and <8,000 in other tissues	38
Lambs fed 21 kg of vegetation containing 16,600 Bq of	Of the Pu ingested, 46% was in liver, 30% in bone, 12% in muscle, and 2% in	39

Table 27. Species, dose, and other variables	Effect	Reference[a]
$^{238+239+240}Pu$ and 14,400 Bq of ^{241}Am over a 14-day period, followed by 4 days on uncontaminated hay	lung; for ^{241}Am, these values were 19% in liver, 15% in bone, bone, 6% in meat, and 0.5% in lungs	
Lamb given single intravenous injection of 23 Bq ^{238}Pu plus 27 Bq ^{241}Am and held for 11 days	Liver retained up to 44% of the injected ^{238}Pu and 28% of the ^{241}Am; bone had 21% of the ^{238}Pu and 20% of the ^{241}Am	39
Great Basin pocket mouse, *Perognathus parvus*; 8.56 Gy, single exposure	LD50, 30 days after exposure; hair loss within 7 days	26
White-footed mouse, *Permoyscus leucopus*; whole body, single brief exposure		
9.5 Gy, both sexes irradiated	No reproduction	40
9.5 Gy, males only irradiated	91% of pairs successful in producing young	40
9.5 Gy, females only irradiated	40% of pairs reproduced successfully	40
10.7 Gy	LD50, 30 days postexposure; this species was the most radioresistant of 10 species of rodents tested	1
Deer mouse, *Peromyscus maniculatus*; 9.19 Gy, single brief exposure	LD50, 30 days after irradiation	26
Oldfield mouse, *Peromyscus polionotus*; 11.25 Gy, single exposure	LD50, 30 days after exposure	29
Norway rat, *Rattus norvegicus*; 8.67 Gy, single exposure	LD50, 30 days postexposure	1
Laboratory white rat, *Rattus* spp.		
0.001, 0.01, or 0.1 Gy; single acute whole body exposure; adult males; killed up to 180 days after exposure	Fertility reduction was zero in the low-dose group, 25% at 0.01 Gy, and 66% at 0.1 Gy. Primary sites of damage were tubuh of the testes and spermatogonia. The high-dose group also had altered serum hormone chemistry after 30 days that persisted for 180 days	41
Pregnant rats given single brief exposure to 0.25, 0.5, 0.75, or 1 Gy on gestational day 15; fetuses examined 24 h after irradiation	No effect at 0.5 Gy and lower on cerebral mantle of developing brain; however, dose- related increase in pyknotic cells and macrophages in cortical mantle of fetus for all doses	42
Pregnant females exposed on day 15 of gestation	Rats irradiated in utero had impaired gait, slower motor	43

Table 27. Species, dose, and other variables	Effect	Reference[a]
to 0.75 Gy; fetuses examined 1-3 months after birth	behavior, difficulty in learning motor tasks, reduced growth rate, and reduced thickness of the cerebral cortex	
<1 Gy, whole body, single exposure	No brain pathology	44
2 Gy, whole body, single exposure	Brain pathology	44
Adult males conditioned to avoid electric shock by pressing a lever were subjected to various whole body acute exposures		
1.5 Gy daily for 5 consecutive days (total 7.5 Gy)	No significant change in performance over 8 weeks	45
4.5 Gy, single exposure	No effect on performance for at least 6 weeks	45
7.5 Gy, single exposure	Significantly decreased response rate over the first 4 weeks; performance normal during weeks 5-6; 9% reduction in body weight	45
9.5 Gy, single exposure	LD100, 30 days postexposure	45
Single local dose of 5 to 20 Gy to various salivary glands	Dose-dependent decrease in salivary flow rate and sodium composition of saliva; at 10 Gy and higher, changes were irreversible	46
25 Gy, single whole body exposure	50% dead 30-60 days postirradiation from fatal stomach damage; significant liver damage in survivors	47
Eastern harvest mouse, *Reithrodontomys humulis*; 9.5 Gy, single exposure	LD50, 30 days after exposure	1
Rodents, single brief exposure		
3.8-13.3 Gy, 13 species	LD50, 30 days postexposure; females more sensitive than males	48
4-8 Gy, 5 species	Dead rodents had histopathology of lymph nodes, thymus, bone marrow, liver, lung, and gonads; male survivors had atrophied testes	26
5.3-10.7 Gy, 10 species	Range of LD50 values (30 days after exposure); survivors showed conjunctivitis, ataxia, diarrhea, passiveness, cessation of feeding, aggressiveness, and graying of pelage	1
Squirrel monkey, *Saimiri* spp.; fetuses, 80-90 days postconception; given 0.1 or 1.0 Gy, single acute	No effect on behavior at 0.1 Gy. At 1 Gy, emotional stability and vision impaired at age 30 days,	44

Table 27. Species, dose, and other variables	Effect	Reference[a]
exposure; young observed from birth to age 2 years	learning impaired at 1 year; normal at 2 years	
Cotton rat, *Sigmodon hispidus*		
Females given 5, 7.5, 9, 10.5, or 12 Gy whole body once a month for 4 months, then released into a 0.4 ha-impoundment with unirradiated males and females for 15 days	Survival was 91% at 5 Gy and 25% at 12 Gy; intermediate doses had intermediate survival	49
9.58 Gy, single brief exposure	LD50, 30 days after exposure	1
11.2 Gy, single acute exposure, adult females	LD50, 30 days after exposure	49
11.3 Gy, single exposure, adult females	LD50, 15 days postexposure	49
Eastern chipmunk, *Tamias striatus*		
2-4 Gy, single exposure	Irreversible injury throughout life, but lifespan was increased	50
Given single exposure of 2 or 4 Gy, then released into environment	Irradiated chipmunks had consistently smaller home ranges and moved shorter distances than did controls	51
Iodine-131, half-time release rate from thyroid; females vs. males		
Summer	2.3 h vs. 263 h	52
Spring	156 h vs. 217 h	52
Autumn	126 h vs. 129 h	52

[a] 1, Dunaway et al. 1969; 2, Handl et al. 1990; 3, Handl and Pfau 1989; 4, Jaenke and Angleton 1990; 5, Benjamin et al. 1990; 6, Lloyd et al. 1991; 7, Bruenger et al. 1990; 8, Lorenz et al. 1954; 9, Egami et al. 1991; 10, Nagasawa et al. 1990; 11, Woloschak et al. 1990a; 12, Mole 1990; 13, ICRP 1991a; 14, Kamiguchi et al. 1990; 15,Kim et al. 1990; 16, Refetoff 1990; 17, Saknaranarayanan 1991c; 18, McLean 1973; 19, United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) 1988; 20, Maier and Landauer 1990; 21, Fanton and Golden 1991; 22, Wood 1991; 23, Hobbs and McClellan 1986; 24, Kitchings et al. 1976; 25, Gilbert et al. 1989; 26, O'Farrell 1969; 27, Iverson and Turner 1976; 28, Dunaway et al. 1971; 29, Golley and Gentry 1969; 30, Muller et al. 1990; 31, Schoeters et al. 1991; 32, Robins 1990; 33, Straume et al. 1991; 34, Rose 1992; 35, Cronkite et al. 1955; 36, King and Landauer 1990; 37, Daburon et al. 1991; 38, Vandecasteele et al. 1989; 39, Ham et al. 1989; 40, Di Gregorio et al. 1971; 41, Canfi et al. 1990; 42, Norton and Kimler 1990; 43, Norton et al. 1991; 44, Mole 1990; 45, Mele et al. 1990; 46, Vissink et al. 1990; 47, Geraci et al. 1991; 48, Whicker and Schultz 1982b; 49, Pelton and Provost 1969; 50, Thompson et al. 1990; 51, Snyder et al. 1976; 52, Kodrich and Tryon 1971.

Survival

Survival time is inversely related to dose in whole-body, acute exposures to ionizing radiation (Fig. 9). In general, hematopoietic organs are most sensitive and gastrointestinal tract and central nervous system are next most sensitive (UNSCEAR 1988). Body weight is an important modifier, and heavier mammals are usually most sensitive to radiation (Fig. 10). Feral rodent populations are at risk from ionizing radiation through the reduction in numbers from direct kill and indirectly from the radiation-caused diminution of reproduction (Di Gregorio et al. 1971).

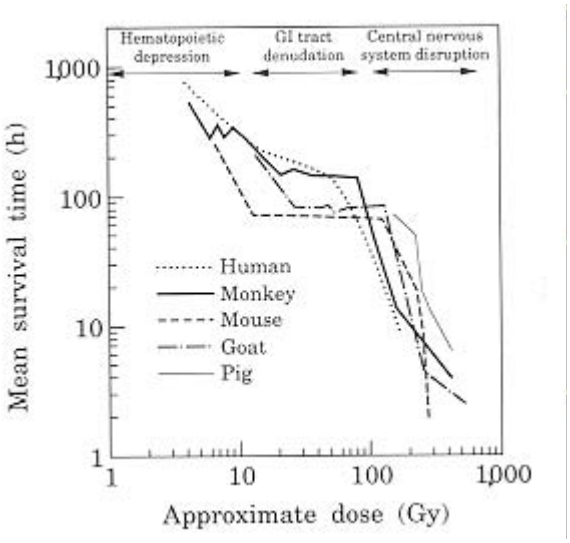

Fig. 9. Survival time and associated mode of death of selected mammals after whole-body doses of gamma radiation (modified from Hobbs and McClellan 1986; UNSCEAR 1988).

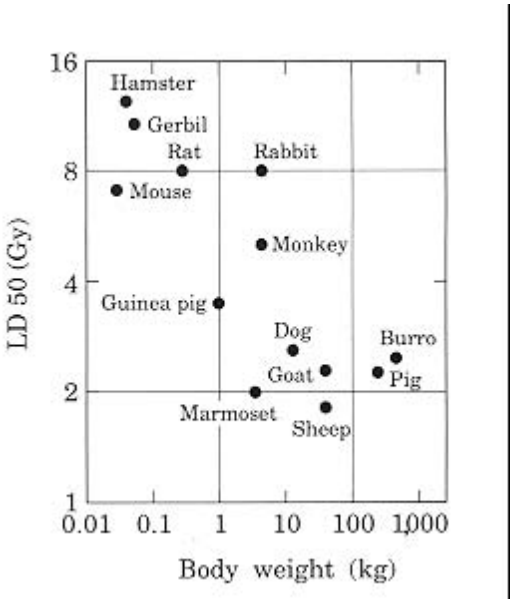

Fig. 10. Relation between body weight and radiation-induced LD50 (30 days postexposure) for selected mammals (modified from UNSCEAR 1988).

Low doses of ionizing radiation are beneficial to many species of mammals; effects of radiation hormesis include increased survival and longevity, lowered sterility, increased fecundity, and accelerated wound healing (Luckey 1980). Low doses of gamma irradiation cause irreversible injury to the eastern chipmunk (*Tamias striatus*), although the life span was significantly longer (Thompson et al. 1990). Acquired radioresistance after exposure to a low dose of ionizing radiation has been described in rats, mice, and yeast (Yonezawa et al. 1990). In mice, for example, low doses of X-irradiation (not higher than 0.15 Gy) enhanced 30-day survival if given 2-months prior to a dose of 7.5 Gy. The low-dose exposure seems to stimulate the recovery of blood-forming stem cells after the second irradiation and favors a decrease in the incidence of bonemarrow death. The exact mechanisms of radiation hormesis are unknown because effects are not related to and not predictable from the high-dose exposure (Yonezawa et al. 1990).

Irradiated small mammals that were released into the environment had a lower survival rate than laboratory populations, suggesting that the extrapolation from laboratory results may overestimate the radioresistance of free-ranging voles and other small animals because of the general level of stress in the population (Iverson and Turner 1976). The opposite was observed in eastern chipmunks after high sublethal doses of X-rays. Chipmunks had an overall reduction in mobility when they were released into the environment and a higher survival rate than controls (Snyder et al. 1976), possibly because of increased predation on the more mobile controls.

Carcinogenicity

The risk of the induction of cancer is a recognized somatic effect of low doses of ionizing radiation, as judged by epidemiological studies of Japanese survivors of the U.S. nuclear bombs (Coggle and Williams 1990) and of Marshall Islanders, underground miners, and radium watchdial workers (Bowden et al. 1990). Yoshimoto et al. (1990), however, conducted a study on the occurrence of malignant tumors in Japanese children that were older than 10 years and born between 1946 and 1982 to survivors of the atomic bombings in 1945 and in a suitable control group. They found no statistically significant increase in malignant tumors in the children of parents who had been exposed to more than 0.01 Sv whole-body radiation (mean gonadal exposure of 0.43 Sv) at the time of the atomic bombings. Nutritional status is important when treating malignant tumors. Unlike tumors of nonanemic individuals, tumors in anemic mice and humans frequently do not satisfactorily respond to radiotheraphy (McCormack et al. 1990).

Ionizing radiation induces basal-cell carcinomas in skin and is active in the initiation of malignant tumors and in the progression of benign to malignant tumors (Bowden et al. 1990). Skin has been widely used in studies of carcinogens because of its accessibility and the visibility of its tumors. All data on experimental radiogenic skin cancer in mice are on a relatively narrow and well defined response curve; however, mouse skin is about 100 times more sensitive than human skin (Coggle and Williams 1990), strongly suggesting that appropriate animal models are necessary in the extrapolation of results to another species.

Thyroidal cancer in dogs and sheep has been induced with repeated administrations of ^{131}I, although single injections of ^{131}I failed to induce thyroid cancer in adult animals except in some strains of laboratory rodents (Walinder 1990). Humans with a prior history of ^{131}I and other radiation exposure in childhood are at a significantly higher risk of thyroid carcinogenesis, and females are at higher risk than males. The minimum latent period in humans is about 4 years, and neoplastic lesions may develop as late as 40 years after irradiation (Kim et al. 1990). Historically, the human thyroid received radiation from irradiation of the scalp for epilation (as much as 0.5 Gy), thymus (as much as 5 Gy), tonsils and adenoids (8 Gy), and facial acne (15 Gy). Higher doses of external irradiation (as much as 50 Gy) were used during 1920-40 for the treatment of hyperthyroidism in adults and are still used for the treatment of cervical malignancies in people of all ages (Refetoff 1990).

Radium-induced bone malignancies after exposure to ^{226}Ra are similar in beagles and humans, and the tibia in dogs is especially sensitive (Lloyd et al. 1991). *Radium jaw* has been described in humans as a late effect of accidental ingestion or therapeutic administration of long-lived radium isotopes such as ^{216}Ra and ^{228}Ra and is characterized by bone tumors, spontaneous fractures, and osteosclerosis (Robins 1990). However, the short-lived ^{224}Ra (Tb½ of 3.6 days) produces similar effects in mice, suggesting that the events that trigger radium-induced bone disorders occur within days of incorporation, even though the consequence is a late effect (Robins 1990).

Aerosol exposures of mice, rats, dogs, and hamsters to radon and its decay products resulted in lifetime shortening, pulmonary emphysema, pulmonary fibrosis, and respiratory tract carcinoma; damages to the skin and kidney were also reported, but the lung seems to be the primary organ that is affected (Cross 1990). Radon and ^{222}Rn daughters have caused problems to miners who work underground in uranium mines. These miners had an excessive incidence of disease of the respiratory system, including lung cancer. The problem is related to the emanation of radon into the mines and the decay of the radon, the short-lived radioactive daughters (^{216}Po, ^{214}Pb, ^{214}Bi, ^{214}Po), which attach to dust particles, eventually resulting in alpha-radiation exposure of the respiratory airways (Hobbs and McClellan 1986). A similar pattern was evident in rats exposed to ^{239}Pu.

Rats exposed to $^{239}PuO_2$ aerosol of about 3,700 Bq/lungs and examined 8 to 18 months after exposure had a high frequency (as much as 80%) of malignant pulmonary neoplasms; genetic mutations were evident in 46% of the radiation-induced tumors (Stegelmeier et al. 1991).

The incidence of ovarian tumors in mice, guinea pigs, and rabbits increased after 3 years of chronic irradiation at doses as low as 1.1 mGy daily (Lorenz et al. 1954). Unlike other tumors, the induction of ovarian tumors depended on a minimum total dose and seemed to be independent of a daily dose (Lorenz et al. 1954). Radiation-induced neoplastic transformation of hamster cells may be associated initially with changes in expression of the genes modifying cytoskeletal elements (Woloschak et al. 1990b).

Mutagenicity

In general, ionizing radiation has produced mutations in every studied plant and animal species. Some genetic risks are associated with exposures, but the risk of inducing a dominant genetic disease is small because radiation-induced mutations are primarily recessive and usually lethal (Sankaranarayanan 1991c).The genetic doubling dose of radiation is the amount of acute or chronic radiation that doubles the naturally occurring spontaneous mutation rate in each generation. For mice, the estimated genetic doubling-dose equivalent is 1.35 Sv from acute exposures and 4.0 Sv from chronic exposures to radiation (Neel and Lewis 1990). For protection from radiation, the estimated genetic risks to humans have largely been based on data from mice (Straume et al. 1991). Studies of children of Japanese survivors of nuclear bomb explosions revealed that the genetic doubling dose equivalent of acute gonadal radiation is about 2.0 Sv (1.69-2.23); from chronic radiation, this value is about 4.0 Sv (Neel and Lewis 1990). Based on results of the study of Japanese survivors of the nuclear explosions, Yoshimoto et al. (1990) and Sankaranarayanan (1991c) concluded that the spontaneous mutation rate did not increase after parents were exposed. The high doubling dose of about 4 Sv estimated from these data is another way of stating that, relative to the assumed spontaneous rates, the rate of induction of mutations that leads to the measured effects is too small (Sankaranarayanan 1991c). The transmission of radiation-induced genetic effects to offspring has not yet been demonstrated in any human population (Straume et al. 1991).

Specific point mutations were identified in ^{239}Pu-induced preneoplastic lesions and in malignant neoplasms in the lungs of rats (Stegelmeier et al. 1991). Mice exposed to a single whole-body dose of 3 Gy produced a radiation-induced mutation that simultaneously generated distinct alleles of the limb deformity and agouti (grizzled fur color) loci, 2 developmentally important--but not adjoining--regions on a single chromosome. This phenomenon was probably associated with DNA breaks that were caused by inversion of a segment in another chromosome (Woychik et al. 1990). The plasma membrane in immature oocytes of mice is the hypersensitive lethal target in producing radiation-induced genetic damage (Straume et al. 1991).

Organ and Tissue Damage

In the abdomen, the kidneys are one of the most sensitive organs to serious or fatal radiation-induced damage (Jaenke and Angleton 1990). The relatively high incidence of kidney disease among mature beagles that were injected with ^{226}Ra and its accompanying ^{210}Bi and ^{210}Po were from alpha irradiation of the kidneys by the substantial amount of ^{210}Po that was in the injected solution (Bruenger et al. 1990). Hepatic injury that was induced by ionizing radiation can be a life-threatening complication. The main responses of the liver to acute radiation exposure include enlargement, dilation of blood vessels, fluid accumulation, and histopathology (Geraci et al. 1991). Damaging effects of ionizing radiation on the fetal cerebral cortex has been recognized for many years (Norton and Kimler 1990). The deleterious effects of ionizing radiation on the developing-brain are prolonged and progressive. Less than 2-Gy doses of gamma radiation are harmful to the developing brain, and mental retardation in humans may occur from doses as low as 0.2 Gy between week 8 and 15 of gestation (Norton et al. 1991).

Irradiated white-footed mice (*Peromyscus leucopus*) frequently had atrophied gonads, degenerating fetuses in the uterus, and greying hair (Di Gregorio et al. 1971). High sublethal doses (7 Gy) of radiation to the pine vole (*Microtus pinetorum*) caused pelage graying wherein unpigmented hair from damaged follicles replaces molted pigmented hair. Pelage graying may decrease survival from increased predation (Dunaway et al 1971), although this needs verification.

Human sperm chromosomes retain a high fertilizing ability after a high dose of X-irradiation, although mammalian spermatozoa have little capacity to repair DNA damage that is induced by radiation (Kamiguchi et al. 1990). Radiation-induced deaths of lymphoid cells in rats are associated with damage to the cell itself but may also be due to secretions from irradiation-activated natural killer cells, which induce pycnosis and interphase death in lymphoid cells (Eidus et al. 1990).

Behavior

Numerous behavioral measures have been evaluated for their usefulness as sensitive indexes of exposure to ionizing radiation. Radiation-related mental retardation is the most likely type of behavioral abnormality in humans; sensitivity peaked between 8 and 15 weeks of conception and doses that were greater than 0.4 Gy (UNSCEAR 1988). No specific mechanism for the production of mental retardation has been established, although proposed mechanisms include the loss of cells, migration of neurons, and failure of synaptogenesis (Norton and Kimler 1990). In studies with rats, operant responses decreased (maintained by positive reinforcement such as food or water) at sublethal radiation doses (3.0-6.75 Gy) under various schedules of reinforcement (Mele et al. 1990). Disrupted operant responses under shock avoidance at greater than LD100 levels are reported in pigs and rhesus monkeys (Mele et al. 1990).

Absorption and Assimilation

The absorption, bioavailability, and retention of radionuclides in mammals are modified by age, sex, species, and diet of the organism; season of collection; the chemical form of the radionuclide in tissue and blood; residence time in the digestive tract; preferential accumulation by selected organs and tissues; and many other variables (Kitchings et al. 1976; Whicker and Schultz 1982a, 1982b; Richmond 1989; Desmet et al. 1991; Harrison 1991). Recommended assimilation fractions of various elements by the International Commission on Radiological Protection are presented in detail by Whicker and Schultz (1982b).

Many radionuclides preferentially accumulate in certain organs or tissues, but the critical organ is different for different radionuclides: liver for ^{54}Mn, erythrocytes and spleen for ^{55}Fe, liver and kidney for cobalt nuclides, liver and prostate for ^{65}Zn, skeletal muscle for ^{137}Cs, and GI tract for ^{95}Zr (Whicker and Schultz 1982a). The persistence of radionuclides in mammals varies with the chemical form, kinetics, species, and other variables. For example, the time for 50% persistence of selected radionuclides in whole-animal studies ranged from 19 h to 14 days at ^{134}Cs; 4 to 35 days at ^{137}Cs; 5 to 12 h at the short-lived component of ^{60}Co, and 5 to 21 days at the long-lived component; 25 to 593 days at ^{90}Sr; and 4 to 26 days at ^{131}I (Kitchings et al. 1976; Whicker and Scultz 1982b).

Some radionuclides act antagonistically when administered together. The combined incorporation of ^{227}Ac and ^{227}Th at tested levels in mice showed a lower biological effect than the sums of the effects of the components administered singly. The less-than-additive effect is in good agreement with experiments with the incorporation of a mixture of β emitters, in which the effects are also less than additive (Muller et al. 1990). Uptake and retention characteristics of essential biological nutrients (i.e., H, C, P, I, K, Ca, Mn, Fe, Co, Zn) are largely controlled by biological processes (Whicker and Schultz 1982a). For example, ^{131}I regardless of route of administration is rapidly absorbed into the bloodstream and concentrated in the thyroid. Ionizing radiation from high levels of ^{131}I destroys the thyroid and affects the thyroid hormone production (Hobbs and McClellan 1986).

Alkali metals (K, Rb, Cs) behave similarly and sometimes one is accumulated preferentially when another is deficient; a similar case is made for Sr and Ca (Whicker and Schultz 1982a). The most important alkali-metal isotope is ^{137}Cs because of its long physical half-life (30 years) and its abundance as a fission product in fallout from nuclear weapons and in the inventory of a nuclear reactor or a fuel-reprocessing plant. Cesium behaves much like potassium. It is rapidly absorbed into the bloodstream and distributes throughout the active tissues of the body, especially muscle. The β and radiation from the decay of ^{137}Cs and its daughter, ^{137}Ba, result in essentially whole-body irradiation that harms bone marrow (Hobbs and McClellan 1986).

Because ^{226}Ra and ^{90}Sr are metabolic analogs of calcium, they are deposited in the skeleton; both isotopes are associated with bone cancers (Hobbs and McClellan 1986). In pregnant rats, the total amount of

^{226}Ra that is transferred from the dam to the 8-10 fetuses in a litter was low after a single injection and did not exceed 0.3% of the maternal content; the retained whole-body burden in dams was 53% at the first, 48% at the second, and 44% at the third pregnancy, mostly in the skeletal system (Kshirsagar 1990). The rare earths (i.e., ^{144}Ce, ^{152}Eu, ^{140}La, ^{147}Pr, ^{151}Sm) are usually not absorbed from the GI tract, and elimination is rapid (Palumbo 1963). Cerium-144 is one of the more biologically hazardous radionuclides in this group because of its half-life (285 days) and the energetic β emissions from it and its daughter, ^{144}Pr (Hobbs and McClellan 1986).

The greatest uncertainty in dose estimates from the ingestion of long-lived alpha emitters is the values used for their fractional absorption from the GI tract (Harrison 1991). For transuranic elements, the fraction of the ingested material that was assimilated by the whole organism was always less than 0.01% and usually nearer 0.003% (Whicker and Schultz 1982b). The major hazard of plutonium nuclides to terrestrial organisms comes from inhalation; uptake by plants is low, and further uptake by humans through the gut is low (Noshkin et al. 1971). Americium-241 is an artificial, toxic bone-seeking radionuclide produced by beta decay of ^{241}Pu (Schoeters et al. 1991). Because of its long half-life, its high-energy alpha irradiation, and its accumulation in the liver and skeleton, consideration should be given to ^{241}Am in risk estimates of latent effects such as induction of liver cancers, bone cancers, and leukemias. In comparison with ^{226}Ra, ^{241}Am is 20 times more effective in reducing the life-span in mice and 13 times more effective in the rate of death from bone cancer (Schoeters et al. 1991).

Although radon has long been known as a health hazard to miners in the uranium industry, global radon contamination of buildings was not recognized before the 1980's; however, years of exposure are required before a health problem develops (Majumdar et al. 1990). Exposure to radon-decay products can be expressed in two different ways: the amount of inhaled decay products (taking into account their potential emission of radiation energy) or the product of the time during which the decay products were inhaled and their concentration in the inhaled air. The potential alpha energy of the inhaled decay products may be expressed in joules (J). The potential alpha energy concentration in air is expressed in joules per cubic meter; for radon in equilibrium with its decay product, this corresponds to 3,700 Bq/m^3 (UNSCEAR 1988).

Rodents that were dosed with tungsten-185 excreted 80% in 24 h; bone was the major retention site; the half-time persistence ranged from 5.7 days in femurs of mice to 86 days in femurs of rats; some components in the bone of rats persisted with a half-time of longer than 3 years (Reed and Martinedes 1971). Niobium-95 is produced directly by nuclear fission and indirectly by decay of ^{95}Zr. Routine discharges of ^{95}Nb from a nuclear fuel-reprocessing plant in the United Kingdom in 1970 contributed about 5% of the bone-marrow dose to a 10-year-old child that resided in the vicinity (Harrison et al. 1990). Gastrointestinal absorption of ^{95}Nb by adult guinea pigs was about 1.1% and supports the now-used values of 1% absorption in adults and 2% in infants to calculate percentage absorption of niobium isotopes by humans (Harrison et al. 1990).

Proposed Criteria and Recommendations

For the protection from radiation, effects of radiation have been characterized as stochastic or nonstochastic. The probability of a stochastic effect--and not its severity--varies as a function of dose in the absence of a threshold, that is, hereditary effects or carcinogenesis. The probability and severity of nonstochastic effects vary with dose, and a threshold for the dose exists, that is, cataract of lens, nonmalignant damage to the skin, cell depletion in the bone marrow that causes hematological deficiencies, gonadal cell damage that leads to impairment of fertility, or pneumotis and pulmonary fibrosis after lung irradiation (ICRP 1977; Hobbs and McClellan 1986; UNSCEAR 1988). Nonstochastic effects can be prevented by setting dose-equivalent limits at sufficiently low levels so that no threshold dose is reached, not even after exposure for the whole of a lifetime or for the total period of a working life (ICRP 1977, 1991a, 1991b). Guides for the protection from radiation are also predicated on the effective half-life of each isotope, the critical organ, the fraction that reaches the critical organ by ingestion and inhalation, and the maximum tolerable whole-body burdens, as judged by radionuclide concentrations in air, water, and diet (Palumbo 1963).

At present, no radiological criteria or standards have been recommended or established for the protection of fishes, wildlife, or other natural resources. All radiological promulgated or proposed criteria are directed towards the protection of human health. It is generally assumed that humans are comparatively radiosensitive and that

guides probably also protect sensitive natural resources (UNSCEAR 1988; ICRP 1991a, 1991b; NCRP 1991; IAEA 1992; Zach et al. 1993), although this needs verification. Numerous radiological criteria exist for the protection of human health (Table 28). Most authorities agree that some adverse effects to humans are probable under the following conditions: more than 5 mSv whole-body exposure of women during the first 2 months of pregnancy; more than 50 mSv whole-body exposure in any single year or more than 2,000 mSv in a lifetime; an annual inhalation intake by a 60 kg individual--in Bq/kg BW--that exceeds 0.67 ^{232}Th, 3.3 ^{241}Am, 3.3 ^{239}Pu, 16 ^{252}Cf, 33 ^{235}U, 1,666 ^{90}Sr, 16,666 ^{60}Co, or 166,666 ^{32}P; an annual ingestion intake by a 60 kg individual--in Bq/kg BW--that exceeds 3,333 ^{129}I, 16,666 ^{125}I, 16,666 ^{131}I, or 66,666 ^{137}Cs; or a total annual intake from all sources--in Bq/kg BW by a 60 kg person--that exceeds 66 ^{210}Pb, 166 ^{210}Po, 333 ^{226}Ra, 666 ^{230}Th, 833 ^{228}Th, or 1,333 ^{238}U (Table 28).

Table 28. Recommended radiological criteria for the protection of human health.

Table 28. Criterion and other variables	Concentration or dose	Reference[a]
Air		
United States; radon-222		
Average	<0.0555 Bq (<1.5 pCi)/L or <55 Bq/m^3	1
Acceptable	<0. 148 Bq (<4.0 pCi)/L	1,2
Allowable emission discharge	<0.74 Bq (<20 pCi)/m^2/s; should not increase the radon 222 concentration in air at or above any location outside the disposal site by >0.0185 Bq (>0.5 pCi)/L or >18 Bq (>500 pCi)/m^3	3
Unacceptable	>0.185 Bq (>5 pCi)/L	2
Astronauts		
Age 25-55 years; expected whole body career dose; females vs. males	1.0-3.0 Sv (100-300 rem) vs. 1.5-4.0 Sv (150-400 rem)	4
Adverse effects expected; lifetime exposure	>2.0 Sv (>200 rem)	4
Cancer risk and birth defects		
Projected 0.04% increase in cancers; 0.01% increase in birth defects	0.11 mSv (0.011 rem) whole body maximum /year; 0.69 mSv (0.069 rem) whole body over 30 years; or 1.00 mSv (0.1 rem) bone marrow over 30 years	5
Projected 0.18% increase in cancers; 0.07% increase in birth defects	0.51 mSv (0.05 rem) whole body maximum/year; 3.30 mSv (0.33 rem) whole body over 30 years; or 4.60 mSv (0.46 rem) bone marrow over 30 years	5
Projected 0.92% increase in cancers; 0.38% increase in birth defects	3.03 mSv (0.3 rem) whole body maximum/year; 19.0 mSv (1.9 rem) whole body over 30 years; or 23.0 mSv (2.3 rem) bone marrow over 30 years	5
Projected 4.4% increase in cancers; 1.8% increase in birth defects	20.1 mSv (2.0 rem) whole body maximum/year; 91.0 mSv (9.1 rem) whole body over 30 years; or 110.0 mSv (11 rem)	5

Table 28. Criterion and other variables	Concentration or dose	Reference[a]
	bone marrow over 30 years	
Diet		
All foods; maximum recommended values		
Adults, Italy	600 Bq (16,200 pCi) cesium-134 + 137/kg fresh weight (FW)	6
Children, Italy	370 Bq (10,000 pCi) cesium-134 + 137/kg FW	6
Sweden, pre-Chernobyl	300 Bq (8,100 pCi) cesium-134 + 137/kg FW	6
Caribou; muscle; North America	<2,260 Bq (<61,000 pCi) cesium-137/kg FW	8
Fish; Great Lakes; muscle	Dose of <0.02 Sv (0.00002 rem)/kg FW fish flesh equivalent to consumers	9
Fish; Sweden	<1,500 Bq (<40,500 pCi) cesium-137/kg FW	10
Fraction of ingested dose absorbed; recommended maximum; selected isotopes		
Americium, curium, neptunium, plutonium, thorium	<0.05%	11
Americium, plutonium	<0.1%	12
Californium, and higher mass radionuclides	<0.1%	11
Uranium	<5.0%	11
Milk; maximum values		
Italy	370 Bq (10,000 pCi) cesium-134 + 137/L	6
Japan	370 Bq (10,000 pCi) cesium-137/L	13
Sweden	300 Bq (8,100 pCi) cesium-137/L	14
Meat and fish; Sweden; maximum values	1,500 Bq (40,500 pCi) cesium-137/kg FW	14
Reindeer meat, game, animal meat, fish, berries, mushrooms; Sweden; post-Chernobyl; maximum values	1,500 Bq (40,500 pCi) cesium-137/kg FW	15
Sheep, muscle	<1,000 Bq (<27,000 pCi) cesium 134 + 137/kg FW	16
Sheep, muscle	<1,000 Bq (<27,000 pCi) cesium-137/kg FW	17
Drinking water		
Natural radioactivity; maximum allowed		
Radium-226 + 228	0.185 Bq (5 pCi)/L	18
Gross alpha	0.555 Bq (15 pCi)/L	18
Artificial radioactivity; maximum		

Table 28. Criterion and other variables	Concentration or dose	Reference[a]
allowed		
Gross beta	1.85 Bq (50 pCi)/L	18
Tritium (Hydrogen-3)	740 Bq (20,000 pCi)/L	18
Strontium-90	0.296 Bq (8 pCi)/L	18
Great Lakes; maximum dose to consumers	10 mSv (0.001 rem)/year	9
General public		
Annual effective dose[b]	<1 mSv (<0.1 rem)	19,29
Cesium-137; total intake		
Sweden	<50,000 Bq (<1,350,000 pCi)/year, equivalent to <1 mSv (<0.l rem)	15
North America	<300,666 Bq (<8,100,000 pCi)/year	8
United Kingdom	<400,000 Bq (<10,800,000 pCi)/year, equivalent to 5 mSv (0.5 rem)	20
Maximum permissible dose		
Eye lens	<15 mSv (<1.5 rem)/year	19
Skin	<50 mSv (<5 rem)/year	19
Whole body		
Individual, except students and pregnant women	<5 mSv (<0.5 rem)/year	9,21,22,23
Students	<1 mSv (<0.1 rem)/year	21
Pregnant women	<5 mSv (<0.5 rem) during the first 2 months of pregnancy	22
Population dose limits, genetic or somatic	<1.7 mSv (<0.17 rem) yearly average	21
Groundwater, maximum allowed		
Radium-226 + 228	0.185 Bq (5 pCi)/L	3
Alpha-emitting radionuclides-including radium-226 + 228, but excluding radon isotopes	0.555 Bq (15 pCi)/L	3
Total beta and gamma radiation	Total annual whole body dose equivalent, or dose to any internal organ, <0.04 mSv (<0.004 rem), based on individual consumption of 2 L daily of drinking water from a groundwater source	3
Radioactive wastes		
Dose limits from spent nuclear fuel or transuranic radioactive wastes		
Whole body	<0.25 mSv (<0.025 rem)/year	3,24
Thyroid	<0.75 mSv (<0.075 rem)/year	3,24
Any other critical organ	<0.25 mSv (<0.025 rem)/year	3,24
Stored for 10,000 years; maximum cumulative release allowed to the accessible environment per 1,000 metric tons of heavy metal during storage		
Americium-241	3.7 trillion (T) Bq (100 TpCi)	3
Americium-243	3.7 TBq (100 TpCi)	3

Table 28. Criterion and other variables	Concentration or dose	Reference[a]
Any alpha emitter with physical half-life >20 years	3.7 TBq (100 TpCi)	3
Any non-alpha emitter radionuclide with physical half-life >20 years	37.0 TBq (1,000 TpCi)	3
Carbon-14	3.7 TBq (100 TpCi)	3
Cesium-135	37.0 TBq (1,000 TpCi)	3
Cesium-137	37.0 TBq (1,000 TpCi)	3
Iodine-129	3.7 TBq (100 TpCi)	3
Neptunium-237	3.7 TBq (100 TpCi)	3
Plutonium-238	3.7 TBq (100 TpCi)	3
Plutonium-239	3.7 TBq (100 TpCi)	3
Plutonium-240	3.7 TBq (100 TpCi)	3
Plutonium-242	3.7 TBq (100 TpCi)	3
Radium-226	3.7 TBq (100 TpCi)	3
Strontium-90	37.0 TBq (1,000 TpCi)	3
Thorium-230	0.37 TBq (10 TpCi)	3
Thorium-232	0.37 TBq (10 TpCi)	3
Tin-126	37.0 TBq (1,000 TpCi)	3
Uranium-233	3.7 TBq (100 TpCi)	3
Uranium-234	3.7 TBq (100 TpCi)	3
Uranium-235	3.7 TBq (100 TpCi)	3
Uranium-236	3.7 TBq (100 TpCi)	3
Uranium-238	3.7 TBq (100 TpCi)	3
Uranium byproduct materials; maximum discharge rates allowed into water		
Radium-226 + 228	0.185 Bq (5 pCi)/L	24
Gross alpha-particle activity, excluding radon and uranium isotopes	0.555 Bq (15 pCi)/L	24
Wastes from uranium fuel cycle entering the environment per billion watts/year of electrical energy produced by the fuel cycle; maximum allowed		
Krypton-85	1.85 TBq (50 TpCi)	24
Iodine-129	185 million Bq (5 billion pCi)	24
Plutonium-239 and other alpha emitting transuranics with Tb1/2 >1 year	2.69 million Bq (72.6 million pCi)	24
Occupational workers		
Annual limit of intake[b] Inhalation vs. oral		
Americium-241	200 Bq (5,400 pCi) vs. 50,000 Bq	25
Californium-252	1,000 Bq (27,000 pCi) vs. 200,000 Bq	25
Cesium-137	6 million Bq (162 million pCi) vs. 4 million Bq	
Cobalt-60	1 million Bq (27 million	25

Table 28. Criterion and other variables	Concentration or dose	Reference[a]
	pCi) vs. 7 million Bq	
Hydrogen-3	3 billion Bq (81 billion pCi) vs. 3 billion Bq	25
Iodine-125	2 million Bq (54 million pCi) vs. 1 million Bq	25
Iodine-129	300,000 Bq (8,100,000 pCi) vs. 200,000 Bq	25
Iodine-131	2 million Bq (54 million pCi) vs. 1 million Bq	25
Phosphorus-32	10 million Bq (270 million pCi) vs. 20 million Bq	25
Plutonium-239	200 Bq (5,400 pCi) vs. 200,000 Bq	25
Polonium-210	20,000 Bq (540,000 pCi) vs. 100,000 Bq	25
Radium-226	20,000 Bq (540,000 pCi) vs. 70,000 Bq	25
Strontium-90	0.1 million Bq (2.7 million pCi) vs. 1.0 million Bq	25
Thorium-232	40 Bq (1,000 pCi) vs. 30,000 Bq	25
Uranium-235	2,000 Bq (54,000 pCi) vs. 500,000 Bq	25
Total intake from all sources; Canada		
Lead-210	<4,000 Bq (<108,000 pCi)	26,27
Polonium-210	<10,000 Bq (<270,000 pCi)	26,27
Radium-226	<20,000 Bq (<540,000 pCi)	26,27
Thorium-228	<50,000 Bq (<1.35 million pCi)	26,27
Thorium-230	<40,000 Bq (<1.08 million pCi)	26,27
Thorium-232	<7,000 Bq (<189,000 pCi)	26,27
Uranium-238	<80,000 Bq (<2.1 million pCi)	26,27
Effective dose[b]		
Average annual	20 mSv (2 rem), not to exceed 50 mSv (5 rem)	28
Five-year maximum	<100 mSv (<10 rem), not to exceed 50 mSv (5 rem) in any year	28
Maximum permissible dose		
Whole body	50 mSv (5 rem) in any 1 year	21,22,29
Long-term accumulation to age N years	(N-18) x 50 mSv (5 rem)	21
Skin	150 mSv (15 rem) in any 1 year	21
Hands	750 mSv (75 rem) in any 1 year; not to exceed 250 mSv (25 rem) in 3 months	21
Forearms	300 mSv (30 rem) in any 1 year; not to exceed 100 mSv (10 rem) in 3 months	21
Skin and hands	500 mSv (50 rem) annually	19,28
Other organs	150 mSv (15 rem) in any 1 year; not to exceed 50	21

Table 28. Criterion and other variables	Concentration or dose	Reference[a]
	mSv (5 rem) in 3 months	
Pregnant women	5 mSv (0.5 rem) in gestation period	21
Eye lens	150 mSv (15 rem) annually	19,28
Soil		
Radium-226; maximum allowed	<185 Bq (<5,000 pCi)/kg over background in top 15 cm; <555 Bq (<15,000 pCi)/kg in soils at depth >15 cm.	3
Total gamma; maximum allowed	<0.2mSv (<0.00002 rem)/h over background	3

[a] 1, Gangopadhyay and Majumdar 1990; 2, Oge and Dickson 1990; 3, United States Code of Federal Regulations (CFR) 1990; 4, Wood 1991; 5, Bair et al. 1979; 6, Battiston et al. 1991; 7, Andersson et al. 1990; 8, Allaye-Chan et al. 1990; 9, Joshi 1991; 10, Hakanson and Andersson 1992; 11, Harrison 1991; 12, Gilbert et al. 1989; 13, Aii et al. 1990; 14, Johanson et al. 1989; 15, Johanson 1990; 16, Moss et al. 1989; 17, Crout et al. 1991; 18, Rose et al. 1990; 19, International Commission on Radiological Protection (ICRP) 1991a; 20, Lowe and Horrill 1991; 21, Hobbs and McClellan 1986; 22, ICRP 1977; 23, Gray et al. 1989; 24, CFR 1991; 25, Mefer 1990; 26, Clulow et al. 1991; 27, Clulow et al. 1992; 28, ICRP 1991b; 29, National Council on Radiation Protection and Measurements (NCRP) 1991.

[b] The Annual Limit of Intake (ALI) for any radionuclide is obtained by dividing the annual average effective dose limit (20 mSv) by the committed effective dose (E) resulting from the intake of 1 Bq of that radionuclide. ALI data for individual radionuclides are given in ICRP (1991b).

Astronauts between 25 and 55 years of age usually receive an average career dose of 2.0 Sv (1.0-3.0 Sv in females; 1.5-4.0 Sv in males), and this theoretically may cause a life shortening of 2,000 to 3,000 days (Wood 1991). Other environmental variables also shorten lives and include cigarette smoking (2,250 days), coal mining (1,100 days), and being 30% overweight (1,300 days); thus, models that assess the harm of a single variable such as radiation on life expectancy have to incorporate all known data and their interacting effects (Wood 1991).

Environmental dose-response models and animal epidemiological data are most frequently used to assess the risk from ionizing radiation. In its ideal form, a risk assessment should clearly present the rationale for an estimate of risk and should include the recognition of the roles of assumptions, approximations, data, theories, models, and deductions in arriving at an inference and a discussion of the involved uncertainties (Cothern et al. 1990). Current risk assessments of ionizing radiation hazards to all living organisms--not just humans--clearly require additional data and reinterpretation of existing data. Specifically, more effort is needed in the following areas: (1) measurement of concentrations of naturally occurring radionuclides and natural background doses in the environment as a baseline for studies of radiation effects (Templeton et al. 1971); (2) refinement of models of radionuclide transfer in food chains to aid in the assessment of radioactive releases from nuclear reactors and other point sources--including possible biomagnification by trophic components and turnover rates by receptor organisms (Kitchings et al. 1976); (3) continuance of protracted exposure studies to measure carcinogenesis in animal and human cell lines and the role of secondary factors--especially chemical agents--in radiation carcinogenesis (Little 1990); (4) research on radiation-induced recessive lethal mutations--the predominant type of radiation-induced mutation--and dominant mutation systems (Sankaranarayanan 1991c); (5) long-term studies to establish sensitive indicators of radiation stress on individuals and communities, including effects on growth and reproduction (Templeton et al. 1971); (6) clarification of the role of enzymes and proteins in repair of radiation-damaged cellular DNA and of mechanisms of enzymatic reactions leading to altered nucleotide sequences (Hagen 1990); (7) reinterpretation of low-level chronic irradiation effects on developing embryos under rigorously controlled conditions (Templeton et al. 1971); and (8) resolution of mathematical shape(s) of radiation dose-response curve(s; Hobbs and McClellan 1986).

Conclusions

Nuclear explosions and nuclear power production are the major sources of human radioactivity in the environment. Other sources include radionuclide use in medicine, industry, agriculture, education, and production; transport and disposal from these activities present opportunities for wastes to enter the environment. Dispersion of radioactive materials is governed by a variety of biogeochemical factors including winds, water currents, and biological vectors. Living organisms normally receive most of their external exposure to radiation from naturally occurring electromagnetic waves and their internal exposure from naturally occurring radionuclides such as potassium-40. Radiation exposure doses from natural sources of radiation are significantly modified by altitude, amount and type of radionuclides in the immediate vicinity, and route of exposure.

Radionuclide concentrations in representative field collections of biota tend to be elevated in the vicinity of nuclear fuel reprocessing, nuclear power production, and nuclear waste facilities; in locations that receive radioactive fallout from nuclear accidents and atmospheric nuclear tests; and near sites of repeated nuclear detonations. Radionuclide concentrations in field collections of living organisms vary significantly with organism age, size, sex, tissue, diet, and metabolism; season of collection; proximity to point source; and other biological, chemical, and physical variables. To date, no extinction of any animal population has been linked to high background concentrations of radioactivity.

The accident at the Chernobyl, Ukraine, nuclear reactor on 26 April 1986 contaminated much of the northern hemisphere, especially Europe, by releasing large amounts of radiocesium-137 and other radionuclides into the environment. In the immediate vicinity of Chernobyl, at least 30 people died, more than 115,000 others were evacuated, and the consumption of locally produced milk and other foods was banned because of radiocontamination. The most sensitive local ecosystems were the soil fauna and pine forest communities. Elsewhere, fallout from Chernobyl measurably contaminated freshwater, marine, and terrestrial ecosystems, including flesh and milk of domestic livestock. Reindeer (*Rangifer tarandus*) calves in Norway showed an increasing frequency of chromosomal aberrations that seemed to correlate with cesium-137 tissue concentrations; tissue concentrations in turn were related to cesium-137 in lichens, an efficient absorber of airborne particles that contain radiocesium and the main food source of reindeer during winter. A pattern similar to that of reindeer was documented in moose (*Alces alces*) in Scandinavia.

A dose and dose-rate dependent radiation effect on growth, survival, organ development, mutagenicity, fatal neoplasms, and other parameters exists for almost all organisms that were tested under laboratory conditions. Some discoveries suggest that low acute exposures of ionizing radiation may extend the life-span of certain species, although adverse genetic effects may occur under these conditions. In living organisms, the sensitivity to radiation is governed by ontogeny and phylogeny. Thus, rapidly dividing cells that are characteristic of embryos and fetuses are most radiosensitive, and evolutionarily advanced organisms such as mammals are more radiosensitive than primitive organisms. Between species in each taxonomic grouping are large variations in sensitivity to acute and chronic exposures of ionizing radiation and in ability to retain selected radionuclides; these processes are modified by many biological and abiotic variables.

Radiosensitive terrestrial plants are adversely affected at single exposures of 0.5-1.0 gray (Gy) and at chronic daily exposures of 0.2-0.65 Gy. Terrestrial insects are comparatively resistant to ionizing radiation; some species show growth stimulation and development at acute doses of 2 Gy--a demonstrably harmful dose for many species of vertebrates. Among aquatic organisms, the developing eggs and young of freshwater fishes are among the most sensitive tested organisms; death was observed at acute doses of 0.3-0.6 Gy and adverse effects on physiology and metabolism at chronic daily exposure rates of 0.01 Gy. The ability of aquatic organisms to concentrate radionuclides from the medium varies substantially with ecosystem, trophic level, radionuclide, proximity to radiation point source, and many other biological, chemical, and physical modifiers. In amphibians, radiation adversely affects limb regeneration, alters DNA metabolism, causes sterility, and increases the frequency of chromosomal aberrations.

Mortality patterns in some species of amphibians begin to stabilize about 200 days after exposure to a single acute dose of ionizing radiation and cannot be evaluated satisfactorily in the typical 30-day postexposure period. In birds, adverse effects on growth were noted at chronic daily exposures as low as 0.9-1.0 Gy and on survival and metabolism at single exposures to 2.1 Gy. Genotoxic effects were associated with whole-body loadings of 2,520 becquerels (Bq) of cesium-137/kg in mallards (*Anas platyrhynchos*). The radionuclide

retention in birds was modified by sex, season, and reproductive state. In mammals, embryos and fetuses of sensitive species were adversely affected at acute doses of 0.011-0.022 Gy. Humans exposed as fetuses to 0.18-0.55 Gy scored significantly lower on tests of intelligence.

No radiological criteria now exist for the protection of fishes, wildlife, or sensitive other natural resources. All current guides for protection from radiation target human health and are predicated on the assumption that protection of comparatively radiosensitive humans confers a high degree of protection to other life forms. Most authorities agree that significant harmful effects on humans occur under the following conditions: exposure of the whole body of women during the first 2 months of pregnancy to more than 5 millisieverts (mSv); exposure of the whole body to more than 50 mSv in any single year or to more than 2,000 mSv in a lifetime; annual inhalation intake by a 60-kg individual, in Bq/kg body weight (BW), of more than 0.7 of thorium-232, 3.3 of americium-241, 3.3 of plutonium-239, 16 of californium-252, 33 of uranium-235, 1,670 of strontium-90, 16,670 of cobalt-60, or 166,670 of phosphorus-32; annual ingestion intake by a 60-kg individual, in Bq/kg BW, of more than 3,330 of iodine-129, 16,670 of iodine-125, 16,670 of iodine-131, or 66,670 of cesium-137; or total annual intake, in Bq/kg BW, from all sources by a 60-kg person exceeds 66 of lead-210, 166 of polonium-210, 333 of radium-226, 670 of thorium-230, 830 of thorium-228, or 1,330 of uranium-238.

Current risk assessments of ionizing radiation hazards to living organisms require additional data and reinterpretation of existing data. Specifically, more effort seems needed in eight areas: (1) establishing a baseline for studies of radiation through measurement of naturally occurring radionuclides and natural background radiation doses; (2) refining radionuclide food-chain transfer models; (3) measuring the role of chemical agents in radiation-induced carcinogenesis; (4) accelerating research on radiation-induced lethal mutations; (5) initiating long-term studies to establish sensitive indicators of radiation stress on individuals and ecosystems; (6) clarifying the role of enzymes and proteins in repair of radiation-damaged cellular DNA; (7) reinterpreting embryotoxic effects of low-level chronic irradiation; and (8) resolving the mathematical shapes of radiation dose-response curves.

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Glossary[1]

Actinides Elements of atomic numbers 89 to 103 (Ac, Th, Pa, U, Np, Pu, Am, Cm, Bk, Cf, Es, Fm, Md, No, Lw).

Activity The activity of a radioactive material is the number of nuclear disintegrations per unit time. Until 1977, the accepted unit of activity was the curie (Ci), equivalent to 37 billion disintegrations/s—a number that approximated the activity of 1 g of radium-226. The present unit of activity is the becquerel (Bq), equivalent to 1 disintegration/s.

Alpha (α) particles An α particle is composed of 2 protons and 2 neutrons and has a charge of +2; essentially, it is a helium nucleus without orbital electrons. Alpha particles usually originate from the nuclear decay of radionuclides of atomic number >82 and are detected in samples that contain U, Th, or Ra. Alpha particles react strongly with matter and consequently produce large numbers of ions per unit length of their paths; as a result, they are not very penetrating and traverse only a few centimeters of air. Alpha particles are unable to penetrate clothing or the outer layer of skin; however, when internally deposited, α particles are often more damaging than most other types of radiations because comparatively large amounts of energy are transferred in a small volume of tissue. Alpha particle absorption involves ionization and orbital electron excitation. Ionization occurs whenever the particle is sufficiently close to an electron to pull it from its orbit. The α also loses kinetic energy by exciting orbital electrons with interactions that are insufficient to cause ionization.

Atom The smallest part of an element that has all the properties of that element. An atom consists of one or more protons and neutrons (in the nucleus) and of one or more electrons.

Atomic number The number of electrons outside the nucleus of a neutral (nonionized) atom plus the number of protons in the nucleus.

Becquerel (Bq) The present accepted unit of activity is the becquerel, equivalent to 1 disintegration/s. About 0.037 Bq = 1 picocurie.

Beta (β) particles Beta particles are electrons that are spontaneously ejected from the nuclei of radioactive atoms during the decay process. They may be either positively or negatively charged. A positively charged beta (β), called a *positron, is* less frequently encountered than its negative counterpart, the *negatron* (β). The *neutrino,* a small particle, accompanies beta emission. The neutrino has little mass and is electrically neutral. But neutrinos conduct a variable part of the energy of transformation and account for the variability in kinetic energies of beta particles that are emitted from a given radionuclide. Positrons (β^+) are emitted by many of the naturally and artificially produced radionuclides; they are considerably more penetrating than α particles but less penetrating than X-rays and rays. Beta particles interact with other electrons and with nuclei in the travel medium. The ultimate fate of a beta particle depends on its charge. Negatrons, after their kinetic energy is spent, combine with a positively charged ion or become free electrons. Positrons also dissipate kinetic energy through ionization and excitation; the collision of positrons and electrons causes annihilation and release of energy that is equal to the sums of their particle masses.

Breeder reactor A nuclear chain reactor in which transmutation produces a greater number of fissionable atoms than the number of consumed parent atoms.

Cosmic rays Highly penetrating radiations that originate in outer space.

Curie (Ci) The Curie is equal to that quantity of radioactive material that produces 37 billion nuclear transformations/s. One millicurie (Ci) = 0.001 Ci; 1 microcurie (Ci) = 1 millionth of a Ci; 1 picocurie (pCi) = 1 millionth of a millionth Ci = 0.037 disintegrations/s. About 27 pCi = 1 becquerel (Bq).

Decay Diminution of a radioactive substance because of nuclear emission of α or β particles or of rays.

Decay product A nuclide resulting from the radioactive disintegration of a radionuclide and found as the result of successive transformations in a radioactive series. A decay product may be either radioactive or stable.

Effective dose equivalent The weighted sum, in sieverts, of the radiation dose equivalents in the most radiosensitive organs and tissues, including gonads, active bone marrow, bone surface cells, and the lung.

Electron An electron is a negatively charged particle with a diameter of 10^{-12} cm. Every atom consists of one nucleus and one or more electrons. Cathode rays and negatrons are electrons.

Electron-volt (eV) Energy acquired by any charged particle that carries unit electronic charge when it falls through a potential difference of 1 volt. One eV = 1.602×10^{-19} joule.

Fission The splitting of an atomic nucleus into two fragments that usually releases neutrons and rays. Fission may occur spontaneously or may be induced by capture of bombarding particles. Primary fission products usually decay by β particle emission to radioactive daughter products. The chain reaction that may result in controlled burning of nuclear fuel or in an uncontrolled nuclear weapons explosion results from the release of 2 or 3 neutrons/fission. Neutrons cause additional fissile nuclei in the vicinity to fission, which produces still more neutrons that in turn produce still more fissions. The speed of the chain reaction is governed by the density and geometry of fissile nuclei and of materials that slow or capture the neutrons. In nuclear reactors, neutron-absorbing rods are inserted to various depths into the reactor core. A nuclear explosion is not physically possible in a reactor because of fuel density, geometry, and other factors.

Fusion A nuclear reaction in which smaller atomic nuclei or particles combine to form larger nuclei or particles with the release of energy from mass transformation.

Gamma ( ) rays Gamma rays have electromagnetic wave energy that is similar to but higher than the energy of X-rays. Gamma rays are highly penetrating and able to traverse several centimeters of lead. See *Photons.*

Genetically significant dose (GSD) A radiation dose that, if received by every member of the population, would produce the same total genetic injury to the population as the actual doses that are received by the various individuals.

Grey(Gy) 1 Gy = 1 Joule/kg = 100 rad.

Half-life The average time in which half the atoms in a sample of radioactive element decay.

Hertz (Hz) A measure of frequency equal to 1 cycle/s.

Indirectly ionizing particles Uncharged particles such as neutrons or protons that directly liberate ionizing particles or initiate nuclear transformations.

Ion An atomic particle, atom, or chemical radical with an either negative or positive electric charge.

Ionization The process by which neutral atoms become either positively or negatively electrically charged by the loss or by the gain of electrons.

Isomer One of two or more radionuclides with the same mass number and the same atomic number but with different energies and radioactive properties for measurable durations.

Isotope One of several radionuclides of the same element (i.e., with the same number of protons in their nuclei) with different numbers of neutrons and different energy contents. A single element may have many isotopes. Uranium, for example, may appear naturally as ^{234}U (142 neutrons), ^{235}U (143 neutrons), or ^{238}U (148 neutrons); however, each uranium isotope has 92 protons.

Joule (J) 1 J = 10^7 ergs.

Latent period Period of seeming inactivity between time of exposure of tissue to an acute radiation dose and the onset of the final stage of radiation sickness.

Linear energy transfer (LET) A function of the capacity of the radiation to produce ionization. LET is the rate at which charged particles transfer their energies to the atoms in a medium and a function of the energy and velocity of the charged particle. See *Radiation dose.*

Linear hypothesis The assumption that any radiation causes biological damage in the direct proportion of dose to effect.

Mass number The total number of neutrons and protons in the nucleus of the element, which is equal to the sum of the atomic number and the number of neutrons.

Meson Particles of mass that are intermediate between the masses of the electron and proton.

Neutrinos Neutrinos and antineutrinos are formed whenever a positron particle is created in a radioactive decay; they are highly penetrating.

Neutrons Neutrons are electrically neutral particles that consist of an electron and a proton and are not affected by the electrostatic forces of the atom's nucleus or orbital electrons. Because they have no charge, neutrons readily penetrate the atom and may cause a nuclear transformation. Neutrons are produced in the atmosphere by cosmic ray interactions and combine with nitrogen and other gases to form carbon-14, tritium and other radionuclides. A free neutron has a life time of about 19 minutes, after which it spontaneously decays to a proton, a β particle, and a neutrino. A high energy neutron that encounters biological material is apt to collide with a proton with sufficient force to dislodge the proton from the molecule. The recoil proton may then have sufficient energy to cause secondary damage through ionization and excitation of atoms and molecules along its path.

Nucleus The dense central core of the atom in which most of the mass and all of the positive charge is concentrated. The charge on the nucleus distinguishes one element from another.

Photons X-rays and gamma ( ) rays, collectively termed photons, are electromagnetic waves with shorter wavelengths than other members of the electromagnetic spectrum such as visible radiation, infrared radiation, and radiowaves. X- and photons have identical properties, behavior, and effects. Gamma rays originate from atomic nuclei, but X-rays arise from the electron shells. All photons travel at the speed of light, but energy is inversely proportional to wavelength. The energy of a photon directly influences its ability to penetrate matter. Many types of nuclear transformations are accompanied by ray emission. For example, α and β decay of many radionuclides is frequently accompanied by photons. When a parent radionuclide decays to a daughter nuclide, the nucleus of the daughter frequently contains excess energy and is unstable; stability is usually achieved through release of one or more photons, a process called isometric transition. The daughter nucleus decays from one energy state to another without a change in atomic number or weight. The most probable fate of a photon with an energy higher than the binding energy of an encountered electron is photoelectric absorption, in which the photon transfers its energy to the electron and photon existence ends. As with ionization from any process, secondary radiations that are initiated by the photoelectron produce additional excitation of orbital electrons.

Planck's constant (h) A universal constant of nature that relates the energy of a photon of radiation to the frequency of the emitting oscillator. Its numerical value is about 6.626 X 10^{-27} ergs/s.

Positron A positively charged particle of equal mass to an electron. Positrons are created either by the radioactive decay of unstable nuclei or by collision with photons.

Proton A positively charged subatomic particle with a mass of 1.67252×10^{-24} g that is slightly less than the mass of a neutron but about 1,836 times greater than the mass of an electron. Protons are identical to hydrogen nuclei; their charge and mass make them potent ionizers.

Radiation The emission and propagation of energy through space or through a material medium in the form of waves. The term also includes subatomic particles such as α, β, and cosmic rays and electromagnetic radiation.

Radiation absorbed dose (rad) Radiation-induced damage to biological tissue results from the absorption of energy in or around the tissue. The amount of energy absorbed in a given volume of tissue is related to the types and numbers of radiations and the interactions between radiations and tissue atoms and molecules. The fundamental unit of the radiation absorbed dose is the rad; 1 rad = 100 erg (absorbed)/g material. In the latest nomenclature, 100 rad = 1 grey (Gy).

Radiation dose The term radiation dose can mean several things, including absorbed dose, dose equivalent, or effective dose equivalent. The absorbed dose of radiation is the imparted energy per unit mass of the irradiated material. Until 1977, the *rad* was the unit of absorbed dose, wherein 1 rad = 0.01 joule/kg. The present unit of absorbed dose is the grey (Gy), equivalent to 1 joule/kg. Thus, 1 rad = 0.01 joule/kg = 0.01 Gy. Different types of radiation have different Relative Biological Effectiveness (RBE). The RBE of one type of radiation in relation to a reference type of radiation (usually X or) is the inverse ratio of the absorbed doses of the two radiations needed to cause the same degree of the biological effect for which the RBE is given. Regulatory agencies have recommended certain values of RBE for radiation protection and absorbed doses of various radiations are multiplied by these values to arrive at radioprotective doses. The unit of this weighted absorbed dose is the roentgen equivalent man (rem). The dose equivalent is the product of the absorbed dose and a quality factor (Q), and its unit is the rem. The quality factor is a function of the capacity to produce ionization, expressed as the linear energy transfer (LET). A Q value is assigned to each type of radiation: 1 to X-rays, rays, and β particles; 10 to fast neutrons; and 20 to α particles and heavy particles. The new unit of the effective dose equivalent is the sievert (Sv), replacing rem, where 1 Sv = 100 rem. In addition to absorbed dose and dose equivalent, there is also the exposure. Exposure is the total electrical charge of ions of one sign produced in air by electrons liberated by X or gamma rays per unit mass of irradiated air. The unit of exposure is Coulomb/kg, but the old unit, the roentgen (R) is still in use. One roentgen = 2.58×10^{-4} Coulomb/kg.

Radioactivity The process of spontaneous disintegration by a parent radionuclide, which releases one or more radiations and forms a daughter nuclide. When half the radioactivity remains, that time interval is designated the half-life (Tb ½). The Tb ½ value gives some insight into the behavior of a radionuclide and into its potential hazards.

Radionuclide An atom that is distinguished by its nucleus composition (number of protons, number of neutrons, energy content), atomic number, mass number, and atomic mass.

Relative biological effectiveness (RBE) The biological effectiveness of any type of ionizing radiation in producing a specific damage (i.e., leukemia, anemia, carcinogenicity). See *Radiation dose.*

Roentgen (R) 1 R = 2.58×10^{-4} Coulombs/kg air = production by X or rays of 1 electrostatic unit of charge/cm^3 of dry air at 0° C and 760 mm Hg = 0.87 rad in air.

Roentgen equivalent man (rem) The amount of ionizing radiation of any type that produces the same damage to humans as 1 roentgen of radiation. One rem = 1 roentgen equivalent physical (rep)/ relative biological effectiveness (RBE). In the latest nomenclature, 100 rem = 1 Sievert (Sv).

Roentgen equivalent physical (rep) One rep is equivalent to the amount of ionizing radiation of any type that results in the absorption of energy of 93 ergs/g and is approximately equal to 1 roentgen of X-radiation in soft tissue.

Shell Extranuclear electrons are arranged in orbits at various distances from the nucleus in a series of concentric spheres called shells. In order of increasing distance from the nucleus the shells are designated the K, L, M, N, O, P, and Q shells; the number of electrons that each shell can contain is limited.

Sievert (Sv) New unit of dose equivalent. One Sv = 100 rem = 1 J/kg. See *Radiation dose.*

Specific activity The ratio between activity (in number of disintegrations/m) and the mass (in grams) of material giving rise to the activity. Biological hazards of radionuclides are directly related to their specific activity and are expressed in Bq/kg mass.

Threshold hypothesis A radiation-dose-consequence hypothesis that holds that biological radiation effects occur only above some minimum dose.

Transmutation A nuclear change that produces a new element from an old one.

Transuranic elements Elements of atomic number >92. All are radioactive and produced artificially; all are members of the actinide group.

X-rays See *Photons.*

[1]Whicker and Schultz 1982a; League of Women Voters (LWV) 1985; Weast 1985; Hobbs and McClellen 1986; United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) 1988; U.S. Code of Federal Regulations (CFR) 1990.