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Radiological Benchmarks for Screening Contaminants of Potential Concern for Effects on Aquatic Biota at Oak Ridge National Laboratory, Oak Ridge, Tennessee

MANAGED BY
BECHTEL JACOBS COMPANY LLC
FOR THE UNITED STATES
DEPARTMENT OF ENERGY

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CONTENTS

TABLES

FIGURES

ABBREVIATIONS

BCF	biological concentration factor
DOE	U.S. Department of Energy
HI	hazard index
HQ	hazard quotient
IAEA	International Atomic Energy Agency
ICRP	International Commission on Radiological Protection
NCRP	National Council on Radiation Protection and Measurements

EXECUTIVE SUMMARY

A hazardous waste site may contain hundreds of contaminants; therefore, it is important to screen contaminants of potential concern for the ecological risk assessment. Often this screening is done as part of a screening assessment, the purpose of which is to evaluate the available data, identify data gaps, and screen contaminants of potential concern (Suter 1995). Screening may be accomplished by using a set of toxicological benchmarks. These benchmarks are helpful in determining whether contaminants warrant further assessment or are at a level that requires no further attention.

Blaylock, Frank, and O'Neal (1993) provide formulas and exposure factors for estimating the dose rates to representative aquatic organisms. Those formulas were used herein to calculate the water and sediment concentrations that result in a total dose rate to fish from selected radionuclides of 1 rad d^{-1}, which is the recommended acceptable dose rate to natural populations of aquatic biota (NCRP 1991). Unlike exposures to chemicals, which are expressed as the concentration in water or sediment, exposures to radionuclides are expressed as the dose rate received by the organism. Dose rates that account for the biological effects to the organism are additive. That is, the total dose rate is the sum of the normalized dose rates for each radionuclide. The screening values presented in this document include internal and external exposures from parent isotopes and all short-lived daughter products. They also include exposures from all major alpha, beta, and gamma emissions for each isotope. If the total dose rate from all radionuclides and exposure pathways exceeds a recommended threshold, further analysis is needed to determine the hazards posed by radionuclides. If, however, the total dose rate falls below an acceptable threshold, radionuclides may be eliminated from further study.

The radiological benchmarks in this report are to be used at the U.S. Department of Energy's Oak Ridge Reservation and at the Portsmouth and Paducah gaseous diffusion plants as screening values only to show the nature and extent of contamination and identify the need for additional site-specific investigation (e.g., biological and chemical testing and realistic exposure modeling). The basis for estimating acceptable dose rates to aquatic biota are presented in Blaylock, Frank, and O'Neal (1993) and not repeated herein. Hence, this document is intended to supplement, rather than replace, Blaylock, Frank, and O'Neal (1993).

The benchmark development approach presented in this document constitutes a significant expansion and improvement over that used in a previous version of this document (Jones 1997). First, sediment screening benchmarks for benthic invertebrates were replaced with sediment screening values for fish because vertebrates are more radiosensitive than invertebrates (NCRP 1991). Second, there are now two suites of benchmarks for water and sediment: those that consider exposures from only one medium and those that incorporate exposures from multiple media. And third, several radionuclides were added.

1. INTRODUCTION

A hazardous waste site may contain hundreds of contaminants; therefore, it is important to screen contaminants of potential concern for the ecological risk assessment. Often this screening is done as part of a screening assessment, the purpose of which is to evaluate the available data, identify data gaps, and screen contaminants of potential concern (Suter 1995). Screening may be accomplished by using a set of toxicological benchmarks. These benchmarks are helpful in determining whether contaminants warrant further assessment or are at a level that requires no further attention.

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The recommended acceptable dose rate to natural populations of aquatic biota is 1 rad d^{-1} (NCRP 1991). Blaylock, Frank, and O'Neal (1993) provide formulas and exposure factors for estimating the dose rates to representative aquatic organisms. Those formulas were used herein to calculate the water and sediment concentrations that result in a total dose rate of 1 rad d^{-1} to fish for selected radionuclides. These radiological benchmarks are intended for use at the U.S. Department of Energy's (DOE's) Oak Ridge Reservation and at the Portsmouth and Paducah gaseous diffusion plants as screening values only to show the nature and extent of contamination and identify the need for additional site-specific investigation (e.g., biological surveys and realistic exposure modeling). The bases for estimating acceptable dose rates to aquatic biota are discussed in Blaylock, Frank, and O'Neal (1993) and are not repeated herein. Hence, this document is intended to supplement, rather than replace, Blaylock, Frank, and O'Neal (1993).

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It should be noted that the DOE Office of Environmental Policy and Assistance is currently developing a screening methodology and associated guidance that could be used in demonstrating compliance with DOE and internationally recommended dose limits for aquatic and terrestrial biota. The methodology is being designed to be prudently conservative to allow for DOE-wide application The methodology will be extensively reviewed within DOE and by other federal (e.g., Nuclear Regulatory Commission and U.S. Environmental Protection Agency) and international (e.g., International Atomic Energy Agency) agencies, consistent with the DOE technical standards review process. It is not yet known how the final national screening values and associated guidance will relate to site-specific screening values already developed or under development. However, the approach taken herein is conceptually consistent with the methodology currently being developed at the national level.

2. BENCHMARK DEVELOPMENT APPROACH AND ASSUMPTIONS

The methodology for estimating radiation dose rates presented in Blaylock, Frank, and O'Neal (1993) is the "Point Source Dose Distribution" approach (IAEA 1976, 1979). This approach uses empirically derived dose rate formulas for selected size categories of organisms, which are represented by ellipsoid geometries of the dimensions presented in Table 1. These geometries are used to estimate the fraction of the energy emitted from a radionuclide that is absorbed by the organism. That is, gamma rays and beta particles with sufficiently high energies can pass all the way through an organism, and large organisms will absorb more of the emitted energy than will small organisms.

Table 1. Dimensions of organisms representing different size categories used in the Point Source Dose Distribution methodology for estimating radiation doses

Organism	Mass (kg)	Length of the major axes of the ellipsoid (cm)
Small fish	2.0×10^{-3}	$3.1 \times 1.6 \times 0.78$
Large fish	1.0	$45 \times 8.7 \times 4.9$

Exposures to radionuclides are expressed as the total dose rate received by the organism. The dose rate from an individual radionuclide is the sum of the internal and external dose rates, which are a function of exposure to the radionuclide and the characteristics of the radiation. The internal dose rate is based on the concentration of the radionuclide in the organism. The external dose rate is based on the concentration of the radionuclide in the surrounding water or sediment.

2.1 ORIGINAL FORMULAS

Blaylock, Frank, and O'Neal (1993) gave the dose rate (μGy h^{-1}) from internal contamination as the following:

$$D = 5.76 \times 10^{-4} \, E \, n \, \Phi \, C_o \, , \tag{1}$$

where

5.76×10^{-4} = the conversion factor from MeV dis^{-1} to μGy h^{-1},
E = the average emitted energy for alpha, beta, or gamma radiations (MeV dis^{-1}),
n = the proportion of transitions producing an emission of energy E,
Φ = the fraction of the emitted energy absorbed by the organism, and
C_o = the concentration of the radionuclide in the organism (Bq kg^{-1} wet weight).

The external dose rate (μGy h^{-1}) from radionuclides in water was given as follows:

$$D = 5.76 \times 10^{-4} \, E \, n \, (1 - \Phi) \, C_w \, , \tag{2}$$

where C_w is the concentration of the radionuclide in water (Bq L^{-1}).

The external dose rate (μGy h^{-1}) to organisms at the sediment-water interface from radionuclides in sediment was given as follows:

$$D = 2.88 \times 10^{-4} \; E \; n \; (1 - \Phi) \; C_s \; , \tag{3}$$

where C_s is the concentration of the radionuclide in sediment (Bq kg^{-1} wet weight) and 2.88×10^{-4} is one half of the MeV dis^{-1} to μGy h^{-1} conversion factor used for organisms immersed in contaminated media.

2.2 CONVERTED FORMULAS

The formulas in Blaylock, Frank, and O'Neal (1993) properly used Standard International units (i.e., becquerels and grays). Those units were converted to curies and rads for convenience; these are the units typically used on the Oak Ridge Reservation for reporting radionuclide activities and evaluating exposures. Specifically, the converted dose rates (Rad d^{-1}) from an individual radioactive isotope in the organism ($D_{internal}$), in the water ($D_{external, w}$), and in the surface sediment ($D_{external, s}$) are given by:

$$D_{internal} = 5.11 \times 10^{-8} \; E \; n \; \Phi \; C_o \; , \tag{4}$$

$$D_{external, w} = 5.11 \times 10^{-8} \; E \; n \; (1 - \Phi) \; C_w \; , \tag{5}$$

$$D_{external, s} = 1.92 \times 10^{-5} \; E \; n \; (1 - \Phi) \; C_s \; , \tag{6}$$

where

C_o = the concentration of the radionuclide in the organism (pCi kg^{-1} wet weight),
C_w = the concentration of the radionuclide in water (pCi L^{-1}), and
C_s = the concentration of the radionuclide in sediment (pCi g^{-1} dry weight).

The formulas were derived using 0.01 Gy per rad and 2.703×10^{-11} Ci per Bq as the unit conversion factors. The sediment conversion factor of 1.92×10^{-5} includes the default wet weight-dry weight conversion factor of 0.75 presented in NCRP (1991).

It is important to note that these formulas are the same for each type of radiation (i.e., alpha, beta, and gamma), but the dose from each must be calculated separately. That is, the emission energy (E) is specific to the isotope and type of radiation. For any given isotope, the total dose rate from each pathway is the sum of the dose rates from each type of radiation. For example:

$$D_{internal, total} = D_{internal, alpha} + D_{internal, beta} + D_{internal, gamma} \tag{7}$$

Then, for each isotope, the total dose rate (D_{Total}) is the sum of the total internal dose ($D_{internal, total}$), the total external dose from water ($D_{external, w, total}$), and the total external dose from surface sediment ($D_{external, s, total}$).

2.3 PARAMETER VALUES AND EXPOSURE ASSUMPTIONS

2.3.1 Adsorbed Dose

The absorbed dose is a function of the emission energy (E) and the absorbed fraction (Φ) of the radiation. Emission energies (E) of selected radionuclides are presented in Table 2. As in Blaylock, Frank, and O'Neal (1993), the average energies were obtained from the International Commission on Radiological Protection Report No. 38 (ICRP 1983) and can be used in place of E and n in Eqs. (4)–(6). The average energies include all radiations that contribute at least 0.1% of the energy per transformation (ICRP 1983). Average beta energies include beta particles, conversion electrons, and Auger radiations. Average gamma energies include x-rays, gamma-rays, and photon radiations. Average alpha energies include alpha particles and recoil nuclei. The maximum beta energies presented in Table 2 are from *The Health Physics and Radiological Health Handbook* and the 1986 supplement thereof (Shleien and Terpilak 1984, 1986), except as noted.

The absorbed fraction (Φ) of a beta or gamma radiation is a function of the energy of that radiation and the size of the target organism. Figures 1–3 present empirically derived absorbed fractions as a function of beta and gamma emission energies for the organism sizes used herein. Table 2 presents the estimated absorbed fractions for small and large fish based on these figures. Figure 3 was derived for maximum beta energies, which were not found for all of the beta-emitters listed in Table 2. In the absence of maximum beta energies, the absorbed fractions were based on average beta energies. This will tend to overestimate the internal dose rate and under estimate the external dose rate. Essentially all of the energy from alpha particles emitted from internal contamination is absorbed within organisms of the sizes listed in Table 1. It is therefore assumed that the absorbed fraction (Φ) is 1 for all alpha emissions and that external alpha radiation from water and sediment is insignificant for these organisms. This also is true for beta emissions into large fish.

The absorbed dose, as calculated in Eqs. (4)–(6), does not account for the relative biological effectiveness of the different types of radiation. A quality factor is normally used to account for the relative effects of the different radiation types (NCRP 1987; Blaylock, Frank, and O'Neal 1993). The standard quality factors for exposure of humans are 1 for gamma and beta radiations and 20 for alpha radiations. However, those factors account for the potential to cause cancer, which is not an endpoint of concern for natural populations of aquatic biota. Similar values might be expected for aquatic organisms because the soft tissue composition of non-human vertebrates is generally similar to humans in water content and basic cell structure (NCRP 1991). In the absence of standard quality factors for non-human biota, the default values for humans were used herein, as recommended by Blaylock, Frank, and O'Neal (1993). That is, the absorbed dose from alpha emissions was multiplied by 20 so that the total dose rate is normalized for the biological effectiveness of the absorbed dose rates of each type of radiation.

2.3.2 Uptake

Radionuclide uptake from water was estimated using the biological concentration factors (BCFs) for freshwater fish presented in IAEA Report No. 364 (IAEA 1994), except as noted (Table 3). The BCF is the ratio of the radionuclide concentration in the organism and the radionuclide concentration in water. Therefore, the internal concentration (C_o) equals the concentration in water times the BCF. The available BCFs were based on analyses of fish muscle, rather than whole fish. These uptake factors will under estimate C_o for radionuclides that are preferentially sequestered in other tissues (e.g., bone, liver, and the gastrointestinal tract). However, the assumption of uniform contamination is unlikely to grossly underestimate the actual dose to the tissues of concern (i.e., reproductive organs), given two conditions. First, the isotope of concern is

Table 2. Emission energies (E) and absorbed fractions (Φ) for selected radionuclides[a]

		Emission Energies (MeV)				Absorbed Fractions[c]		
						Beta	Gamma	
Radionuclide (yield)	Half-life	Average Alpha	Maximum Beta[b]	Average Beta	Average Gamma	Small Fish	Small Fish	Large Fish
Antimony-125	2.77y		6.12e-01	9.93e-02	4.30e-01	1	0.012	0.09
Barium-140	12.74d		1.01e+00	3.11e-01	1.82e-01	1	0.01	0.1
Lanthanum-140	40.27h		2.20e+00	5.33e-01	2.31e+00	0.91	0.0095	0.07
Cerium-141	32.501d		5.80e-01	1.70e-01	7.61e-02	1	0.01	0.06
Cerium-144	284.3d		3.18e-01	9.10e-02	2.07e-02	1	0.17	0.6
Praseodymium-144m (98.22% of Ce-144)	7.2m			4.72e-02	1.27e-02	1	0.4	0.81
Praseodymium-144 (1.78% of Ce-144)	17.28m		3.00e+00	1.21e+00	3.18e-02	0.81	0.053	0.35
Cesium134	2.062y		6.58e-01	1.63e-01	1.55e+00	1	0.01	0.08
Cesium-137	30y		1.17e+00	1.87e-01		0.99		
Barium-137m (94.6% of Cs-137)	2.55m			6.51e-02	5.96e-01	1	0.012	0.12
Chromium-51	27.704d			3.86e-03	3.26e-02	1	0.04	0.33
Cobalt-60	5.271y		3.18e-01	9.65e-02	2.50e+00	1	0.0095	0.07
Europium-154	8.8y		1.85e+00	2.88e-01	1.22e+00	0.95	0.01	0.1
Europium-155	4.96y		2.47e-01	6.26e-02	6.05e-02	1	0.013	0.09
Hydrogen-3	12.35y		1.86e-02	5.68e-03		1		
Iodine-131	8.04d		8.07e-01	1.90e-01	3.80e-01	1	0.012	0.09
Xenon-131m (1.11% of I-131)	11.9d			1.44e-01	2.00e-02	1	0.17	0.62
Niobium-95	35.15d		1.60e-01	4.44e-02	7.66e-01	1	0.012	0.12
Phosphorous-32	14.29d		1.71e+00	6.95e-01		0.96		
Potassium-40	1.28e+09y		1.32e+00	5.23e-01	1.56e-01	0.98	0.01	0.1
Ruthenium-103	39.28d		7.10e-01	7.45e-02	4.68e-01	1	0.012	0.1
Rhodium-103m (99.7% of Ru-103)	56.12m			3.80e-02	1.75e-03	1	0.7	0.94
Ruthenium-106	368.2d		3.90e-02	1.00e-02		1		

Table 2 (continued)

Radionuclide (yield)	Half-life	Emission Energies (MeV)				Absorbed Fractions[c]		
						Beta	Gamma	
		Average Alpha	Maximum Beta[b]	Average Beta	Average Gamma	Small Fish	Small Fish	Large Fish
Rhodium-106	29.9s		3.54e+00	1.41e+00	2.01e-01	0.71	0.012	0.1
Sodium-24	15h		1.39e+00	5.53e-01	4.12e+00	0.98	0.005	0.08
Strontium-90	29.12y		5.46e-01	1.96e-01		1		
Yttrium-90	64h		2.28e+00	9.35e-01	1.69e-06	0.9	1	1
Techetium-99	213000y		2.95e-01	1.01e-01		1		
Uranium-237	6.75d		2.48e-01	1.94e-01	1.42e-01	1	0.01	0.11
Zinc-65	243.9d		3.30e-01	6.87e-03	5.84e-01	1	0.012	0.12
Zirconium-95	63.98d		1.23e+00	1.16e-01	7.39e-01	0.99	0.012	0.12
Plutonium-239	24065y	5.23e+00		6.65e-03	7.96e-04	1	1	1
Plutonium-240	6537y	5.24e+00		1.06e-02	1.73e-03	1	0.7	0.94
Thorium-232	1.405e+10y	4.07e+00		1.25e-02	1.33e-03	1	0.7	0.94
Radium-228	5.75y		5.50e-02	1.69e-02	4.14e-09	1	1	1
Actinium-228	6.13h		2.08e+00	4.60e-01	9.30e-01	0.93	0.012	0.11
Thorium-228	1.9131y	5.49e+00		2.05e-02	3.30e-03	1	0.7	0.94
Radium-224	3.66d	5.78e+00		2.21e-03	9.89e-03	1	0.7	0.94
Radon-220	55.6s	6.40e+00		8.91e-06	3.85e-04	1	1	1
Polonium-216	0.15s	6.91e+00		1.61e-07	1.69e-05	1	1	1
Lead-212	10.64h		5.86e-01	1.75e-01	1.48e-01	1	0.01	0.1
Bismuth-212	60.55m	2.22e+00	2.26e+00	4.69e-01	1.85e-01	0.91	0.01	0.1
Polonium-212 (64.07% of Bi-212)	0.305us	8.95e+00						
Thallium-208 (35.93% of Bi-212)	3.07m		2.38e+00	5.91e-01	3.36e+00	1	0.0085	0.08
Americium-241	432.2y	5.57e+00		5.19e-02	3.24e-02	1	0.04	0.34
Neptunium-237	2.14e+06y	4.84e+00		6.85e-02	3.43e-02	1	0.034	0.3
Protactinium-233	27d		5.68e-01	1.95e-01	2.03e-01	1	0.012	0.1

Table 2 (continued)

		Emission Energies (MeV)				Absorbed Fractions[c]		
						Beta	Gamma	
Radionuclide (yield)	**Half-life**	**Average Alpha**	**Maximum Beta[b]**	**Average Beta**	**Average Gamma**	**Small Fish**	**Small Fish**	**Large Fish**
Uranium-233	158500y	4.89e+00		6.08e-03	1.31e-03	1	0.7	0.94
Thorium-229	7340y	4.95e+00		1.14e-01	9.54e-02	1	0.01	0.08
Radium-225	14.8d		3.20e-01	1.07e-01	1.37e-02	1	0.32	0.78
Actinium-225	10d	5.86e+00		2.17e-02	1.79e-02	1	0.2	0.67
Francium-221	4.8m	6.41e+00		9.81e-03	3.10e-02	1	0.053	0.37
Astatine-217	0.0323s	7.19e+00		3.66e-05	3.08e-04	1	1	1
Bismuth-213	45.65m	1.29e-01	1.42e+00	4.40e-01	1.33e-01	0.98	0.01	0.11
Polonium-213 (97.84% of Bi-213)	4.2us	8.54e+00						
Lead-209 (2.16% of Bi-213)	3.253h		6.37e-01	1.98e-01		1		
Uranium-238	4.468e+09y	4.26e+00		1.00e-02	1.36e-03	1	0.7	0.94
Thorium-234	24.1d		1.93e-01	5.92e-02	9.34e-03	1	0.7	0.94
Protactinium-234m	1.17m		1.50e+00	8.20e-01	1.13e-02	0.98	0.4	0.88
Protactinium-234	6.7h		1.40e+00	4.22e-01	1.75e+00	0.98	0.01	0.08
Uranium-234	2.445e+05y	4.84e+00		1.32e-02	1.73e-03	1	0.7	0.94
Thorium-230	7.7e+4y	4.74e+00		1.46e-02	1.55e-03	1	0.7	0.94
Radium-226	1600y	4.86e+00		3.59e-03	6.74e-03	1	0.7	0.94
Radon-222	3.8235d	5.59e+00		1.09e-05	3.98e-04	1	1	1
Polonium-218	3.05m	6.11e+00		1.42e-05	9.12e-06	1	1	1
Lead-214 (99.98% of Po-218)	26.8m		9.80e-01	2.91e-01	2.48e-01	1	0.012	0.09
Astatine-218 (0.02% of Po-218)	2s	6.82e+00		4.00e-02	6.72e-03	1	0.7	0.94
Bismuth-214 (100% of Pb-214 and At-218)	19.9m		3.27e+00	6.48e-01	1.46e+00	0.76	0.01	0.08
Polonium-214	164.3us	7.83e+00		8.19e-07	8.33e-05	1	1	1
Lead-210	22.3y		6.30e-02	3.80e-02	4.81e-03	1	0.7	0.94
Bismuth-210	5.012d		1.16e+00	3.89e-01		0.99		

Table 2 (continued)

Radionuclide (yield)	Half-life	Emission Energies (MeV)				Absorbed Fractions[c]		
						Beta	Gamma	
		Average Alpha	Maximum Beta[b]	Average Beta	Average Gamma	Small Fish	Small Fish	Large Fish
Polonium-210	138.38d	5.40e+00		8.18e-08	8.50e-06	1	1	1
Uranium-235	7.038e+08y	4.47e+00		4.80e-02	1.54e-01	1	0.01	0.1
Thorium-231	25.52h		3.05e-01	1.63e-01	2.55e-02	1	0.08	0.45
Protactinium-231	3.276e+04y	5.04e+00		6.28e-02	4.76e-02	1	0.018	0.18
Actinium-227	21.773y	6.91e-02	4.30e-02	1.56e-02	2.31e-04	1	1	1
Thorium-227 (98.62% of At-227)	18.718d	5.95e+00		4.57e-02	1.06e-01	1	0.01	0.1
Francium-223 (1.38% of At-227)	21.8m		1.15e+00	3.91e-01	5.88e-02	0.99	0.013	0.1
Radium-223 (100% of Th-227 and Fr-223)	11.434d	5.75e+00		7.46e-02	1.33e-01	1	0.01	0.11
Radon-219	3.96s	6.88e+00		6.30e-03	5.58e-02	1	0.014	0.11
Polonium-215	0.178e-02s	7.52e+00		6.30e-06	1.76e-04	1	1	1
Lead-211	36.1m		1.39e+00	4.54e-01	5.03e-02	0.98	0.017	0.16
Bismuth-211	2.14m	6.68e+00	6.00e-01	9.78e-03	4.66e-02	1	0.019	0.19
Thallium-207 (99.72% of Bi-211)	4.77m		1.44e+00	4.93e-01	2.21e-03	0.98	0.7	0.94
Polonium-211 (0.28% of Bi-211)	0.516s	7.59e+00		1.69e-04	7.79e-03	1	0.7	0.94
Curium-244	18.11y	5.89e+00		8.59e-03	1.70e-03	1	0.7	0.94
Plutonium-238	87.74y	5.58e+00		1.06e-02	1.81e-03	1	0.7	0.94

[a]Selected isotopes are those presented in Blaylock, Frank, and O'Neal (1993) plus several minor daughter products and Cm-244 and Pu-238. Indented radionuclides are the daughter products of the preceding long-lived radionuclide, as presented in Blaylock, Frank, and O'Neal (1993). Yields, half-lives, and average energies are from ICRP (1983).

[b]Maximum beta energies presented are from *The Health Physics and Radiological Health Handbook* and the 1986 supplement thereof (Shleien and Terpilak 1984, 1986). The exception is actinium-228, which is from Kocher (1981).

[c]The estimated absorbed fractions for small and large fish are based on Figs. 1–3. The absorbed fraction for all alpha emissions and for beta emissions into large fish are assumed to be 1. Absorbed fractions of beta energies are based on the maximum beta energies for each radionuclide except where maximum energies were unavailable, when they were based on average beta energies.

Table 3. Distribution co-efficients (Kds) and biological concentration factors (BCFs) for selected radionuclides[a]

	Kd (L kg^{-1})		BCF (L kg^{-1})	
Radionuclide	**Original[b]**	**Corrected[c]**	**Original[d]**	**Corrected[e]**
Actinium	450[f]	337.5	15[g]	75
Americium	5000	3750	30	150
Antimony	45[f]	33.75	100	500
Barium	60[h]	45	4	20
Cerium	10000	7500	30	150
Cesium	1000	750	2000	10000
Chromium	30[f]	22.5	200	1000
Cobalt	5000	3750	300	1500
Curium	5000	3750	30	150
Europium	500	375	50	250
Hydrogen			1	1
Iodine	10	7.5	40	200
Lead	270[f]	202.5	300	1500
Neptunium	10	7.5	30	150
Niobium	160[f]	120	300	1500
Phosphorous	9	6.75	50000	250000
Plutonium	100000	75000	30	150
Polonium	150[f]	112.5	50	250
Potassium	5.5[h]	4.125	10000[g]	50000
Protactinium	540[h]	405	10	50
Radium	500	375	50	250
Ruthenium	55[f]	41.25	10	50
Sodium	100[h]	75	20	100
Strontium	1000	750	60	300
Techetium	5	3.75	20	100
Thorium	10000	7500	100	500
Uranium	50	37.5	10	50
Zinc	500	375	1000	5000
Zirconium	1000	750	300	1500

[a]Includes all radionuclides presented in Blaylock, Frank, and O'Neal (1993) that were not minor progeny plus ^{244}Cm and ^{238}Pu. Kds and BCFs are the same for different isotopes of the same element.
[b]Kds are the expected values for freshwater sediment Kd from IAEA (1994), except as noted.
[c]Kds were multiplied by a default wet weight-dry weight conversion factor of 0.75, which assumes that the fresh weight of sediment is 25% water (NCRP 1991).
[d]BCFs are the expected values for freshwater fish muscle from IAEA (1994), except as noted.
[e]BCFs were divided by a default wet weight-dry weight conversion factor of 0.2, which assumes that the fresh weight of fish is 80% water (NCRP 1991).
[f]The lowest expected soil-water Kd from IAEA (1994).
[g]BCF is from the CRITR2 code (Baker and Soldat 1992).
[h]The default soil Kd from Baes et al. (1984).

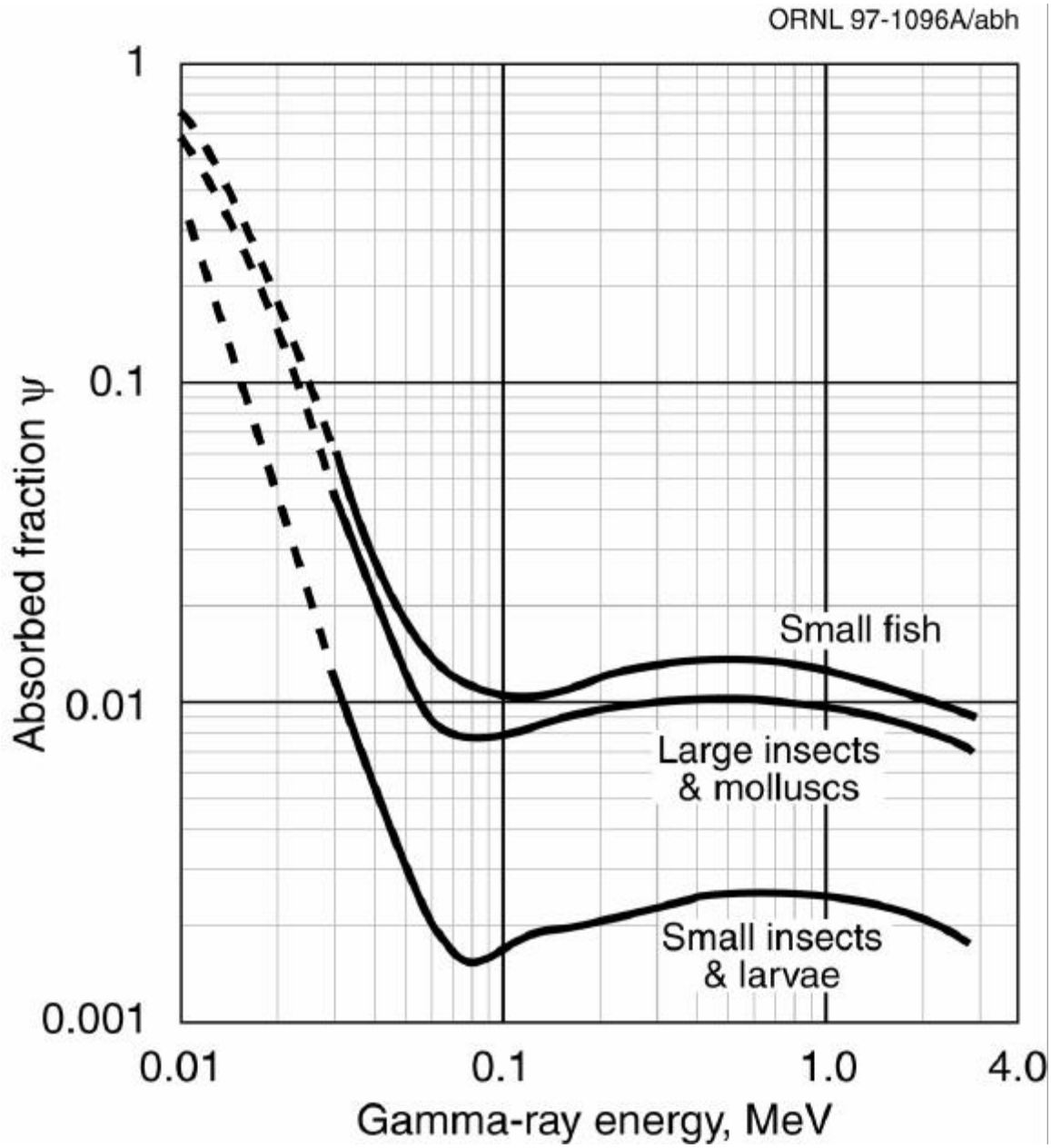

Fig. 1. Derived absorbed fractions as a function of gamma energy for small fish, large insects and molluscs, and small insects and larvae. Source: NCRP (1991). Figures were enhanced by adding grid lines.

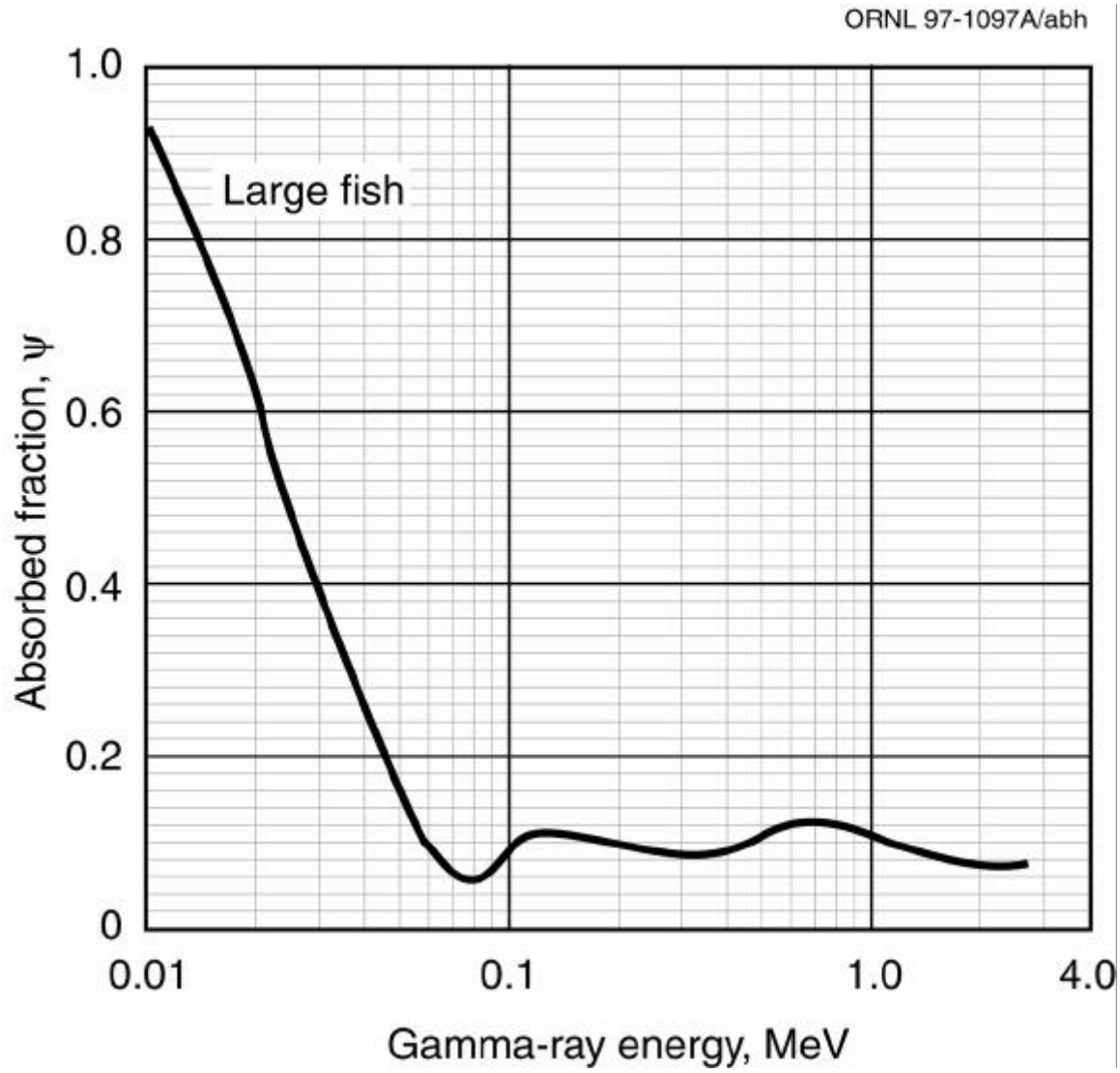

Fig. 2. Derived absorbed fractions as a function of gamma energy for large fish. Source: NCRP (1991). Figures were enhanced by adding grid lines.

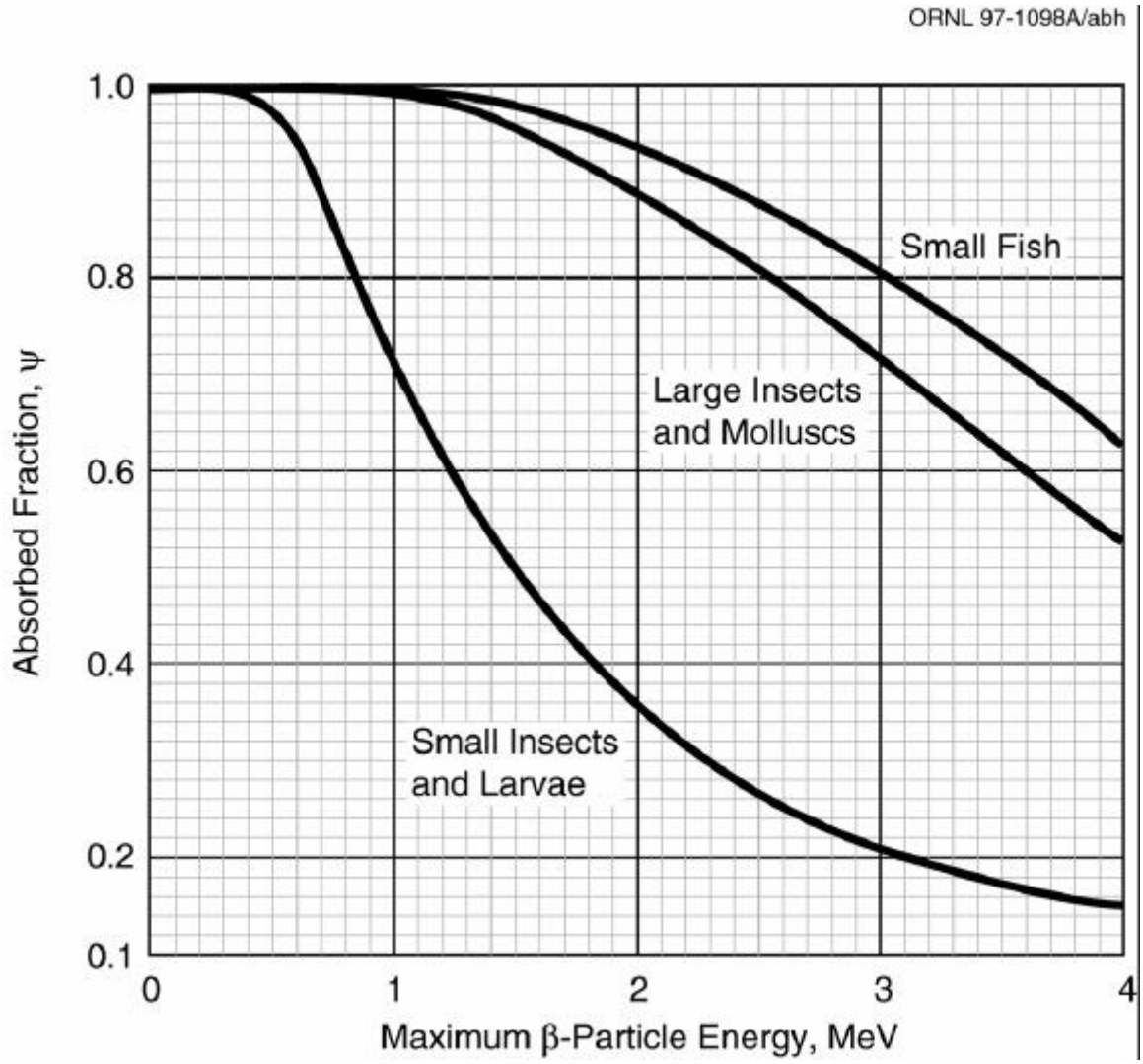

Fig. 3. Derived absorbed fractions as a function of beta energy for small fish, large insects and molluscs, and small insects and larvae. Source: NCRP (1991). Figures were enhanced by adding grid lines.

not preferentially sequestered in or near the reproductive tissues. Some elements are known to be preferentially deposited in bone (e.g., strontium). However, reproductive tissues are not generally expected to be hyper-accumulators of radionuclides, based on the available human and animal data (Garten 1981, Garten et al. 1987, and Kaye and Dunaway 1962). The second condition is that the acceptable dose rate to the reproductive tissues are comparable to the acceptable whole-body dose rates. This should be a reasonable assumption if the data used to derive the acceptable limits are based primarily on studies of exposure to high-energy photons (e.g., ^{137}Cs or ^{60}Co), which is generally the case for non-human biota (NCRP 1991). That is, the reproductive organs would not be shielded by other tissues (e.g., muscle, bone, or skin) because high-energy photons would penetrate the organism completely.

The BCF for potassium is from IAEA (1982) because it was excluded from IAEA (1994) without explanation. It is included here because K-40 is a naturally occurring isotope commonly reported in gamma spectral analyses. However, caution is advised in evaluating this and other biologically well-regulated elements (e.g., phosphorous, sodium, and zinc). The use of uptake factors depends on the assumption that the concentration of chemicals in organisms is a linear no-threshold function of the concentration in water. This will not be the case for essential nutrients or toxicants for which the organism has inducible mechanisms for metabolism or excretion. Well-regulated chemicals will have nearly constant concentrations regardless of water concentrations, at least within the effective concentration range for the regulating mechanism. Hence, the reliability of the BCFs (and, consequently, the benchmarks) for such chemicals should be questioned if the screening indicates that they pose an unacceptable risk.

Radiation exposures to fish at the sediment/water interface are likely to be driven by external exposures to contaminated sediments. The exception is for alpha-emitters, such as ^{234}U and ^{230}Th, for which internal exposure is the pathway of concern. Internal concentrations must be estimated from the sediment concentrations if collocated water samples were not collected (see Sect. 2.4). Standard sediment-to-fish transfer factors are not available. Therefore, the concentration in fish was estimated by dividing the sediment concentration (C_s) by the sediment-water partition coefficient (Kd) presented in Table 3 to give the water concentration (C_w). The internal concentration (C_o) was then estimated as the concentration in water times the BCF. This assumes that concentrations in the water near the sediment surface (e.g., the bottom 2–5 cm) are equal to that of undisturbed sediment interstitial water. This is conservative, especially for lotic systems.

The corollary is that the sediment concentration must be estimated from the water concentration if collocated sediment samples were not collected. This was done by multiplying the water concentration (C_w) by the sediment-water partition coefficient (Kd). The resulting sediment concentration was then used to estimate the external dose from sediment. The preferred partition coefficient was the freshwater sediment Kd presented in IAEA Report No. 364 (IAEA 1994). The lowest expected soil-water Kd was used in the absence of a sediment Kd (IAEA 1994). The default soil Kd from Baes et al. (1984) was used only for radionuclides not included in IAEA (1994).

The BCFs and Kds were converted from a wet weight to dry weight basis (Table 3), because they are not clearly defined regarding the use of wet or dry weight tissue and sediment concentrations in their derivation (Baes 1984, IAEA 1982, IAEA 1994, Baker and Soldat 1992). The BCFs were divided by a default wet weight/dry weight conversion factor of 0.2, which assumes that the fresh weight of fish is 80% water (NCRP 1991). This will be conservative for BCFs based on wet weight tissue concentrations (i.e., it will overestimate internal concentrations). The BCF of 1 for tritium was not increased to 5 because ^{3}H is predominantly stored in body water, rather than being organically bound in tissues (NCRP 1979).

The Kds were multiplied by a default wet weight/dry weight conversion factor of 0.75, which assumes that the fresh weight of sediment is 25% water (NCRP 1991). The corrected Kds were used to estimate sediment concentrations from water concentrations (Table 3). This will be conservative for Kds based on dry weight concentrations (i.e., the sediment concentration will be overestimated by a factor of 1.33). However, the uncorrected (original) Kds were used to estimate water concentrations from sediment concentrations. This assumption is conservative because the exposure model assumes that the Kds are based on dry weight concentrations. The use of wet weight sediment concentrations to derive the Kds would result in a smaller Kd values, which would overestimate the concentrations in water.

It is important to note that the BCFs and the Kds used herein are the "expected" values. A more conservative approach would have been to use the most extreme values. As an example, using the highest BCF and the lowest Kd would maximize the estimated transfer from sediment to water and from water to fish. This was not done because many of the extreme values did not appear to be credible (e.g., the maximum BCF for thorium was 10,000), but re-evaluating the original studies was beyond the scope of this effort. Rather, it was presumed that the expected values were not (necessarily) the average but were chosen based on a critical evaluation of the data. This is supported by an inspection of the range of values. For example, the "expected" BCF and sediment Kd values for zirconium were the highest and lowest reported values, respectively (IAEA 1994).

2.3.3 Radioactive Decay Chains

Many radionuclides decay to produce radioactive isotopes (i.e., daughter products) that are relatively short-lived. These progeny may contribute as much or more dose to the target organism than does the parent isotope. For example, ^{137}Cs decays to ^{137m}Ba, which has a half-life of 2.55 minutes and is the sole source of the gamma signature commonly associated with ^{137}Cs. Models of human exposures typically consider isotopes with half-lives less than 180 days to be short-lived (Yu 1993). Uptake of these progeny is not explicitly considered. Instead, the fate of the long-lived parent is modeled, and the short-lived progeny are assumed to be in secular equilibrium with the parent isotope. For example, the internal activity of ^{90}Y is taken to be the same as that of its parent isotope, ^{90}Sr, given sufficient time for in-growth of the daughter. The activity of the daughter may be less than that of the parent if that parent decays to more than one daughter product. For example, ^{218}Po decays to ^{214}Pb 99.98% of the time and to ^{218}At 0.02% of the time. The activity of the short-lived progeny is estimated as the parent activity times the yield of the progeny (e.g., ^{218}At = ^{218}Po * 0.0002). In this example, both daughter products yield 100% Bi-214, producing an activity of ^{214}Bi equal to the activity of ^{218}Po.

The isotopes considered in this manual are, at minimum, those presented in Blaylock, Frank, and O'Neal (1993). Several minor daughter products were added (as were ^{244}Cm and ^{238}Pu). However, the decay chains are fundamentally unchanged, such that progeny with half-lives less than approximately 30 days are considered short-lived and are incorporated into the benchmarks for the parent isotope.

Progeny that account for less than 0.1% of the immediate parent were not considered, with the exception of ^{218}At, which was explicitly included in Blaylock, Frank, and O'Neal (1993).

2.4 BENCHMARK VALUES

The recommended acceptable dose rate to natural populations of aquatic biota is 1 rad d^{-1} (0.4 mGy h^{-1}), based on results of the reviews summarized in NCRP Report No. 109 (NCRP 1991). That recommended limit was intended to apply to the most radiosensitive populations of aquatic organisms. Invertebrates are much more radioresistant than are vertebrates (e.g., fish). For example,

a dose rate of 24 rad d^{-1} delivered during the life cycle of a snail did not significantly reduce reproduction (NCRP 1991). Those reviews are the basis for DOE's guideline for limiting the radiation dose to aquatic biota to 1 rad d^{-1}. Therefore, the aforementioned formulas and assumptions were used to calculate the water and sediment concentrations of selected radionuclides that would result in a total dose rate of 1 rad d^{-1} to two size categories of fish (Table 1).

Two types of benchmarks were derived: single-media benchmarks (Table 4) and multimedia benchmarks (Table 5). All include exposures from parent isotopes and all short-lived daughter products. They also include exposures from all major alpha, beta, and gamma emissions for each isotope. The single-media benchmarks are based on exposures to radionuclides in one medium but not the other. The $Water_{(w)}$ benchmarks include internal and external exposures from water only. The $Sediment_{(s)}$ benchmarks include only external exposures from sediment. These benchmarks are intended to be used when both water and sediment data are available. That is, measured sediment concentrations should be compared to the $Sediment_{(s)}$ values and collocated water measurements should be compared to the $Water_{(w)}$ values (see Chap. 3).

The multimedia benchmarks are for use when only one medium was sampled at a site. The $Water_{(w+s)}$ benchmarks account for internal exposures, external exposures to water, and external exposures to sediment. The sediment concentrations are estimated from the water concentrations using the radionuclide-specific Kds. The $Sediment_{(s+w)}$ benchmarks account for external exposures to sediment plus internal exposures, which were estimated based on the radionuclide-specific transfer factors (i.e., the Kd and BCF). There is considerable uncertainty associated with use of generic transfer factors. Hence, the single-media benchmarks are considered to be more reliable than the multimedia benchmarks because they do not rely on generic sediment/water transfer coefficients. The screening concentrations are presented in pCi L^{-1} for water and pCi g^{-1} for sediment because these are the units typically used on the Oak Ridge Reservation for reporting radionuclide activities. However, the benchmarks can be converted to Bq L^{-1} or Bq g^{-1} by multiplying by 3.7×10^{-2}.

Table 4. Single-media benchmarks for selected radionuclides in water and sediment

	$Water_{(w)}$[b] (pCi L^{-1})		**$Sediment_{(S)}$[c] (pCi g^{-1})**	
Radionuclide[a]	**Small Fish**	**Large Fish**	**Small Fish**	**Large Fish**
Antimony-125	3.72e+05	2.82e+05	1.23e+05	1.33e+05
Barium-140 + D	1.03e+06	8.59e+05	2.07e+04	2.25e+04
Cerium-141	7.62e+05	7.45e+05	6.91e+05	7.28e+05
Cerium-144 + D	1.23e+05	1.00e+05	1.91e+05	1.82e+06
Cesium134	1.10e+04	6.82e+03	3.39e+04	3.65e+04
Cesium-137 + D	7.72e+03	6.19e+03	9.32e+04	1.05e+05
Chromium-51	3.77e+06	1.34e+06	1.66e+06	2.38e+06
Cobalt-60	1.07e+05	4.78e+04	2.10e+04	2.24e+04
Europium-154	2.69e+05	1.89e+05	4.26e+04	4.74e+04
Europium-155	1.23e+06	1.15e+06	8.72e+05	9.46e+05
Hydrogen-3	3.45e+09	3.45e+09		
Iodine-131 + D	4.94e+05	4.30e+05	1.39e+05	1.51e+05
Niobium-95	2.41e+05	9.54e+04	6.88e+04	7.73e+04
Phosphorous-32	1.17e+02	1.13e+02	1.87e+06	

Table 4 (continued)

Radionuclide[a]	Water$_{(W)}$[b] (pCi L^{-1})		Sediment$_{(S)}$[c] (pCi g^{-1})	
	Small Fish	Large Fish	Small Fish	Large Fish
Potassium-40	7.61e+02	7.27e+02	3.16e+05	3.71e+05
Ruthenium-103 + D	3.05e+06	2.31e+06	1.13e+05	1.24e+05
Ruthenium-106 + D	3.82e+05	2.71e+05	8.57e+04	2.88e+05
Sodium-24	3.24e+05	2.13e+05	1.27e+04	1.37e+04
Strontium-90 + D	6.29e+04	5.77e+04	5.57e+05	
Techetium-99	1.94e+06	1.94e+06		
Uranium-237	1.97e+06	1.84e+06	3.70e+05	4.12e+05
Zinc-65	2.80e+05	5.08e+04	9.03e+04	1.01e+05
Zirconium-95	1.05e+05	6.36e+04	7.12e+04	8.01e+04
Plutonium-239	1.25e+03	1.25e+03		
Plutonium-240	1.24e+03	1.24e+03	1.00e+08	5.02e+08
Thorium-232 + D	4.78e+02	4.77e+02	5.47e+04	6.29e+04
Thorium-228 + D	6.01e+01	6.01e+01	3.31e+04	3.69e+04
Americium-241	1.17e+03	1.17e+03	1.67e+06	2.44e+06
Neptunium-237 + D	1.34e+03	1.34e+03	2.23e+05	2.52e+05
Uranium-233	4.00e+03	4.00e+03	1.33e+08	6.63e+08
Thorium-229 + D	5.94e+01	5.94e+01	1.81e+05	2.22e+05
Uranium-238 + D	4.55e+03	4.55e+03	1.75e+06	9.99e+06
Uranium-234	4.04e+03	4.04e+03	1.00e+08	5.02e+08
Thorium-230	4.13e+02	4.13e+02	1.12e+08	5.60e+08
Radium-226 + D	1.60e+02	1.60e+02	2.82e+04	3.32e+04
Lead-210 + D	3.06e+04	3.02e+04	9.77e+06	1.80e+08
Polonium-210	7.25e+02	7.25e+02		
Uranium-235 + D	4.37e+03	4.37e+03	2.96e+05	3.41e+05
Protactinium-231	3.88e+03	3.88e+03	1.11e+06	1.33e+06
Actinium-227 + D	3.97e+02	3.97e+02	1.28e+05	1.52e+05
Curium-244	1.11e+03	1.11e+03	1.02e+08	5.11e+08
Plutonium-238	1.17e+03	1.17e+03	9.59e+07	4.80e+08

[a] "+D" indicates isotopes for which the benchmarks includes the dose rate from short-lived progeny.

[b]Water$_{(W)}$ benchmarks are the concentrations in water that produce a dose rate of 1 rad d^{-1}. They include internal exposures and external exposures from water.

[c]Sediment$_{(S)}$ benchmarks are the dry weight concentrations in sediment that produce a dose rate of 1 rad d^{-1}. They account only for external exposures to sediment. Blank entries indicate isotopes for which external exposure is not significant (i.e., the absorbed fraction is 1).

Table 5. Multimedia benchmarks for selected radionuclides in water and sediment

	Water$_{(W+S)}$[b] (pCi L^{-1})		Sediment$_{(S+W)}$[c] (pCi g^{-1})	
Radionuclide[a]	**Small Fish**	**Large Fish**	**Small Fish**	**Large Fish**
Antimony-125	3.37e+05	2.63e+05	1.48e+04	1.16e+04
Barium-140 + D	3.18e+05	3.16e+05	1.61e+04	1.62e+04
Cerium-141	8.22e+04	8.59e+04	6.34e+05	6.63e+05
Cerium-144 + D	2.11e+04	7.08e+04	1.65e+05	6.46e+05
Cesium134	8.82e+03	5.98e+03	8.29e+03	5.75e+03
Cesium-137 + D	7.27e+03	5.93e+03	7.13e+03	5.84e+03
Chromium-51	3.58e+06	1.32e+06	1.06e+05	3.95e+04
Cobalt-60	5.33e+03	5.31e+03	2.02e+04	2.05e+04
Europium-154	7.99e+04	7.58e+04	3.25e+04	3.17e+04
Europium-155	8.05e+05	7.88e+05	3.62e+05	3.58e+05
Hydrogen-3	3.45e+09	3.45e+09		
Iodine-131 + D	4.81e+05	4.21e+05	4.81e+03	4.21e+03
Niobium-95	1.70e+05	8.31e+04	2.49e+04	1.28e+04
Phosphorous-32	1.17e+02	1.13e+02	1.06e+00	1.01e+00
Potassium-40	7.61e+02	7.27e+02	4.19e+00	4.00e+00
Ruthenium-103 + D	1.44e+06	1.31e+06	6.93e+04	6.43e+04
Ruthenium-106 + D	3.22e+05	2.61e+05	1.70e+04	1.42e+04
Sodium-24	1.11e+05	9.84e+04	9.29e+03	8.48e+03
Strontium-90 + D	5.80e+04	5.77e+04	5.65e+04	5.77e+04
Techetium-99	1.94e+06	1.94e+06	9.69e+03	9.69e+03
Uranium-237	1.65e+06	1.58e+06	7.88e+04	7.61e+04
Zinc-65	1.29e+05	4.28e+04	5.50e+04	2.03e+04
Zirconium-95	4.99e+04	3.99e+04	4.25e+04	3.55e+04
Plutonium-239	1.25e+03	1.25e+03	1.25e+05	1.25e+05
Plutonium-240	1.24e+03	1.24e+03	1.24e+05	1.24e+05
Thorium-232 + D	4.49e+02	4.52e+02	4.40e+03	4.44e+03
Thorium-228 + D	5.93e+01	5.93e+01	5.90e+02	5.91e+02
Americium-241	1.17e+03	1.17e+03	5.83e+03	5.84e+03
Neptunium-237 + D	1.34e+03	1.34e+03	1.34e+01	1.01e+01
Uranium-233	4.00e+03	4.00e+03	2.00e+02	1.50e+02
Thorium-229 + D	5.93e+01	5.93e+01	5.92e+02	5.93e+02
Uranium-238 + D	4.55e+03	4.55e+03	2.27e+02	2.27e+02
Uranium-234	4.04e+03	4.04e+03	2.02e+02	2.02e+02
Thorium-230	4.13e+02	4.13e+02	4.13e+03	4.13e+03
Radium-226 + D	1.60e+02	1.60e+02	7.99e+01	7.99e+01
Lead-210 + D	3.06e+04	3.02e+04	8.25e+03	8.16e+03

Table 5 (continued)

	Water$_{(W+S)}$[b] (pCi L^{-1})		Sediment$_{(S+W)}$[c] (pCi g^{-1})	
Radionuclide[a]	**Small Fish**	**Large Fish**	**Small Fish**	**Large Fish**
Polonium-210	7.25e+02	7.25e+02	1.09e+02	1.09e+02
Uranium-235 + D	4.36e+03	4.36e+03	2.18e+02	2.18e+02
Protactinium-231	3.87e+03	3.88e+03	2.09e+03	2.09e+03
Actinium-227 + D	3.97e+02	3.97e+02	1.79e+02	1.79e+02
Curium-244	1.11e+03	1.11e+03	5.54e+03	5.54e+03
Plutonium-238	1.17e+03	1.17e+03	1.17e+05	1.17e+05

[a]"+D" indicates isotopes for which the benchmarks includes the dose rate from short-lived progeny.

[b]Water$_{(W+S)}$ benchmarks are the concentrations in water that produce a dose rate of 1 rad d^{-1}. They account for internal exposures, external exposures from water, and external exposures from sediment.

[c]Sediment$_{(S+W)}$ benchmarks are the dry weight concentrations in sediment that produce a dose rate of 1 rad d^{-1}. They account for internal exposures and external exposures from sediment. Hydrogen-3 is blank because a Kd was not available (or appropriate).

3. BENCHMARK USE IN ECOLOGICAL RISK ASSESSMENTS

3.1 GENERAL CONSIDERATIONS

These benchmarks are to be used as screening values only. They are for use in ecological risk assessments of natural *populations* of aquatic biota. The recommended limit of 1 rad d^{-1} is not considered appropriate for the assessment of risks to individual organisms, e.g., threatened and endangered species (NCRP 1991). Although these benchmarks can be used in the screening phase of a baseline ecological risk assessment, they must not be used as the sole measure of water or sediment toxicity. Field studies and media toxicity tests will be the primary indicators of toxicity; benchmarks may be used to determine which, if any, radionuclides measured in the ambient water or sediment are likely to contribute (along with other stressors and modifying factors) to the observed effects. This integrative approach allows a more accurate evaluation of adverse ecological impacts, which is necessary in a baseline risk assessment. These benchmarks also do not represent remediation goals. Remediation goals must consider other issues, including adverse effects on habitat and remobilization of contaminants caused by removal or remediation of sediments.

Proper application and interpretation of the benchmarks requires an understanding of the concepts upon which the recommended dose rate limit is based. To wit, it is worth reviewing the conclusions of two critical evaluations of the available data. First, NCRP Report No. 109 (1991) presents the following conclusions:

> "Adoption of a reference level of 0.4 mGy h-1 [1 rad d^{-1}] appears to represent a reasonable compromise based on current information, i.e., considering both the nature of the effects observed at this dose rate and the limited amount of information on effects of ionizing radiation in natural populations, including interactions between ionizing radiation and ecological conditions. Populations exposed to dose rates approaching 0.4 mGy h-1 may also be at risk from other factors such as overexploitation and environmental stressors which might, in combination, result in an undesirable impact. In such circumstances, it would seem highly desirable to conduct a comprehensive ecological evaluation of the radiation exposure regime along with other factors in order to determine the radiological consequences. Thus, it is suggested that where the results of radiological modeling and/or dosimetric measurements indicate that a radiation dose of 0.1 mGy h-1 [0.25 rad d^{-1}] will be exceeded, such an evaluation be conducted."

It is based on this statement that some have suggested using 0.25 rad d^{-1} to derive the benchmark concentrations. This was not done because the consideration of "...other factors such as overexploitation and environmental stressors..." is more appropriately a component of the characterization of risks. The use of an arbitrary safety factor might be warranted, but it should not be embedded in the benchmarks, to the extent practical. Second, an expert panel has subsequently concluded that "the existing data support the application of the [1 rad d^{-1}][1] limit for populations of ...aquatic fauna to *representative* rather than *maximally exposed* individuals" (Barnthouse 1995).

[1] The final version indicated that the limit was 0.1 rad d^{-1} for aquatic biota. Discussions with panel member J. R. Trabalka determined that this was a misstatement of the findings.

These somewhat contradictory conclusions are reflective of the considerable uncertainty associated with the effects of low-level radiation on natural systems.

3.2 APPLICATION OF THE BENCHMARKS

The concentrations of radionuclides detected in water or sediment at a site should be screened against radiological benchmarks by calculating a hazard quotient (HQ) for each radionuclide (Suter 1995). The HQ is the ambient radionuclide concentration divided by the screening value. The dose rate from an individual radionuclide exceeds the acceptable dose rate limit if the HQ is >1. The radiological benchmarks are normalized in an attempt to account for the biological effectiveness of the different types of radiation (i.e., alpha dose rates are multiplied by 20). This allows for the calculation of a hazard index (HI), which is the sum of the HQs for the individual radionuclides in the ambient media. The HI is a measure of the total dose rate to the organism and should account for all three exposure pathways: the total internal dose, the total external dose from water, and the total external dose from sediment.

As mentioned previously, two types of benchmarks have been derived. Single-media benchmarks include the $Water_{(w)}$ and the $Sediment_{(s)}$ benchmarks, which are for use when collocated water and sediment samples are available (Table 4). The HI for an organism (e.g., small fish) at a location is calculated as the sum of the HQs for the isotopes in water (e.g., the concentration in water divided by the small fish $Water_{(w)}$ benchmark) and the HQs for the isotopes in sediment (e.g., the concentration in sediment divided by the small fish $Sediment_{(s)}$ benchmark). If the HI exceeds 1, then the total dose rate to a small fish is estimated to exceed the recommended limit. The $Water_{(w)}$ benchmark includes the external dose from immersion in contaminated water. Use of this benchmark in conjunction with the $Sediment_{(s)}$ benchmark is conservative: the fish would not be exposed to water from below because the use of $Sediment_{(s)}$ benchmark assumes that the fish resides at the sediment surface. This could be accounted for by dividing the external dose from water by two, which is comparable to what is done to estimate the sediment exposure (i.e., use one half of the MeV dis^{-1} to rad d^{-1} conversion factor). However, this is considered to be a relatively minor source of error. The expected Kds for all of the selected radionuclides are greater than two, and most are more than an order of magnitude greater than two. Hence, radiation from the sediment will dominate the external dose rates.

Multimedia benchmarks include the $Water_{(w+s)}$ and $Sediment_{(s+w)}$ benchmarks, which are for use when only one of the media was sampled (Table 5). In this case, the HI for an organism is simply the sum of the HQs for the isotopes measured in that medium (see Sect. 2.4). The $Water_{(w+s)}$ benchmark assumes that the fish resides at the sediment surface and it includes external exposure from water under the fish, which is comparable to using the $Water_{(w)}$ and $Sediment_{(s)}$ benchmarks together. The $Sediment_{(s+w)}$ benchmarks do not include the external dose from water. As noted above, this is considered a minor source of error relative to the external radiation from the sediment.

The collection of collocated sediment and water samples is strongly recommended, so that single-media benchmarks can be used instead of multimedia benchmarks. The $Water_{(w+s)}$ and $Sediment_{(s+w)}$ benchmarks are calculated using default sediment/water partition coefficients. There is considerable uncertainty associated with these factors, which make the multimedia benchmarks inherently less reliable than the single-media benchmarks.

Given the uncertainties mentioned in Sect. 3.1, a compromise approach to screening is recommended. Specifically, the maximum exposure concentration (e.g., the upper 95% confidence limit on the mean water concentration or the maximum sediment concentration) should be compared with a threshold of 0.25 rad d^{-1} as an initial screening tool. If the threshold is exceeded (i.e., the HI is >0.25), further evaluation could include comparing representative exposures (e.g., mean water and

median sediment concentrations) to the 0.25 rad d^{-1} threshold. Of course, any screening approach presumes that the media were sufficiently sampled to provide reliable estimates of exposure, an issue which is beyond the scope of this document. Other modifying factors and site-specific data also should be considered if the potential for radiological risks is indicated by the screening.

3.3 EXAMPLE APPLICATION

Two screening examples are worked below to aid interpretation of these benchmarks. The water and sediment data are taken from the *Remedial Investigation of Bear Creek Valley at the Oak Ridge Y-12 Plant* (DOE 1997). The concentrations in water are the upper 95% confidence limit on the mean for Reach 1, which is the headwaters of Bear Creek. The concentrations in sediment are from the single sediment sample collected at station SD02, which is in Reach 1.

In Example 1, the water and sediment data are treated as collocated samples (Table 6). The water concentrations are screened against the $Water_{(w)}$ benchmarks for small fish, and the sediment concentrations are screened against the $Sediment_{(s)}$ benchmarks for small fish. The HI-Water is the sum of the HQs for water, and the HI-Sediment is the sum of the HQs for sediment. The HI-Total is the sum of all HQs for small fish. The HI-Total of 0.03 suggests that the radionuclides measured at this location pose a negligible risk to aquatic biota.

In Example 2, the water and sediment data are evaluated separately, i.e., as if one or the other were available but not both (Table 7). The water concentrations are screened against the $Water_{(w+s)}$ benchmarks for small fish, and the sediment concentrations are screened against the $Sediment_{(s+w)}$ benchmarks. The HI-Total is the sum of all HQs for a medium. In this example, the HI based on water data only (0.0314) is equal to the HI-Total based on the collocated water and sediment samples (Table 6). The HI based on sediment data only (0.211) is nearly an order of magnitude higher than the HI-Total in Example 1. This is because the dose rates from most of these isotopes are dominated by internal exposures. The need to estimate the internal concentration in the organism from the sediment concentration using the Kd adds a level of conservatism for such radionuclides. Conversely, the dose rate from isotopes that are primarily sources of external radiation (e.g., ^{137}Cs and ^{60}Co) would likely be conservatively estimated when using the $Water_{(w+s)}$ benchmarks. As in Example 1, the screenings against multimedia benchmarks also suggests that the radionuclides measured at this location pose a negligible risk to aquatic biota, though somewhat less convincingly than the single-media values. Again, both screenings presume that the media were sufficiently sampled to provide reliable estimates of exposure.

Table 6. Example 1: Use of single-media benchmarks for the calculation of hazard quotients (HQs) and hazard indices (HIs)

Radionuclide	Concentration[a]	Benchmark[b]	HQ[c]
	Water (pCi L^{-1})		
Strontium-90	1.33	6.29e+04	2.11e-05
Technetium-99	327	1.94e+06	1.69e-04
Thorium-228	0.144	6.01e+01	2.40e-03
Thorium-230	0.117	4.13e+02	2.83e-04
Thorium-232	0.081	4.78e+02	1.69e-04
Uranium-233/234	37.9	4.00e+03	9.48e-03
Uranium-235	2.33	4.37e+03	5.33e-04
Uranium-238	83.1	4.55e+03	1.83e-02
		HI - Water	3.13e-02
	Sediment (pCi g^{-1})		
Americium-241	0.06	1.67e+06	3.59e-08
Cesium-137	0.18	9.32e+04	1.93e-06
Technetium-99[d]	8.74	N/A	N/A
Thorium-228	1.45	3.31e+04	4.38e-05
Thorium-230	1.03	1.12e+08	9.20e-09
Thorium-232	0.99	5.47e+04	1.81e-05
Uranium-234	16.77	1.00e+08	1.68e-07
Uranium-235	0.73	2.96e+05	2.47e-06
Uranium-238	27.38	1.75e+06	1.56e-05
		HI - Sediment	8.22e-05
		HI - Total	**3.14e-02**

[a]Water and sediment concentrations are from DOE (1997).

[b]Benchmarks are the $Water_{(W)}$ and $Sediment_{(S)}$ values for small fish.

[c]The hazard quotient is the media concentration divided by the benchmark value. The hazard index is the sum of the hazard quotients.

[d]Technetium-99 does not have a $Sediment_{(S)}$ benchmark because external exposure is not a significant pathway.

Table 7. Example 2: Use of multimedia benchmarks for the calculation of hazard quotients (HQs) and hazard indices (HIs)

Radionuclide	Concentration[a]	Benchmark[a]	HQ[c]
	Water (pCi L^{-1})		
Strontium-90	1.33	5.80e+04	2.29e-05
Technetium-99	327	1.94e+06	1.69e-04
Thorium-228	0.144	5.93e+01	2.43e-03
Thorium-230	0.117	4.13e+02	2.83e-04
Thorium-232	0.081	4.49e+02	1.80e-04
Uranium-233/234	37.9	4.00e+03	9.48e-03
Uranium-235	2.33	4.36e+03	5.34e-04
Uranium-238	83.1	4.55e+03	1.83e-02
		HI - Total	**3.14e-02**
	Sediment (pCi g^{-1})		
Americium-241	0.06	5.83e+03	1.03e-05
Cesium-137	0.18	7.13e+03	2.52e-05
Technetium-99	8.74	9.69e+03	9.02e-04
Thorium-228	1.45	5.90e+02	2.46e-03
Thorium-230	1.03	4.13e+03	2.49e-04
Thorium-232	0.99	4.40e+03	2.25e-04
Uranium-234	16.77	2.02e+02	8.30e-02
Uranium-235	0.73	2.18e+02	3.35e-03
Uranium-238	27.38	2.27e+02	1.21e-01
		HI - Total	**2.11e-01**

[a]Water and sediment concentrations are from DOE (1997).

[b]Benchmarks are the $Water_{(W+S)}$ and $Sediment_{(S+W)}$ values for small fish.

[c]The hazard quotient is the media concentration divided by the benchmark value. The hazard index is the sum of the hazard quotients.

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