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Created By: Tamsen.Dozier@nrc.gov

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<AdvancedReactors-GEISDocsNPR.Resource3@nrc.gov>
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DOE-STD-1153-2002
July 2002

DOE STANDARD

A GRADED APPROACH FOR EVALUATING RADIATION DOSES TO AQUATIC AND TERRESTRIAL BIOTA

U.S. Department of Energy
Washington, D.C. 20585

AREA ENVR

Foreword

1. Department of Energy (DOE) activities may expose populations of plants and animals to radioactive materials in environmental media, or to radioactive materials released in waste streams. This DOE voluntary consensus technical standard provides methods, models, and guidance within a graded approach that DOE personnel and contractors may use to characterize radiation doses to aquatic and terrestrial biota that are exposed to radioactive materials.

2. The graded approach to biota dose evaluation can be used to address requirements for radiological protection of the environment contained in DOE Orders. It can also be used to support radiological protection of the environment program elements within Environmental Management Systems (EMS) at DOE sites.

3. These methods (and the Biota Concentration Guides contained in them) are not intended to be used as design criteria, indicators of the severity of accidental releases of radioactive materials, or guides for mitigating the consequences of accidental releases. Furthermore, this technical standard does not apply to the irradiation of biota for experimental purposes, nor to research or experimental studies.

4. This technical standard and the RAD-BCG Calculator (an electronic calculational tool provided with the technical standard) can be downloaded from the Department's Biota Dose Assessment Committee (BDAC) web site (http://homer.ornl.gov/oepa/public/bdac).

5. The graded approach to biota dose evaluation and associated guidance contained in this technical standard is also intended for use with the RESRAD-BIOTA code. The RESRAD-BIOTA dose evaluation code was designed to be consistent with the graded approach and the BCGs contained herein.

6. DOE technical standards, such as this standard, do not establish requirements. However, all or part of the provisions in a DOE standard can become requirements under the following circumstances:

   (a) they are explicitly stated to be requirements in a DOE requirements document; or

   (b) the organization makes a commitment to meet a standard in a contract or in an implementation plan or program plan required by a DOE requirements document.

   Throughout this standard, the word "shall" is used to denote actions which must be performed if the objectives of this standard are to be met. If the provisions in this standard are made requirements through one of the two ways discussed above, then the "shall" statements would become requirements. However, "should" statements would not automatically be converted to "shall" statements if provisions in this standard become requirements, as this action would violate the consensus process used to approve this standard.

7. This technical standard has undergone extensive review throughout its development: (1) it was prepared and reviewed by the Department's Biota Dose Assessment Committee (BDAC), an approved DOE Technical Standards Program topical committee; (2) it has undergone a formal DOE review and comment resolution process as required by the Department's Technical Standards Program; (3) it was made available to other federal agencies for their review and comment through the Interagency Steering Committee on Radiation Standards (ISCORS); (4) it was reviewed by an independent external technical expert; and (5) five papers on the graded approach methodology and associated guidance contained in this technical standard have undergone external peer review for publication in scientific journals.

8. Comments in the form of recommendations, pertinent data, and lessons learned from implementation of DOE's graded approach to biota dose evaluation that may improve future versions of this technical standard, the RAD-BCG Calculator, or the RESRAD-BIOTA code, are welcome and should be sent to:

Mr. Stephen Domotor
U.S. Department of Energy
Office of Environment, Safety and Health
Air, Water, and Radiation Division (EH-412)
1000 Independence Avenue, S.W.
Washington, DC 20585-0119
Stephen.Domotor@eh.doe.gov

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Acknowledgments

This voluntary consensus technical standard was prepared by the Department's Air, Water and Radiation Division (EH-412) and the Core Team of the Biota Dose Assessment Committee (BDAC). The BDAC is a technical standards topical committee organized under the Department of Energy Technical Standards Program. The purpose of the BDAC is (a) to assist, consistent with DOE needs, in developing and promoting technical standards and associated guidance for DOE-wide applications in assessing radiation dose to biota, (b) to serve as a major forum within DOE for obtaining technical assistance, discussing technical issues, and sharing lessons learned regarding biota dose standards and assessment methods, and (c) to serve as a technical resource and advisory group for DOE program and field elements in the design and review of site-specific biota dose assessments. The committee has broad representation from DOE Offices, national laboratories, universities, and the private sector. The BDAC charter can be obtained from the BDAC web site at: http://homer.ornl.gov/oepa/public/bdac.

A guiding principle for the BDAC is that both "developers" and "users" be part of the methods development process. Consistent with the BDAC's values and guiding principles documented in the BDAC charter, this technical standard was prepared using an interdisciplinary team approach. Each member of the Core Team brought with them specific expertise in health physics, ecology, radioecology, environmental monitoring, or risk assessment. The collective knowledge gained through this teaming orientation proved to be essential for developing the methods and implementation guidance presented in this technical standard.

The Core Team consists of the following members: Mr. Ernest Antonio, Pacific Northwest National Laboratory (PNNL); Dr. Gordon Bilyard, PNNL; Mr. Stephen Domotor, DOE-EH-412; Dr. Gary Friday, Westinghouse Savannah River Company (WSRC); Dr. Kathryn Higley, Oregon State University; Mr. Daniel Jones, Oak Ridge National Laboratory (ORNL); Dr. David Kocher, SENES-Oak Ridge; Dr. Randall Morris, Environmental Science and Research Foundation, and TREC, Inc.; Dr. Bradley Sample, CH_2MHill; and Ms. Patricia Scofield, ORNL.

Members of the Core Team, and other members of the BDAC, served as lead developers or contributors for several key areas of the technical standard. These individuals, and their specific contributions, are highlighted below. We are grateful to them for their contributions.

Technical Standard Development

BDAC Chairperson: Mr. Stephen Domotor (EH-412); technical standard preparation, integration, and coordination: Mr. Stephen Domotor (EH-412), with support from Ms. Audrey Lamanna, Ms. Melissa Hatcher, Mr. Jamie McDonald, and Mr. Clyde Lichtenwalner (Energetics, Inc.); screening methodology concepts and development: Dr. Kathryn Higley (Oregon State University) and Dr. David Kocher (SENES-Oak Ridge); kinetic/allometric modeling concepts and development, and RAD-BCG Calculator design: Dr. Kathryn Higley (Oregon State University).

Specific Contributions

Conceptual framework and application of the graded approach: Mr. Stephen Domotor (EH-412), with support from the BDAC Core Team; primer on ecological risk assessment concepts and issues concerning the evaluation of radiation as a stressor: Mr. Daniel Jones (ORNL); interpretation and application of biota dose limits: Dr. David Kocher (SENES-Oak Ridge); sources, receptors, and routes of exposure: Dr. Gordon Bilyard (PNNL); defining the area of evaluation: Dr. Randall Morris (Environmental Science and Research Foundation, and TREC, Inc.); dealing with high background levels of naturally-occurring radionuclides: Dr. Randall Morris (Environmental Science and Research Foundation, and TREC, Inc.) and Mr. Daniel Jones (ORNL); soil sampling guidance: Dr. Gordon Bilyard (PNNL) and Mr. Daniel Jones (ORNL); biota sampling guidance: Mr. Daniel Jones (ORNL) and Dr. Bradley Sample (CH_2MHill); guidance on radiation weighting factor for alpha particles: Dr. David Kocher (SENES-Oak Ridge); evaluating dose to individual organisms: Dr. David Kocher (SENES-Oak Ridge); derivation of dose coefficients, dose equations and models, and BCGs: Dr. Kathryn Higley (Oregon State University), Dr. David Kocher (SENES-Oak Ridge), Mr. Ernest Antonio (PNNL), and Mr. Stephen Domotor (EH-412); preparation of example applications of the graded approach: Mr. Ernest Antonio (PNNL) and Ms. Patricia Scofield (ORNL).

We are grateful to Dr. Gary Friday of the Westinghouse Savannah River Corporation (WSRC) for his valuable comments and recommendations concerning ecological risk issues and technical considerations.

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We thank all of the BDAC members who provided comments and recommendations on previous drafts of this technical standard, and who provided recommendations on the screening and analysis methods which were piloted at the 1999 BDAC Meeting. We also thank the representatives from federal and state agencies, and international organizations, who provided review comments on previous drafts of this technical standard.

We thank Mr. Fred Baes (ORNL) for BDAC web site design, database design, and maintenance support.

We thank Mr. Richard Serbu of the Department's Technical Standards Program (DOE-TSP) for guidance and support provided in establishing the BDAC.

We thank the DOE-TSP Managers and points of contact for their helpful comments and suggestions on this technical standard during the DOE-TSP formal review process.

We thank Mr. Andrew Lawrence, Director of the Office of Environmental Policy and Guidance (EH-41); Mr. Andrew Wallo III, Director of the Air, Water, and Radiation Division (EH-412); and Mr. Harold Peterson, EH-412 Environmental and Radiological Evaluation Unit Leader, for sponsoring the BDAC and supporting the development of this technical standard.

We thank Mr. Raymond Berube, Deputy Assistant Secretary for Environment (EH-4), for his support and recognition of the BDAC's pioneering spirit and efforts in developing the biota dose evaluation methods in this technical standard.

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Scope, Purpose and Organization

This technical standard provides methods, models, and guidance within a graded approach that the U.S. Department of Energy (DOE) and its contractors may use to evaluate doses of ionizing radiation to populations of aquatic animals, terrestrial plants, and terrestrial animals from DOE activities for the purpose of demonstrating protection relative to Dose Rate Guidelines. It provides dose evaluation methods that can be used to meet the requirements of DOE Order 5400.5, “Radiation Protection of the Public and the Environment” (1990a) and DOE Order 5400.1, “General Environmental Protection Program” (1990b). The technical standard assumes a threshold of protection for plants and animals at the following doses: for aquatic animals, 1 rad/d (10 mGy/d); for terrestrial plants, 1 rad/d (10 mGy/d); and for terrestrial animals, 0.1 rad/d (1 mGy/d). Available data indicate that dose rates below these limits cause no measurable adverse effects to populations of plants and animals.

The DOE graded approach includes a screening method and three more detailed levels of analysis for demonstrating compliance with applicable dose limits for protection of biota. The general screening method provides appropriately conservative limiting concentrations of radionuclides in environmental media (termed "Biota Concentration Guides" or BCGs). Radionuclide concentrations in samples of environmental media are easily compared with the BCGs to evaluate compliance with biota dose limits. The three more detailed analysis methods require more effort, but yield more accurate and realistic biota dose evaluations.

This technical standard is designed to be user-friendly, and is organized into three principal Modules for ease of implementation. Material in each Module is cross-referenced to pertinent sections in other Modules. There is some duplication of material across Modules by design, in order to allow each to be used separately, if desired. Module 1 serves as the principal users guide for step-by-step implementation of the graded approach to biota dose evaluation. Module 2 serves as a resource guide, providing detailed guidance for implementing key elements of the graded approach identified in Module 1, and providing a “primer” on technical issues to be considered when evaluating radiation as a stressor to the environment. Module 3 serves as a technical reference source, providing the technical basis for the derivation of dose models, screening values, and selection of default assumptions and parameters applied in the graded approach. The organization and content of the technical standard are provided in Figure 1.

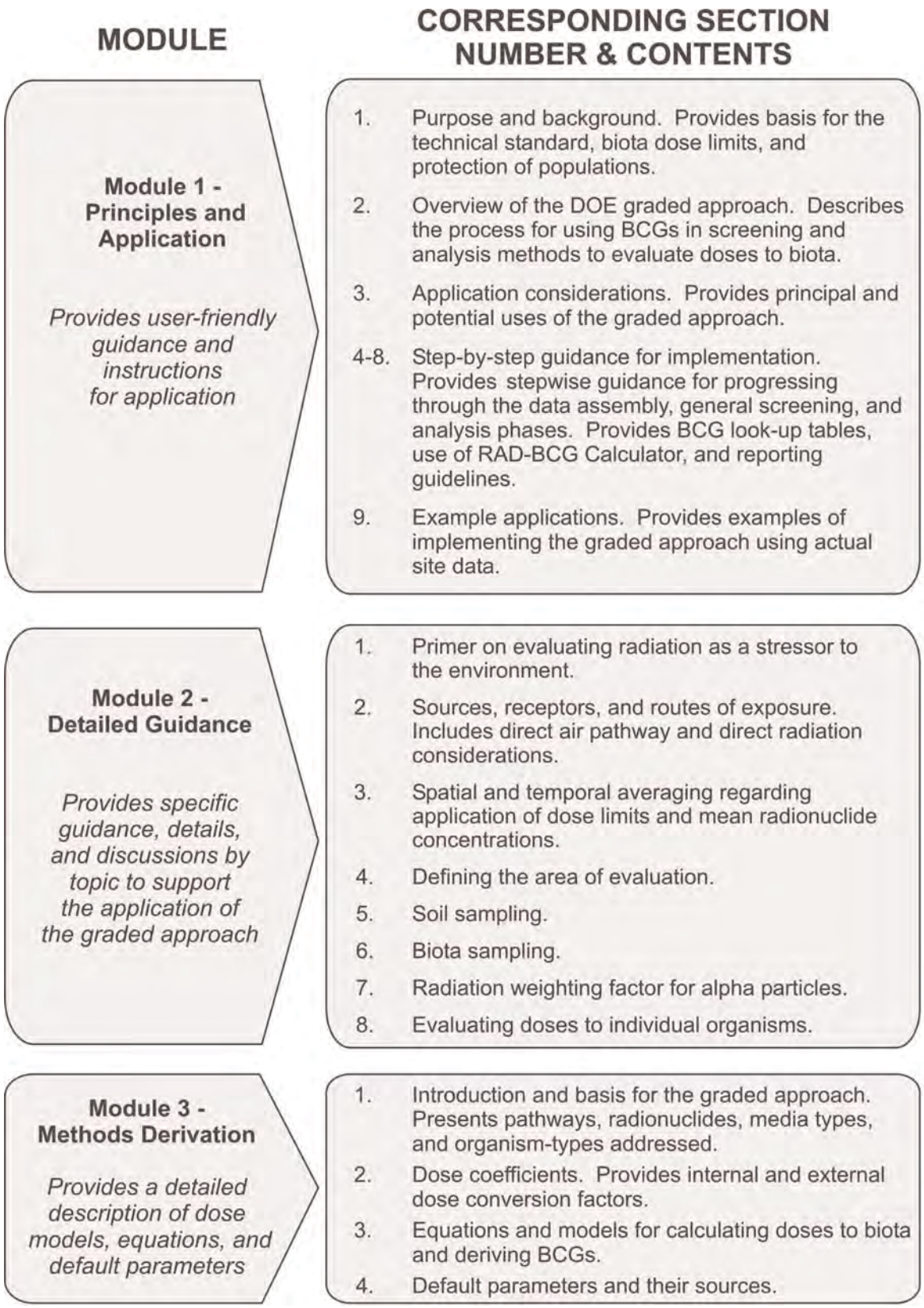

Figure 1 Organization and Contents of the DOE Technical Standard

References

Addy, C. E. 1956. *Guide to Waterfowl Banding*. U.S. Fish and Wildlife Service, Laurel, Maryland.

American Society for Quality Control (ASQC). 1994. *American National Standard: Specifications and Guidelines for Quality Systems for Environmental Data Collection and Environmental Technology Programs*. ANSI/ASQC E4-1994, ASQC, Milwaukee, Wisconsin.

American Society for Testing and Materials (ASTM). 1995. *Standard Guide for Developing Conceptual Site Models for Contaminated Sites.* ASTM E 1689-95, Philadelphia, Pennsylvania.

American Society for Testing and Materials (ASTM). 1997. *Annual Book of ASTM Standards. Vol. 11.05. Biological Effects and Environmental Fate; Biotechnology; Pesticides*. ASTM, West Conshohoken, Pennsylvania.

Amiro, B. D. 1997. "Radiological Dose Conversion Factors for Generic Non-Human Biota Used for Screening Potential Radiological Impacts." *J. Environ. Radioactivity* 35(1):37-51.

Anderson, R. O., and R. M. Neumann. 1996. "Length, Weight, and Associated Structural Indices." In *Fisheries Techniques, 2nd Edition*, eds. B. R. Nurphy and D. Willis, pp. 447-482. American Fisheries Society, Bethesda, Maryland.

Baes, C. F., R. D. Sharp, A. L. Sjoreen, and R. W. Shor. 1984. *A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides through Agriculture*. ORNL-5786, Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Baker, D. A., and J. K. Soldat. 1992. *Methods for Estimating Doses to Organisms from Radioactive Materials Released into the Aquatic Environment*. PNL-8150, Pacific Northwest Laboratory, Richland, Washington.

Barnthouse, L. W. 1995. *Effects on Ionizing Radiation on Terrestrial Plants and Animals: A Workshop Report.* Environmental Sciences Division, ORNL/TM-13141, Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Bastian, R.K., and W.B. Jackson. 1975. "137 Cs and 60 Co in a Terrestrial Community at Enewetak Atoll." In *Radioecology and Energy Resources*, Publication No. 1, May, ed. C.E. Cushing, pp. 12-14. Oregon State University, Oregon.

Bilyard, C. R., H. Beckert, J. J. Bascietto, C. W. Abrams, S. A. Dyer, and L. A. Haselow. 1997. *Using the Data Quality Objectives Process During the Design and Conduct of Ecological Risk Assessments*. DOE/EH-0544, U.S. Department of Energy, Office of Environmental Policy and Assistance, Washington, D.C.

Blaylock, B.G., M.L. Frank, and B.R. O'Neal. 1993. *Methodology for Estimating Radiation Dose Rates to Freshwater Biota Exposed to Radionuclides in the Environment*, ES/ER/TM-78. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Breckenridge, R. P., and A. Crocket. 1998. "Determination of Background Concentrations of Inorganics in Soils and Sediments at Hazardous Waste Sites." *Environ. Monit. Assess*. 51:621-656.

Breda, N., A. Granier, F. Barataud, and C. Moyne. 1995. "Soil Water Dynamics in an Oak Stand." *Plant and Soil* 172: 17-27.

Bub, H. 1990. *Bird Trapping and Bird Banding.* Cornell University Press, Ithaca, New York.

Burmaster, D. E., and D. A. Hull. 1997. "Using Lognormal Distributions and Lognormal Probability Plots in Probabilistic Risk Assessments." *Human Ecol. Risk Assess*. 3:235-255.

Burt, W. H., and R. P. Grossenheider. 1980. *A Field Guide to the Mammals of North America North of Mexico*. Houghton Mifflin Co., Boston, Massachusetts.

Bysshe, S. E. 1988. "Uptake by Biota." In *Environmental Inorganic Chemistry: Properties, Processes, and Estimation Methods*. eds. I. Bodek, W. J. Luman, W. F. Reehl and D. H. Rosenblatt, pp. 4-1 to 4.7-1. Pergamon Press, New York.

Calder, W. A., III. 1984. *Size, Function, and Life History*. Harvard University Press, Cambridge, Massachusetts.

California Environmental Protection Agency (California EPA). 1997. *Selecting Inorganic Constituents as Chemicals of Potential Concern at Risk Assessments at Hazardous Waste Sites and Permitted Facilities*. Final Policy Report, Human and Ecological Risk Division, Department of Toxic Substances Control, California EPA, Sacramento, California.

Call, M. Y. 1986. "Rodents and Insectivores." In *Inventory and Monitoring of Wildlife Habitat*, eds. A. Y. Cooperrider, R. J. Boyd, and H. R. Stuart, pp. 429-452. U.S. Dept. Inter., Bur. Land Manage. Service Center, Denver, Colorado.

Canadell, J., R.B. Jackson, J.R. Ehleringer, H.A. Mooney, O.E. Sala, and E.D. Schulze. 1996. "Maximum Rooting Depth of Vegetation Types at a Global Scale." *Oecologia* 108: 583-595.

Canadian Nuclear Safety Commission, Advisory Committee on Radiological Protection (CNSC-ACRP). 2002. *Protection of Non-Human Biota From Ionizing Radiation*. Document No. INFO-0730. June 2002.

Cooper, R. J., and R. C. Whitmore. 1990. "Arthropod Sampling Methods in Ornithology." *Studies in Avian Biology* 13:29–37.

Conant, R. 1975. *A Field Guide to Reptiles and Amphibians of Eastern and Central North America*. Houghton Miflin Co., Boston.

Copplestone, D., S. Bielby, S. Jones. D. Patton, P. Daniel, and I. Gize. 2001. *Impact Assessment of Ionizing Radiation on Wildlife*. Environment Agency, Rio House, Bristol (BS32 4UD), UK.

Craig, J. F. 1980. "Sampling with Traps." In *Guidelines for Sampling Fish in Inland Waters*, eds. T. Backiel and R. L. Welcomme, pp. 55–70. Food and Agricultural Organization of the United Nations, Rome, Italy.

Cummins, C.L. 1994. *Radiological Bioconcentration Factors for Aquatic, Terrestrial, and Wetland Ecosystems at the Savannah River Site*. WSRC-TR-94-0391. Westinghouse Savannah River Company, Aiken, South Carolina.

Dietrick, E. J., E. I. Schlinger, and R. Van den Bosch. 1959. "A New Method for Sampling Arthropods Using a Suction Collection Machine and a Berlese Funnel Separator." *Journal of Economic Entomology* 52:1085-1091.

Dowdy, S., and S. Wearden. 1983. *Statistics for Research*. John Wiley & Sons, New York.

Dunning, J. B., Jr. 1984. *Body Weights of 686 Species of North American Birds*. Western Bird Banding Association, Monograph No. 1, Eldon Publishing, Cave Creek, Arizona.

Dunning, J. B., Jr. 1993. *CRC Handbook of Avian Body Masses*. CRC Press, Inc., Boca Raton, Florida.

Eisler, R. 1994. *Radiation Hazards to Fish, Wildlife, and Invertebrates: a Synoptic Review*. Biological Report 26, National Biological Service, U.S. Department of Interior, Washington, D.C.

Endangered Species Act. 1973. Public Laws 93-205 through 100-707, as amended, 87 stat. 884, 16 USC 1531 et seq.

Eyman, L.D., and J.R. Trabalka. "Patterns of Transuranic Uptake by Aquatic Organisms: Consequences and Implications." In *Transuranic Elements in the Environment*, ed. Wayne Hanson, pp. 612-624.

Foxx, T.S., G.D. Tierney, and J.M. Williams. 1984. *Rooting Depths of Plants on Low-Level Waste Disposal Sites.* LA-10253-MS. Los Alamos National Laboratory, Los Alamos, New Mexico.

Friday, G.P., C.L. Cummins, and A.L. Schwartzman. 1996. *Radiological Bioconcentration Factors for Aquatic, Terrestrial, and Wetlands Ecosystems at the Savannah River Site*. WSRC-TR-96-0231. Westinghouse Savannah River Company, Aiken, South Carolina.

Gamble, J.F. 1971. *A Proposed Mechanism for Recycling Radiocesium in Florida Soil Systems*. Third National Symposium on Radioecology, conf-71501-p1. Oak Ridge, Tennessee.

Gentner, N. 2002. "What Harm Do We Wish to Protect Against and How Should it be Assessed." Nuclear Energy Agency, Radiological Protection of the Environment: "The Path Forward to a New Policy?" Taormina, Sicily, Italy, February 12-14, 2002.

Garten, C. T., Jr. 1981. "Comparative Uptake of Actinides by Plants and Rats from the Shoreline of a Radioactive Pond." *J. Environ. Qual*. 10:487-491.

Garten, C. T., Jr., E. A. Bondietti, J. R. Trabalka, R. L. Walker, and T. G. Scott. 1987. "Field Studies on the Terrestrial Behavior of Actinide Elements in East Tennessee." In *Environmental Research on Actinide Elements*, eds. J. E. Pinder III, J. J. Alberts, K. W. McLeod, and R. G. Schreckhise, pp. 109-119 in CONF-841142, U.S. Department of Energy, Washington, D.C.

Gerzabek, M.H., F. Strebl, and B. Temmel. 1998. *Plant uptake of radionuclides in lysimeter experiments.* Environmental Pollution 99: 93-103.

Gilbert, R.O. 1987. *Statistical Methods for Environmental Pollution Monitoring*. John Wiley and Sons, New York.

Gilman, E.F. 1989. "Tree Root Depth Relative to Landfill Tolerance." *HortScience* 245:857.

Green, R. H. 1979. *Sampling Design and Statistical Methods for Environmental Biologists*. John Wiley and Sons, New York.

Greenleaf-Jenkins, J., and R. J. Zasoski. 1986. "Distribution, Availability, and Foliar Accumulation of Heavy Metals from Dewatered Sludge Applied to Two Acid Forest Soils." In *Nutritional and Toxic Effects of Sewage Sludge in Forest Ecosystems*, eds. S. D. West and R. J. Zasoski, College of Forest Resources, University of Washington, Seattle.

Gullet, P. A. 1987. "Euthanasia." In *Field Guide to Wildlife Diseases: General Field Procedures for Migratory Birds*, ed. M. Friend, pp. 59-63, Res. Publ.167, U.S. Fish and Wildlife Service, Washington, D.C.

Gunn, A. 1992. "Use of Mustard to Estimate Earthworm Populations." *Pedobiologia*. 36:65–67.

Hamley, J. M. 1980. "Sampling with Gillnets." In *Guidelines for Sampling Fish in Inland Waters*, eds. T. Backiel and R. L. Welcomme, pp. 37-53. Food and Agricultural Organization of the United Nations, Rome, Italy.

Hartley, W. G. 1980. "The Use of Electrical Fishing for Estimating Stocks of Freshwater Fish." In *Guidelines for Sampling Fish in Inland Waters*, eds. T. Backiel and R. L. Welcomme, pp. 91-95. Food and Agricultural Organization of the United Nations, Rome, Italy.

Hays, R. L., C. Summers, and W. Seitz. 1981. *Estimating Wildlife Habitat Variables*. FWS/OBS-81/47. U.S. Fish and Wildlife Service, Washington, D.C.

Heyer, W. R., M. A. Donnelly, R. W. McDiarmid, L. C. Hayek, and M. S. Foster. 1994. *Measuring and Monitoring of Biological Diversity: Standard Methods for Amphibians*. Smithsonian Institution Press, Washington, D.C.

Howe, J.R., and M.k. Lloyd. 1986. "Radio-Iodine in Thyroid Glands of Swans, Farm Animals and Humans, also in Algae and River Water from the Thames Valley, England." In *Science and the Total Environment* 48:13-31.

Hubbell, J. H. 1969. *Photon Cross Sections - Attenuation Coefficients and Energy Absorption Coefficients from 10 KeV to 100 GeV*. NSRDS-NBS-29, National Bureau of Standards, U.S. Government Printing Office, Washington, D.C.

Hubert, W. A. 1983. "Passive Capture Methods." In *Fisheries Techniques*, eds. L. A. Nielsen and D. L. Johnson, pp. 95-122. American Fisheries Society, Bethesda, Maryland.

International Atomic Energy Agency (IAEA). 1976. *Effects of Ionizing Radiation Aquatic Organisms and Ecosystems*. Technical Report No. 172, IAEA, Vienna, Austria.

International Atomic Energy Agency (IAEA). 1990. *Recommendations of the International Commission on Radiological Protection*. Publication 60, Pergamon Press, Oxford and New York.

International Atomic Energy Agency (IAEA). 1992. *Effects of Ionizing Radiation on Plants and Animals at Levels Implied by Current Radiation Protection Standards*. Technical Report Series No. 332, IAEA, Vienna, Austria.

International Atomic Energy Agency (IAEA). 1994. *Manual of Parameter Values for the Prediction of Radionuclide Transfer in Temperate Environments*. Technical Report Series No. 364, IAEA, Vienna, Austria.

International Atomic Energy Agency (IAEA). 1999. *Protection of the Environment from the Effects of Ionizing Radiation: A Report for Discussion*. IAEA-TECDOC-1091, IAEA, Vienna, Austria.

International Commission on Radiological Protection (ICRP). 1977. *Recommendations of the International Commission on Radiological Protection*. ICRP Publication 26, Pergamon Press, New York.

International Commission on Radiological Protection (ICRP). 1979a. *Limits for the Intake of Radionuclides by Workers*. ICRP Publication 30 Part 1, Pergamon Press, New York.

International Commission on Radiological Protection (ICRP). 1979b. *Limits for the Intake of Radionuclides by Workers*. ICRP Publication 30 Supplement to Part 1, Pergamon Press, New York.

International Commission on Radiological Protection (ICRP). 1980. *Limits for the Intake of Radionuclides by Workers*. ICRP Publication 30 Part 2, Pergamon Press, New York.

International Commission on Radiological Protection (ICRP). 1981a. *Limits for the Intake of Radionuclides by Workers*. ICRP Publication 30 Supplement to Part 2, Pergamon Press, New York.

International Commission on Radiological Protection (ICRP). 1981b. *Limits for the Intake of Radionuclides by Workers*. ICRP Publication 30 Part 3, Pergamon Press, New York.

International Commission on Radiological Protection (ICRP). 1982a. *Limits for the Intake of Radionuclides by Workers*. ICRP Publication 30 Supplement A to Part 3, Pergamon Press, New York.

International Commission on Radiological Protection (ICRP). 1982b. *Limits for the Intake of Radionuclides by Workers*. ICRP Publication 30 Supplement B to Part 3, Pergamon Press, New York.

International Commission on Radiological Protection (ICRP). 1983. *Annals of the ICRP*, "Radionuclide Transformations: Energy and Intensity of Emissions." ICRP Publication 38, Vols. 11-13, Pergamon Press, New York.

International Commission on Radiological Protection (ICRP). 1988. *Limits for the Intake of Radionuclides by Workers*. ICRP Publication 30 Part 4, Pergamon Press, New York.

International Commission on Radiological Protection (ICRP). 1991. *Recommendations of the International Commission on Radiological Protection*. ICRP Publication 60, Pergamon Press, New York.

Isom, B. G. 1978. "Benthic Macroinvertebrates." In *Methods for the Assessment and Prediction of Mineral Mining Impacts on Aquatic Communities: a Review and Analysis Workshop Proceedings*, ed. W. T. Mason, Jr., pp. 67-74. U.S. Fish and Wildlife Service, Harpers Ferry, West Virginia.

Jackson, R.B., J. Canadell, J.R. Ehleringer, H.A. Mooney, O.E. Sala, and E.D. Schulze. 1996. "A Global Analysis of Root Distributions in Terrestrial Biomes." *Oecologia* 108: 389-411.

Jacobs, K. E., and W. D. Swink. 1982. "Estimations of Fish Population Size and Sampling Efficiency of Electrofishing and Rotenone in Two Kentucky Tailwaters." *North American Journal of Fisheries Management* 2:239–248.

Jenkins, J.H., and T.T. Fendley. 1971. *Radionuclide Biomagnification in Coastal Plain Deer*. Third National Symposium on Radioecology, conf-710501-p1, pp. 116-122. Oak Ridge, Tennessee.

Jiang, Q. Q., and B. R. Singh. 1994. "Effect of Different Forms and Sources of Arsenic on Crop Yield and Arsenic Concentration." *Water, Air, Soil Pollut*. 74:321-343.

Johnson, C. G., and L. R. Taylor. 1955. "The Development of Large Suction Traps for Airborne Insects." *Annals of Applied Biology* 43:51–62.

Jones, C., W. J. McShea, M. J. Conroy, and T. H. Kunz. 1996. "Capturing Mammals." In *Measuring and Monitoring Biological Diversity: Standard Methods for Mammals*, eds. D. E. Wilson, F. R. Cole, J. D. Nichols, R. Rudran, and M.S. Foster, pp. 115-155. Smithsonian Inst. Press, Washington, D.C.

Jones, D.S. 2000. "Radiological Benchmarks for Effects on Aquatic Biota at the Oak Ridge Reservation." *Human and Ecological Risk Assessment*, 6, 789-807.

Jones, K. B. 1986. "Amphibians and Reptiles." In *Inventory and Monitoring of Wildlife Habitat*, eds. A. Y. Cooperrider, R. J. Boyd, and H. R. Stuart, pp. 267-290. Bur. Land Manage. Service Center, U.S. Dept. Inter., Denver, Colorado.

Julliet, J. A. 1963. "A Comparison of Four Types of Traps Used for Capturing Flying Insects." *Canadian Journal of Zoology* 41:219–223.

Kaye, S. V., and P. B. Dunaway. 1962. "Bioaccumulation of Radioactive Isotopes by Herbivorous Small Mammals." *Health Phys*. 7:205-217.

Kennedy, W. E., Jr., and D. L. Strenge. 1992. *Residual Radioactive Contamination from Decommissioning*. NUREG/CR-5512, PNL-7994, Vol. 1, Pacific Northwest Laboratory, Richland, Washington.

Keyes, B. E., and C. E. Grue. 1982. "Capturing Birds with Mist Nets: a Review." *North American Bird Bander* 7:2–14.

Klepper, E.L., K.A. Gano and L.L. Cadwell. 1985. *Rooting depth and distributions of deep-rooted plants in the 200 area control zone of the Hanford Site.* PNL-5247/UC-11. Pacific Northwest National Laboratory, Richland, Washington.

Kocher, D. C. 1980. *A Radionuclide Decay Data Base—Index and Summary Table*. NUREG/CR-1413, ORNL/NUREG-70, Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Kocher, D. C. 1981. *Radioactive Decay Data Tables—A Handbook of Decay Data for Application to Radiation Dosimetry and Radiological Assessments*. DOE/TIC-11026, U.S. Department of Energy, Oak Ridge, Tennessee.

Kocher, D. C. 1983. "Dose Conversion Factors for External Exposure to Photons and Electrons." *Health Physics* 45(3):665-686.

Kocher, D. C., and K. F. Eckerman. 1981. "Electron Dose-Rate Conversion Factors for External Exposure of the Skin." *Health Phys.* 40(4):467-475.

Krebs, C. J. 1989. *Ecological Methodology*. Harper & Row, New York.

Kunz, T. H. 1988a. *Ecological and Behavioral Methods for the Study of Bats*. Smithsonian Inst. Press, Washington, D.C.

Kunz, T. H. 1988b. "Methods of Assessing the Availability of Prey to Insectivorous Bats." In *Ecological and Behavioral Methods for the Study of Bats*, ed. T. H. Kunz, pp. 191-210. Smithsonian Inst. Press, Washington, D.C.

Kunz, T. H., R. Rudran, and G. Gurry-Glass. 1996. "Human Health Concerns." In *Measuring and Monitoring Biological Diversity: Standard Methods for Mammals*, eds. D. E. Wilson, F. R. Cole, J. D. Nichols, R. Rudran, and M. S. Foster, pp. 255-264. Smithsonian Inst. Press, Washington, D.C.

Land, C. E. 1975. "Tables of Confidence Limits for Linear Functions of the Normal Mean and Variance." In *Selected Tables in Mathematical Statistics*. Vol. III, pp. 385-419. Am. Math. Soc., Providence, Rhode Island.

Layher, W. G., and O. E. Maughan. 1984. "Comparison Efficiencies of Three Sampling Techniques for Estimating Fish Populations in Small Streams." *Progress in Fisheries Culture* 46:180–184.

Ludwig, J. A, and J. F. Reynolds. 1988. *Statistical Ecology: a Primer and Methods and Computing*. John Wiley & Sons, New York.

Murkin, H. R., D. A. Wrubleski, and F. A. Ried. 1994. "Sampling Invertebrates in Aquatic and Terrestrial Habitats." In *Research and Management Techniques for Wildlife and Habitats*, 5th Edition, ed. T. A. Bookhout, pp. 349–369. The Wildlife Society, Bethesda, Maryland.

Nagorsen, D. W., and R. L. Peterson. 1980. *Mammal Collectors' Manual: a Guide for Collecting, Documenting, and Preparing Mammal Specimens for Scientific Research*. Royal Ontario Museum, Life Sciences Miscellaneous Publications, Toronto, Canada.

Napier, B. A., R. A. Peloquin, D. L. Strenge, and J. V. Ramsdell. 1988a. *Conceptual Representation. Volume 1 of GENII - The Hanford Environmental Radiation Dosimetry Software System*. PNL-6584, Vol. 1, Pacific Northwest Laboratory, Richland, Washington.

Napier, B. A., R. A. Peloquin, D. L. Strenge, and J. V. Ramsdell. 1988b. *User's Manual. Volume 2 of GENII - The Hanford Environmental Radiation Dosimetry Software System*. PNL-6584, Vol. 2, Pacific Northwest Laboratory, Richland, Washington.

Napier, B. A., R. A. Peloquin, D. L. Strenge, and J. V. Ramsdell. 1988c. *Code Maintenance Manual. Volume 3 of GENII - The Hanford Environmental Radiation Dosimetry Software System*. PNL-6584, Vol. 3, Pacific Northwest Laboratory, Richland, Washington.

National Council on Radiation Protection and Measurements (NCRP). 1991. *Effects of Ionizing Radiation on Aquatic Organisms*. NCRP Report No. 109, NCRP, Bethesda, Maryland.

Newman, M. C. 1995. *Quantitative Methods in Aquatic Ecotoxicology*. Lewis Publishers, Boca Raton, Florida.

Ng, Y. C. 1982. "A Review of Transfer Factors for Assessing the Dose from Radionuclides in Agricultural Products." *Nuclear Safety* 23(1):57-71.

Oak Ridge National Laboratory (ORNL). 1997. *Oak Ridge Reservation Annual Site Environmental Report for 1996*. ES/ESH-73, Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Onishi, Y., R. J. Serne, E. M. Arnold, C. E. Cowan, and F. L. Thompson. 1981. *Critical Review: Radionuclide Transport, Sediment Transport, and Water Quality Mathematical Modeling; and Radionuclide Adsorption/Desorption Mechanisms*. NUREG/CR-1322, PNL-2901, Pacific Northwest Laboratory, Richland, Washington.

Ophel, I.L., and C.D. Fraser. 1971. *The Fate of Cobalt-60 in a Natural Freshwater Ecosystem*. Third National Symposium on Radioecology, conf-710501-p1.

Parker, M.M. and D.H. Van Lear. 1996. "Soil Heterogeneity and Root Distribution of Mature Loblolly Pine Stands in Piedmont Soils." *Soil Sci. Soc. Am. J.* 60: 1920-1925.

Polikarpov, G. G. 1994. "CIS Workshop in Radioecology." *IUR Newsletter* 16:6-7.

Price, K. R. 1971. *A Critical Review of Biological Accumulation, Discrimination and Uptake of Radionuclides Important to Waste Management Practices*. BNWL-B-148, Pacific Northwest Laboratories, Richland, Washington.

Provost, L. P. 1984. "Statistical Methods in Environmental Sampling." In *Environmental Sampling for Hazardous Wastes*, eds., G. F. Schweitzer and J. A. Santolucito. American Chemical Society Symposium Series, Washington, D.C.

Raw, F. 1959. "Estimating Earthworm Populations Using Formalin." *Nature* 148:1661–1662.

Reiss, M.J. 1989. *The Allometry of Growth and Reproduction*. Cambridge University Press, Cambridge, England.

Reynolds, J. B. 1983. "Electrofishing." In *Fisheries Techniques*, eds. L. A. Nielsen and D. L. Johnson, pp. 147–164. American Fisheries Society, Bethesda, Maryland.

Rice, T.R. 1961. *Review of Zinc in Ecology*. Proceedings of the First National Symposium of Radioecology, pp. 619-631, eds. V. Schultz and A.W. Klement.

Rickard, W.H. and L.J. Kirby. 1987. "Trees as Indicators of Subterranean Waster Flow from a Retired Radioactive Waste Disposal Site." *Health Physics* 52(2): 201-206.

Romney, E.M., W.A. Rhoads, A. Wallace, and R.A. Wood. 1971. *Persistence of radionuclides in soil, plants, and small mammals in areas contaminated with radioactive fallout*. Third National Symposium on Radioecology, conf-71501-p1.

Ryti, R., EE. Kelley, M. Hooten, G. Gonzales, G. McDermott, and L. Soholt. 1999. *Screening Level Ecological Risk Assessment Methods*. LA-UR-99-1405, Los Alamos National Laboratory.

Sadana, U. S., and B. Singh. 1987. "Yield and Uptake of Cadmium, Lead and Zinc by Wheat Grown in a Soil Polluted with Heavy Metals." *J. Plant Sci. Res*. 3:11-17.

Sample, B. E., M. S. Aplin, R. A. Efroymson, G. W. Suter II, and C. J. E. Welsh. 1997. *Methods and Tools for Estimation of the Exposure of Terrestrial Wildlife to Contaminants*. ORNL/TM-13391, Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Satchell, J. E. 1970. "Measuring Population and Energy Flow in Earthworms." In *Methods of Study in Soil Ecology*, ed. J. Phillipson, pp. 261–267. UNESCO, New York.

Schemnitz, S. D. 1994. "Capturing and Handling Wild Animals." In *Research and Management Techniques for Wildlife and Habitats*, 5th Edition, ed. T. A. Bookhout, pp. 106–124. The Wildlife Society, Bethesda, Maryland.

Schmidt-Neilsen, K. 1977. "Problems of Scaling: Locomotion and Physiological Constraints." In *Scale Effects in Animal Locomotion*, ed. T.J. Pedley, pp. 1-22. Academic Press, London, England.

Shine, R. 1986. *Diets and Abundances of Aquatic and Semi-aquatic Reptiles in the Alligator Rivers Region*. Technical Memorandum No. 16, Australian Government Publishing Service, Canberra, Australia.

Shleien, B., L. A. Slayback, Jr., and B. K. Kirby, eds. 1998. *Handbook of Health Physics and Radiological Health*. 3rd ed. Williams & Wilkins, Baltimore, Maryland.

Silva, M., and J. A. Dowling. 1995. *CRC Handbook of Mammalian Body Masses*, pp. 359. CRC Press, Inc., Boca Raton, Florida.

Smock, L. A., J. E. Gladden, J. L. Riekenberg, L. C. Smith, and C. R. Black. 1992. "Lotic Macroinvertebrate Production in Three Dimensions: Channel Surface, Hyporheic, and Floodplain Environments." *Ecology* 73:876–886.

Snedecor, G. W., and W. G. Cochran. 1980. *Statistical Methods*. 7th Edition. Iowa State Univ. Press, Ames, Iowa.

Sokal, R. R., and F. J. Rohlf. 1981. *Biometry: the Principles and Practice of Statistics in Biological Research*, 2nd Edition. W. H. Freeman and Co., New York.

Southwood, T.R.E. 1978. *Ecological Methods*. Chapman and Hall, New York.

Springett, J. A. 1981. "A New Method for Extracting Earthworms from Soil Cores, with a Comparison of Four Commonly Used Methods for Estimating Populations." *Pedobiologia* 21:217–222.

Stebbins, R. C. 1966. *A Field Guide to Western Reptiles and Amphibians*. Houghton Mifflin Co., Boston, Massachusetts.

Suter, G. W., II. 1993. *Ecological Risk Assessment*. Lewis Publishers, Boca Raton, Florida.

Suter, G. W. 1995. *Guide for Performing Screening Ecological Risk Assessments at Doe Facilities*. ES/ER/TM-153, Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Suter, G. W. 1996. *Guide for Developing Conceptual Models for Ecological Risk Assessment.* ES/ER/TM-186, Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Suter, G. W., II., B. E. Sample, D. S. Jones, and T. L. Ashwood. 1995. *Approach and Strategy for Performing Ecological Risk Assessments on the Oak Ridge Reservation*. 1995 rev. ES/ER/TM-33/R2, Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Suter, G.W., II, R.A. Efroymson, B.E. Sample, and D.S. Jones. 2000. *Ecological Risk Assessment for Contaminated Sites*. Lewis Publishers, Boca Raton, Florida.

Talmage, S. S., and B. T. Walton. 1993. "Food Chain Transfer and Potential Renal Toxicity to Small Mammals at a Contaminated Terrestrial Field Site." *Ecotoxicology* 2:243-256.

Temple, P. J., and R. Wills. 1979. "Sampling and Analysis of Plants and Soil." In *Manual of Methodology for the Assessment of Air Pollution Effects on Vegetation*, eds. W. W. Heck, S. V. Krupa, and S. N. Linzon, pp. 13-1–13-23.

Tierney, G.D. and T.S. Foxx. 1987. *Rooting lengths of plants on Los Alamos National Laboratory Lands. LA-10865-MS/UC-48*. Los Alamos National Laboratory, Los Alamos, New Mexico.

Till, J.E. and H.R. Meyer. (1983) *Radiological Assessment: A Textbook on Environmental Dose Analysis*. NUREG/CR-3332, ORNL-5968. Prepared for Division of Systems Integration, Office of Nuclear Reactor Regulation, U.S. Nuclear Regulatory Commission. Washington, DC.

United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR). 1996. *Effects of Radiation on the Environment*. A/AC.82/R.549, United Nations, Vienna.

U.S. Department of Defense (DOD), U.S. Department of Energy (DOE), U.S. Environmental Protection Agency (EPA), and U.S. Nuclear Regulatory Commission (NRC). 2000. *Multi-Agency Radiation Survey and Site Investigation Manual (MARSSIM).* Revision 1, NRC Report NUREG-1575, Washington, D.C.

U.S. Department of Energy (DOE). 1984. *Atmospheric Science and Power Production*. DOE/TIC-27601, D. Randerson, ed., DOE, Washington, D.C.

U.S. Department of Energy (DOE). 1984. "Iodine in Terrestrial Wildlife on the U.S. Department of Energy's Hanford Site in South central Washington." *Environmental Monitoring and Assessment* 4:379-388.

U.S. Department of Energy (DOE). 1987. *The Environmental Survey Manual*. DOE/EH-0053, U.S. Department of Energy, Office of Environmental Audit, Washington, D.C.

U.S. Department of Energy (DOE). 1990a. *Radiation Protection of the Public and the Environment*. DOE Order 5400.5.

U.S. Department of Energy (DOE). 1990b. *General Environmental Protection Program*. DOE Order 5400.1.

U.S. Department of Energy (DOE). 1991. *Environmental Regulatory Guide for Radiological Effluent Monitoring and Environmental Surveillance*. DOE/EH-0173T, DOE, Washington, D.C.

U.S. Department of Energy (DOE). 1993. "10 CFR 834, Radiation Protection of the Public and the Environment; Proposed Rule." *Federal Register*, March 25, 1993. Volume 58, No. 56, pp. 16268-16322.

U.S. Department of Energy (DOE). 1995. *Estimating the Cold War Mortgage. The 1995 Baseline Environmental Management Report*. DOE/EM-0232. Office of Environmental Management, U.S. Department of Energy.

U.S. Department of Energy (DOE). 1997. *Site Conceptual Exposure Model Builder User Manual*. Office of Environmental Policy and Guidance, RCRA/CERCLA Division (EH-413).

U.S. Department of Energy (DOE). 1996. *Baseline Environmental Management Report.* DOE/EM-0290. Office of Environmental Management, U.S. Department of Energy.

U.S. Department of Energy (DOE). 1998. *Compendium of EPA-Approved Analytical Methods for Measuring Radionuclides in Drinking Water.* Office of Environmental Policy and Assistance, U.S. Department of Energy.

U.S. Department of Energy (DOE). 1999a. *U.S. Department of Energy Biota Dose Assessment Committee Summary Report*. 1999 Annual Topical Committee Meeting. Office of Environmental Policy and Guidance, Air, Water and Radiation Division. September 27, 1999.

U.S. Department of Energy (DOE). 1999b. *Quality Assurance*. DOE Order 414.1A.

U.S. Department of Energy (DOE). 2000a. *Availability of DOE Technical Standard, "A Graded Approach for Evaluating Radiation Doses to Aquatic and Terrestrial Biota" (Project ENVR-0011), for use in DOE Compliance and Risk Assessment Activities.* Washington, DC: DOE memorandum from Dr. David Michaels (EH-1); 2000 (July 19).

U.S. Department of Energy (DOE). 2000b. *Guidance Memorandum: Guidance for the Preparation of Department of Energy (DOE) Annual Site Environmental Reports for Calendar Year 1999.* Dr. David Michaels (EH-1) to Distribution. April 21, 2000.

U.S. Environmental Protection Agency (EPA). 1993. *Wildlife Exposure Factors Handbook*. EPA/600/R-93/187 a and b, Vols. I and II, EPA, Washington, D.C.

U.S. Environmental Protection Agency (EPA). 1994a. *Vegetation Assessment Field Protocol*. ERT SOP #2037. Rev. 0.0, EPA, Washington, D.C.

U.S. Environmental Protection Agency (EPA). 1994b. *Field Studies for Ecological Risk Assessment*. Eco Update 2(3). EPA 540-F-94-014, EPA, Washington, D.C.

U.S. Environmental Protection Agency (EPA). 1994c. *Guidance for the data quality objectives process.* EPA QA/G-4. Quality Assurance Management Staff, Washington, D.C.

U.S. Environmental Protection Agency (EPA). 1996. *Vegetation Assessment Field Protocol*. ERT SOP #2038. Rev. 0.0, EPA, Washington, D.C.

U.S. Environmental Protection Agency (EPA). 1997a. *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessment, Interim Final*. U.S. Environmental Protection Agency, Environmental Response Team, Edison, New Jersey.

U.S. Environmental Protection Agency (EPA). 1997b. *Guidance on Cumulative Risk Assessment*. Part 1. Planning and Scoping. Science Policy Council, EPA, Washington, D.C.

U.S. Environmental Protection Agency (EPA). 1998. *Guidelines for Ecological Risk Assessment.* EPA/630/R-95/002F, U.S. Environmental Protection Agency, Risk Assessment Forum, EPA, Washington, D.C.

U.S. Environmental Protection Agency (EPA). 1999. *Ecological Risk Assessment and Risk Management Principles for Superfund Sites*. OSWER Directive 9285. 7-28 P. Office of Solid Waste and Emergency Response, Washington, D.C.

U.S. Fish and Wildlife Service and Canadian Wildlife Service. 1977. *North American Bird Banding Manual*. Vol. II. U.S. Dept. of Interior, Washington, D.C.

Verkhovskaja, I.N., P.P. Vavilov, and V.I. Maslov. 1967. *The Migration of Natural Radioactive Elements*. International Symposium in Stockholm, April 1966. Eds. B. Aberg and F.P. Hungate, Pergamon Press, Oxford, England.

Washington State Department of Ecology (WADOE). 1992. *Statistical Guidance for Ecology Site Managers*. Publ. No. 92-54, Washington State Dept. of Ecology, Toxics Cleanup Program, Olympia, Washington.

Washington State Department of Ecology (WADOE). 1994. *Natural Background Soil Metals Concentrations in Washington State*. Publ. No. 94-115, Washington State Dept. of Ecology, Toxics Cleanup Program, Olympia, Washington.

West, G.B., Brown, J.H., and B.J. Enquist. 1977. "A General Model for the Origin of Allometric Scaling Laws in Biology." *Science* 276:122-126.

Whicker, F.W. 1997. "Impacts on Plant and Animal Populators." *Health impacts for large releases of radionuclides,* 74-93. Ciba Foundation, London, England.

Whicker, F.W., Pinder, J.E., III, Bowling, J.W., Alberts, J.J., and I.L. Brisban. 1990. "Distribution of Long-Lived Radionuclides in an Abandoned Reactor Cooling Reservoir." *Ecological Monographs* 60: 471-496.

Whicker, F.W. and V. Schultz. 1982. *Radioecology: Nuclear Energy and the Environment*. Vol. I and II. CRC Press, Inc., Boca Raton, Florida.

Whicker, F.W., Shaw, G., Voigt, G, and E. Holm. 1999. "Radioactive Contamination: State of the Science and its Application to Predictive Models." *Environmental Pollution* 100: 133-149.

Wiley, M. L., and C-F. Tsai. 1983. "The Relative Efficiencies of Electrofishing Vs. Seines in Piedmont Streams of Maryland." *North American Journal of Fisheries Management* 3:243–253.

Wilkie, D.R. 1977. "Metabolism and Body Size." In *Scale Effects in Animal Locomotion*. Ed. T.J. Pedley. Academic Press, London, England.

Williams, D. D., and H.B.N. Hynes. 1973. "The Occurrence of Benthos Deep in the Substratum of a Stream." *Freshwater Biology* 4:233–256.

Wilson, D. E., F. R. Cole, J. D. Nichols, R. Rudran, and M. S. Foster. 1996. *Measuring and Monitoring Biological Diversity: Standard Methods for Mammals*. Smithsonian Inst. Press, Washington, D.C.

Woodhead, D. S. 1998. "The Impact of Radioactive Discharges on Native British Wildlife and the Implications on Environmental Protection." *R & D Technical Report P135*, Environmental Agency, Bristol, UK, 80 pp.

Yu, C., J.J. Cheng, L.G. Jones, Y.Y. Wang, Y.P. Chia, and E. Fallaice. 1993. *Data Collection Handbook to Support Modeling Impacts of Radioactive Material in Soil*. Argonne National Laboratory, Argonne, Illinois.

Zach, R. 1985. "Contribution of Inhalation by Food Animals to Man's Ingestion Dose." *Health Physics* 49: 737-745.

Zar, J. H. 1984. *Biostatistical Analysis*, 2nd Edition. Prentice Hall, Inc., Englewood Cliffs, New Jersey.

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Definitions

As defined and used in this technical standard:

Absorbed Dose (D) is the energy imparted to matter by ionizing radiation per unit mass of irradiated material at the place of interest in that material. More specifically, for any radiation type and any medium, absorbed dose (D) is the total energy (e) absorbed per unit mass (m) of material: D = e/m. The absorbed dose is expressed in units of rad (gray), where 1 rad = 0.01 joule/kg material (1 gray = 100 rad). For the purposes of this technical standard, the absorbed dose in an organism is assumed to be the average value over the whole organism.

Allometric refers to the relative growth of a part in relation to the entire organism.

Alpha Particle is a helium-4 nucleus consisting of two protons and two neutrons, given off by the decay of many heavy elements, including uranium and plutonium. Because the particles are slow moving as well as heavy, alpha radiation can be blocked by a sheet of paper. However, once an alpha emitter is in living tissue, it can cause substantial damage because of the high ionization density along its path.

Aquatic Biota is plant or animal life living in or on water.

Arithmetic Mean is the most commonly used measure of central tendency, commonly called the “average.” Mathematically, it is the sum of all the values of a set divided by the number of values in the set:

$$\bar{X} = \frac{\sum_{i=1}^{n} X_i}{n}$$

Assessment Endpoint is an explicit expression of the environmental value that is to be protected, operationally defined by an ecological entity and its attributes. For example, salmon are valued ecological entities; reproduction and age class structure are some of their important attributes. Together "salmon reproduction and age class structure" form an assessment endpoint.

Average - See “Arithmetic Mean.”

Beta Particle is an electron. It has a short range in air. Beta particles are moderately penetrating and can cause skin burns from external exposure, but can be blocked by a sheet of plywood.

Bias is a consistent underestimation or overestimation of the true values representing a population.

Bioaccumulation is the ratio of the contaminant concentration in the organism relative to the contaminant concentration in an environmental medium resulting from the uptake of the

contaminant from one or more routes of exposure. This ratio is typically described through a bioaccumulation factor (B_{iv}).

Biomagnification is the tendency of some contaminants to accumulate to higher concentrations at higher levels in the food web through dietary accumulation.

Biota is plant and animal life of a particular region.

Biota Concentration Guide (BCG) is the limiting concentration of a radionuclide in soil, sediment, or water that would not cause dose limits for protection of populations of aquatic and terrestrial biota (as used in this technical standard) to be exceeded.

Carnivore is a flesh-eating animal.

Chronic refers to an extended continuous exposure to a stressor or the effects resulting from such an exposure.

Community is an assemblage of populations of different species within a specified location in space and time.

Conceptual Model is a written description and visual representation of predicted relationships between ecological entities and the stressors to which they may be exposed.

Data Quality Objectives (DQOs) are qualitative and quantitative statements that clarify technical and quality objectives for a study, define the appropriate type of data, and specify tolerable levels of uncertainty that a data user is willing to accept in the decision. DQOs specify the problem to be solved, the decision, the inputs to the decision, the boundaries of the study, the decision rule, and the limits of uncertainty.

Deterministic Effects are those for which the severity is a function of dose, and for which a threshold usually exists.

Discharge Point is a conduit through which any radioactively contaminated gas, water, or solid is discharged to the atmosphere, waters, or soils.

Distribution Coefficient is the ratio of the mass of solute species absorbed or precipitated on the soil or sediment to the solute concentration in the water. This ratio is typically described through a K_d factor.

Ecological Relevance is one of three criteria for assessment endpoint selection. Ecologically relevant endpoints reflect important characteristics of the system and are functionally related to other endpoints.

Ecological Risk Assessment is the process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors.

Effluent is any treated or untreated air emission or liquid discharge, including storm water runoff.

Effluent Monitoring is the collection and analysis of samples or measurements of liquid, gaseous, or airborne effluents for the purpose of characterizing and quantifying contaminant levels and process stream characteristics, assessing radiation exposures to members of the public and the environment, and demonstrating compliance with applicable standards.

Environmental Medium is a discrete portion of the total environment, animate or inanimate, that may be sampled or measured directly.

Environmental Surveillance is the collection and analysis of samples of air, water, soil, foodstuffs, biota, and other media and the measurement of external radiation and radioactive materials for purposes of demonstrating compliance with applicable standards, assessing radiation exposures to members of the public, and assessing effects, if any, on the local environment.

Error is the difference between an observed or measured value and its true value.

Exposure is the co-occurrence or contact between the endpoint organism and the stressor (e.g., radiation or radionuclides).

Facility means a building, structure, or installation subject to the regulations/standards pertinent to this technical standard.

Forb is an herb other than grass.

Gamma Rays are high-energy electromagnetic photons similar to X-rays. They are highly penetrating and several inches of lead or several feet of concrete are necessary to shield against them.

Geometric Mean is mathematically expressed as the n^{th} root of the product of all values in a set of n values:

$$\bar{X}_g = \left[\prod_{i=1}^{n} X_i\right]^{1/n}$$

or as the antilogarithm of the arithmetic mean of the logarithms of all the values of a set of n values:

$$\bar{X}_g = \text{antilog}\left[\frac{\sum_{i=1}^{n} \log X_i}{n}\right]$$

The geometric mean is generally used when the logarithms of a set of values are normally distributed, as is the case for much of the monitoring and surveillance data.

Geometric Standard Deviation is mathematically expressed as the antilog of the standard deviation of the logarithms of the measurements:

$$S_g = \text{antilog}\left[\sum_{i=1}^{n}\left[\frac{\log X_i - \frac{\sum_{i=1}^{n}\log X_i}{n}}{n-1}\right]^2\right]^{1/2} \quad X_i > 0$$

Grab Sample is a single sample acquired over a short interval of time.

Herbivore is a plant-eating animal.

Lentic refers to living in or relating to still waters (as lakes, ponds, or swamps).

Lotic refers to living in or relating to actively moving water (as streams or rivers).

Median is the middle value of a set of data when the data are ranked in increasing or decreasing order. If there is an even number of values in the set, the median is the arithmetic average of the two middle values; if the number of values is odd, it is the middle value.

Mode refers to the value occurring most frequently in a data set.

Monitoring is the use of instruments, systems, or special techniques to measure liquid, gaseous, solid, and/or airborne effluents and contaminants.

Nuclide refers to an isotope, either stable or unstable, of any chemical element.

Phylogenetic refers to the evolution of a genetically related group of organisms as distinguished from the development of the individual organism.

Poikilothermic refers to a cold-blooded organism.

Population is an aggregate of individuals of a species within a specified location in space and time.

Proportional Sample is a sample consisting of a known fraction of the original stream.

Quality refers to the totality of features and characteristics of a material, process, product, service, or activity that bears on its ability to satisfy a given purpose.

Quality Assurance (QA) refers to those planned and systematic actions necessary to provide adequate confidence that a measurement represents the sampled population. Quality assurance includes quality control (QC), which comprises all those actions necessary to control and verify the features and characteristics of a material, process, product, or service to specified requirements.

Quality Control (QC) refers to those actions necessary to control and verify the features and characteristics of a material, process, product, service, or activity to specified requirements. The aim of quality control is to provide quality that is satisfactory, adequate, dependable, and economical.

Rad is a unit of absorbed dose of ionizing radiation equal to an energy of 100 ergs per gram of irradiated material.

Radiation (Ionizing) refers to alpha particles, beta particles, photons (gamma rays or x-rays), high-energy electrons, and any other particles capable of producing ions.

Radioactive Material refers to any material or combination of materials that contain radionuclides that spontaneously emits ionizing radiation.

Radionuclide is an unstable nuclide that undergoes spontaneous transformation, emitting radiation. There are approximately 2,200 known radionuclides, both man-made and naturally occurring. A radionuclide is identified by the number of neutrons and protons in the atomic nucleus and its half-life.

Random Error refers to variations of repeated measurements made within a sample set that are random in nature and individually not predictable. The causes of random error are assumed to be indeterminate or non-assignable. Random errors are generally assumed to be normally distributed.

Random Samples are samples obtained in such a manner that all items or members of the lot, or population, have an equal chance of being selected in the sample.

Range is the difference between the maximum and minimum values of a set of values.

Relative Biological Effectiveness (RBE) is defined as the ratio of the absorbed dose of a reference radiation (normally gamma rays or X rays) required to produce a level of biological response to the absorbed dose of the radiation of concern required to produce the same level of biological response, all other conditions being kept constant.

Representative Individual is an individual organism within a population that receives a radiation dose which is equivalent to the value of the appropriate measure of central tendency (i.e., mean, median, mode) of the distribution of doses received by that population. The individual is assumed to be representative of the population as a whole.

Representative Sample is a sample taken to depict the characteristics of a lot or population as accurately and precisely as possible. A representative sample may be a “random sample” or a “stratified sample” depending upon the objective of the sampling and the characteristics of the conceptual population.

Riparian Organisms are those organisms related to, living, or located on the bank of a natural watercourse (as a river) or sometimes of a lake or a tidewater.

Safety Factor is a factor applied to an observed or estimated toxic concentration or dose to arrive at a criterion or standard that is considered safe.

Sample has two definitions: 1) A subset or group of objects selected from a larger set, called the “lot” or “population;” and 2) an extracted portion or subset of an effluent stream or environmental media.

Sampling is the extraction of a prescribed portion of an effluent stream or of an environmental medium for purposes of inspection and/or analysis.

Sequential Sampling refers to timed samples collected from an effluent stream.

Site refers to the land or property upon which DOE facilities or activities are located and access to which is subject to Departmental or DOE contractor control.

Source (Radioactive) is either (1) a known amount of radioactive material emanating a characteristic amount of energy in the form of alpha, beta, gamma, neutron, or x-ray emissions (or a combination of such emissions), or (2) a single process or release point that contributes to or causes a release to the environment and that can be separated from other processes by a break in the flow of material.

Standard Deviation is an indication of the dispersion of a set of results around the average of samples collected or the mean of a population; it is the positive square root of the sample variance. For samples taken from a population, the standard deviation, s, is calculated as:

$$s = \left[\frac{\sum_{i=1}^{n} (X_i - \bar{X})^2}{n - 1} \right]^{1/2}$$

where $\bar{X}$ = average value of the samples measured;

n = number of samples measured; and

X_i = individual measurement value for sample I.

For a finite population, the standard deviation (σ) is

$$\sigma = \left[\frac{\sum_{i=1}^{N} (X_i - \mu)^2}{N} \right]^{1/2}$$

where μ is the mean value of the population and N is the number of values within the population.

Stochastic Effects are those for which the probability of occurrence is a function of dose, but the severity of the effects is independent of dose.

Stratified Sample (Stratified Random Sample) refers to a sample consisting of various portions that have been obtained from identified subparts or subcategories (strata) of the total lot or population. Within each category or stratum, the samples are taken randomly. The objective of taking stratified samples is to obtain a more representative sample than might be obtained by a completely random sampling.

Systematic Error is the condition in which there is a consistent deviation of the results from the actual or true values by a measurement process. The cause for the deviation, or bias, may be known or unknown; however, it is considered “assignable” (i.e., the cause can be reasonably determined).

Terrestrial Biota is plant and animal life living on or in land.

Variability is a general term for the dispersion of values in a data set.

Variance is a measure of the variability of samples within a subset or the entire population. Mathematically, the sample variance (s^2) is the sum of squares of the differences between the individual values of a set and the arithmetic average of the set, divided by one less than the number of values:

$$s^2 = \frac{\sum_{i=1}^{n} (X_i - \bar{X})^2}{n - 1}$$

where X_i = value of sample i;

$\bar{X}$ = average of samples measured; and

n = number of samples measured.

For a finite population, the variance (σ^2) is the sum of squares of deviations from the arithmetic mean, divided by the number of values in the population:

$$\sigma^2 = \frac{\sum_{i=1}^{N} (X_i - \mu)^2}{N}$$

where μ is the mean value of the population and N is the number of values within the population.

Acronyms and Abbreviations

λ_{bio}	biological decay constant
λ_{eff}	the combination of biological and radiological decay constants
λ_{rad}	radiological decay constant
ACRP	Advisory Committee on Radiation Protection
ASTM	American Society for Testing and Materials
B_{iv}	bioaccumulation factor
BCG	Biota Concentration Guide
BDAC	Biota Dose Assessment Committee
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CFR	Code of Federal Regulations
CV	coefficient of variation
D	absorbed dose
H	dose equivalent
DOE	U.S. Department of Energy
DQO	data quality objectives
EE/CA	engineering evaluation/cost analysis
EH	DOE's Office of Environment, Safety, and Health
EMS	Environmental Management System
EPA	U.S. Environmental Protection Agency
IAEA	International Atomic Energy Agency
ICRP	International Commission on Radiological Protection
K_d	solid/solution distribution coefficient
M&O	management and operating (contractor)

NCRP	National Council on Radiation Protection and Measurements
NEA	Nuclear Energy Agency
NEPA	National Environmental Policy Act
NIST	National Institute of Standards and Technology
NOAEL	No Observed Adverse Effects Levels
NRC	U.S. Nuclear Regulatory Commission
NRDA	Natural Resource Damage Assessment
PRA	population-relevant attribute
QA	quality assurance
QC	quality control
QF	quality factor
RBE	relative biological effectiveness
RCRA	Resource Conservation and Recovery Act
RI/FS	remedial investigation/feasibility study
UNSCEAR	United Nations Scientific Committee on the Effects of Atomic Radiation
USFWS	U.S. Fish and Wildlife Service
w_T	tissue or organ weighting factor

A Graded Approach for Evaluating Radiation Doses to Aquatic and Terrestrial Biota

MODULE 1

PRINCIPLES AND APPLICATION

MODULE 1: PRINCIPLES AND APPLICATION

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1 Introduction

The U.S. Department of Energy (DOE) is accountable to Congress and the public for the safe conduct of its activities, including facility operation, waste management and disposal activities, and remediation of environmental contamination. These routine activities may result in releases of radionuclides to the air and water, accumulation of radionuclides in soil and sediment, and the potential for plants, animals, and members of the public to be exposed to radiation. DOE Order 5400.5, “Radiation Protection of the Public and the Environment” (1990a), lists the environmental radiation protection requirements that DOE and DOE-contractor employees must meet to protect aquatic animals. In addition, dose limits below which deleterious effects on populations of aquatic and terrestrial organisms have not been observed, as discussed by the National Council on Radiation Protection and Measurements (NCRP 1991), and the International Atomic Energy Agency (IAEA 1992), are considered by DOE to be relevant to the protection of all aquatic and terrestrial biota on DOE sites.

1.1 Purpose

This DOE technical standard provides a graded approach (including screening methods and methods for detailed analyses) and related guidance that DOE and DOE contractors may use to evaluate compliance with specified limits on radiation dose to populations of aquatic animals, terrestrial plants, and terrestrial animals due to anthropogenic sources at DOE sites. Specifically, the technical standard provides dose evaluation methods that can be used to meet the requirements for protection of biota in DOE Orders 5400.1, "General Environmental Protection Program" (DOE 1990b), 5400.5 (DOE 1990a), and the dose limits for protection of biota developed or discussed by the NCRP (1991) and IAEA (1992). Accordingly, this technical standard uses the biota dose limits specified below within a graded approach to demonstrate that populations of plants and animals are adequately protected from the effects of ionizing radiation:

- **Aquatic Animals**. The absorbed dose to aquatic animals should not exceed 1 rad/d (10 mGy/d) from exposure to radiation or radioactive material releases into the aquatic environment. This dose limit is specified in DOE Order 5400.5.

- **Terrestrial Plants**. The absorbed dose to terrestrial plants should not exceed 1 rad/d (10 mGy/d) from exposure to radiation or radioactive material releases into the terrestrial environment.

- **Terrestrial Animals**. The absorbed dose to terrestrial animals should not exceed 0.1 rad/d (1 mGy/d) from exposure to radiation or radioactive material releases into the terrestrial environment.

Avoiding measurable impairment of reproductive capability is deemed to be the critical biological endpoint of concern in establishing the dose limits for aquatic and terrestrial biota. Module 1, Section 1.2.2 discusses this issue further. Guidance for interpreting and applying

these dose limits with respect to the length of time and geographic area over which actual doses should be compared with the limits is provided in Module 2, Section 3.

DOE has proposed these dose limits for aquatic and terrestrial biota under proposed rule Title 10, Code of Federal Regulations, Part 834 (10 CFR 834), “Radiation Protection of the Public and the Environment” *(*DOE 1993). DOE has decided not to promulgate these dose limits until guidance for demonstrating compliance has been developed. Consequently, this technical standard was developed, in part, in response to comments and recommendations received by DOE through the proposed rule comment period. Principal themes in the comments included: (1) requests for development of cost-effective methods to support the use of DOE's existing and proposed biota dose limits, (2) support for a multi-tiered approach to include screening, (3) requests for guidance on biota monitoring, and (4) requests for development of a generic method to promote consistency, while retaining some flexibility for site-specific methods and information. These themes served as the guiding principles for development of the methods contained in this technical standard.

The specific methods and guidance in this technical standard are acceptable for use by DOE and DOE-contractors when evaluating doses to biota in relation to the above dose limits. The methods and guidance in this technical standard should also be useful to ecological risk assessors who must evaluate risks to biota from radionuclides that occur on DOE sites. Using the graded approach provided in this technical standard, risk assessors can use soil, sediment, and water radionuclide concentration data to determine whether radionuclide concentrations at a site are likely to result in doses in excess of those listed above and would, therefore, have the potential to impact resident populations of plants and animals. The methods can also give risk assessors an immediate qualitative assessment of the importance of doses of ionizing radiation to the resident receptors. The dose equations in this technical standard also provide methods of estimating upper-bound (e.g., conservatively derived) doses to specific plants and animals. Refer to Module 1, Section 3, for a description of intended and potential applications of the DOE graded approach.

1.2 Background

1.2.1 Increasing Interest and Need for Biota Dose Evaluation Methods

There is growing national and international interest in establishing a regulatory framework (e.g., to include standards or criteria) and supporting evaluation methodologies for demonstrating protection of the environment from the effects of ionizing radiation. Regarding environmental protection, the ICRP statement that "...if man is adequately protected then other living things are also likely to be sufficiently protected" (ICRP 1977; 1991) uses human protection to infer environmental protection from the effects of ionizing radiation. This assumption is most appropriate in cases where humans and other biota inhabit the same environment and have common routes of exposure, and less appropriate in cases where human access is restricted or pathways exist that are much more important for biota than for humans. The inclusion of

radiation as a stressor within ecological risk assessments is also a consideration. Ecological risk assessments at contaminated sites being considered for remediation under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) generally require an assessment of all stressors, including radiation. Assessments of radiation impacts on contaminated ecosystems are currently underway in the U.S. under CERCLA regulations (EPA 1988).

Nationally and internationally, no standardized methods have been adopted for evaluating doses and demonstrating protection of plants and animals from the effects of ionizing radiation. In 1999, the IAEA convened a technical committee examining protection of the environment from the effects of ionizing radiation and provided recommendations and discussion points for moving forward with the development of protection frameworks and dose assessment methods. The resulting IAEA Technical Document, "Protection of the Environment from the Effects of Ionizing Radiation" (1999) references multi-tiered screening as a potentially cost-effective and easy way of demonstrating compliance with radiation criteria for protection of biota. The IAEA has subsequently hosted a series of Specialists' Meetings on radiological protection of the environment, and the Nuclear Energy Agency (NEA) and the ICRP have sponsored a series of fora on this issue. It is hoped that the methods and guidance provided in this DOE technical standard will serve as a platform for national and international discussion of radiation protection frameworks, standards, and dose assessment methods for biota.

Benefits of a Screening Process

"A multi-tiered screening approach is normally used in ecological risk assessments. Screening may also be a potentially cost-effective and easy way of demonstrating compliance with radiation criteria or standards for protection of the environment. Screening values should be used to identify radionuclides in situations of concern, and to determine whether these radionuclides warrant further assessment, or if they are at levels that require no further attention. In practice, this initial screening is expected to be sufficient in the majority of cases. When initial screening fails, additional analysis or assessment may be needed. A two- or three-tiered scheme would help ensure that the magnitude of the assessment effort would be scaled to the likelihood and severity of environmental impacts."

From: IAEA-TECDOC-1091, Protection of the Environment from the Effects of Ionizing Radiation: A Report for Discussion (July 1999)

1.2.2 Basis for Biota Dose Limits Applied in this Technical Standard

A dose limit for controlling radiological impacts from DOE activities to native aquatic animals is specified in DOE Order 5400.5. At present, DOE Orders do not specify dose limits for terrestrial organisms. However, an intended objective of DOE Orders 5400.1 and 5400.5 is to protect the aquatic and terrestrial environment, including populations of plants and animals, within and beyond the boundaries of DOE sites from impacts of routine DOE activities. The dose limits in this technical standard are consistent with (a) the intent of DOE Orders 5400.1

and 5400.5, (b) the dose limit for aquatic animals specified in DOE Order 5400.5, and (c) findings of the IAEA and NCRP regarding doses below which deleterious effects on populations of aquatic and terrestrial organisms have not been observed. They are also consistent with the intent of the IAEA document, "The Principles of Radioactive Waste Management" (IAEA 1995), in which Principle 2 states that "radioactive waste shall be managed in such a way as to provide an acceptable level of environmental protection." The background for the dose limits for aquatic and terrestrial biota is briefly discussed below. These dose limits represent expected safe levels of exposure, and are consensus No Adverse Effects Levels (NOAELs) for effects on population-relevant attributes in natural populations of biota.

1.2.2.1 Aquatic Organisms

At the request of DOE, the NCRP (1991) reviewed the literature on the effects of radiation on aquatic organisms and prepared a report on the then-current understanding of such effects. The report also provided guidance for protecting populations of aquatic organisms, concluding that a chronic dose of no greater than 1 rad/d (0.4 mGy/h) to the maximally exposed individual in a population of aquatic organisms would ensure protection of the population.

The IAEA examined and summarized the conclusions regarding aquatic organisms of several previous reviews (IAEA 1992):

- Aquatic organisms are no more sensitive than other organisms; however, because they are poikilothermic animals, temperature can control the time of expression of radiation effects.

- The radiosensitivity of aquatic organisms increases with increasing complexity, that is, as organisms occupy successively higher positions on the phylogenetic scale.

- The radiosensitivity of many aquatic organisms changes with age, or, in the case of unhatched eggs, with the stage of development.

- Embryo development in fish and the process of gametogenesis appear to be the most radiosensitive stages of all aquatic organisms tested.

- The radiation-induced mutation rate for aquatic organisms appears to be between that for *Drosophila* (fruit flies) and mice.

Furthermore, the 1992 review found that the conclusions of an earlier IAEA review (1976) were still supported; namely, that appreciable effects in aquatic populations would not be expected at doses lower than 1 rad/d (10 mGy/d) and that limiting the dose to the maximally exposed individuals to less than 1 rad/d would provide adequate protection of the population.

1.2.2.2 Terrestrial Organisms

The IAEA (1992) summarized information about the effects of acute ionizing radiation on terrestrial organisms as follows:

- Reproduction (encompassing the processes from gametic formation through embryonic development) is likely to be the most limiting endpoint in terms of survival of the population.

- Lethal doses vary widely among different species, with birds, mammals, and a few tree species being the most sensitive among those considered.

- Acute doses of 10 rad (100 mGy) or less are very unlikely to produce persistent and measurable deleterious changes in populations or communities of terrestrial plants or animals.

The IAEA (1992) also summarized information about the effects of chronic radiation on terrestrial organisms:

- Reproduction (encompassing the processes from gametogenesis through embryonic development) is likely to be the most limiting endpoint in terms of population maintenance.

- Sensitivity to chronic radiation varies markedly among different taxa; certain mammals, birds, reptiles, and a few tree species appear to be the most sensitive.

- In the case of invertebrates, indirect responses to radiation-induced changes in vegetation appear more critical than direct effects.

- Irradiation at chronic dose rates of 1 rad/d (10 mGy/d) or less does not appear likely to cause observable changes in terrestrial plant populations.

- Irradiation at chronic dose rates of 0.1 rad/d (1 mGy/d) or less does not appear likely to cause observable changes in terrestrial animal populations. The assumed threshold for effects in terrestrial animals is less than that for terrestrial plants, primarily because some species of mammals and reptiles are considered to be more radiosensitive.

- Reproductive effects on long-lived species with low reproductive capacity may require further consideration.

The NCRP and IAEA concluded for aquatic organisms and the IAEA concluded for terrestrial organisms that the statement by the ICRP (1977; 1991), "...if man is adequately protected, then other living things are also likely to be sufficiently protected" was reasonable within the

limitations of the generic exposure scenarios examined. A similar assessment was made at a DOE-sponsored workshop (Barnthouse 1995) held to evaluate the adequacy of existing effects data and approaches to radiation protection of aquatic and terrestrial organisms to support moving forward with setting regulatory limits. DOE workshop participants agreed that protecting humans generally protects biota, except under the following conditions: (1) human access to a contaminated area is restricted but access by biota is not restricted, (2) unique exposure pathways exist for plants and animals that do not affect exposure of humans, (3) rare or endangered species are present, or (4) other stresses on the plant or animal population are significant.

1.2.2.3 Additional Summaries and Reviews of Radiation Effects Data on Biota Confirming NCRP and IAEA Findings

UNSCEAR. In 1996, the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) summarized and reviewed information on the responses to acute and chronic radiation of plants and animals, both as individuals and as populations (UNSCEAR 1996). The conclusions from the UNSCEAR review were consistent with findings and recommendations made earlier by the NCRP and IAEA concerning biota effects data and appropriate dose limits for protection of biota. In 2002, UNSCEAR reported that these dose rate criteria (1 rad/d for aquatic animals and terrestrial plants; 0.1 rad/d for terrestrial animals) remain defensible for protection of populations of plants and animals. The UNSCEAR plans to develop a new scientific annex to further address radioecology and effects of radiation on the environment (Gentner 2002).

UK Environment Agency. In 2001, the Environment Agency of the United Kingdom (UK) conducted a review of the available body of radiation effects data on biota (Copplestone et al. 2001). They concluded that it is unlikely that there will be any significant effects in:

- populations of freshwater and coastal organisms at chronic dose rates below 400 uGy/h (or 1 rad/d; 10 mGy/d);

- terrestrial plant populations at chronic dose rates below 400 uGy/h (or 1 rad/d; 10 mGy/d); and

- terrestrial animal populations at chronic dose rates below 40 uGy/h (or 0.1 rad/d; 1 mGy/d).

It is noteworthy that the UK Environment Agency's review findings are largely consistent with the findings and biota dose recommendations of the NCRP, the IAEA, and UNSCEAR cited above. Additionally, they concluded that it is unlikely that there will be any significant effects in populations of organisms in the deep ocean at chronic dose rates below 1,000 uGy/h (or 2.5 rad/d; 25 mGy/d).

ACRP. In 2002, the Advisory Committee on Radiation Protection (ACRP), charged with providing advice to the Canadian Nuclear Safety Commission (CNSC) regarding approaches needed for the radiological protection of the environment, provided recommendations concerning appropriate dose rate criteria for protection of biota. The ACRP recommended that the generic dose rate criterion for protecting biota should be in the range of 1-10 mGy/d (0.1-1 rad/d). The ACRP indicated that this dose rate criterion is based on population-level effects and, given the current state of knowledge and consensus views of radiation effects on biota, represents the level at which ecosystems will suffer no appreciable deleterious effects. The criterion is specified in terms of daily dose rather than annual dose. The intent is to avoid, for example, what would be the annual dose at this dose rate criterion being received in a few days. The ACRP further recommended that there should be some flexibility in the averaging time used in interpreting this dose rate criterion (CNSC-ACRP 2002).

1.2.2.4 Application of Biota Dose Limits as "Dose Rate Guidelines" for Evaluating Doses to Biota

The biota dose limits specified in this technical standard are based on the current state of science and knowledge regarding effects of ionizing radiation on plants and animals. They should not be interpreted as a "bright line" that, if exceeded, would trigger a mandatory regulatory or remedial action. Rather, they should be interpreted and applied more as "Dose Rate Guidelines" that provide an indication that populations of plants and animals could be impacted from exposure to ionizing radiation and that further investigation and action is likely necessary.

1.2.3 Protection of Populations

The intent of the graded approach (i.e., the screening and analysis methods) in this technical standard is to protect populations of aquatic animals, terrestrial animals, and terrestrial plants from the effects of exposure to anthropogenic ionizing radiation. As noted above, certain taxa are more sensitive to ionizing radiation than others. Based on this observation, it is generally assumed that protecting the more sensitive taxa will adequately protect other, less sensitive taxa. Hence, in cases where site-specific evaluations may be required, receptors should be selected that (1) are important to the structure and function of the community, (2) are expected to receive a comparatively high degree of exposure (e.g., expected to receive a radiation dose to reproductive tissues which is relatively high per unit of radionuclide present in the ecosystem, in comparison with other receptors in the same community), and (3) have a comparatively high degree of radiosensitivity (e.g., radiation effects of concern occur at relatively low doses, in comparison with other receptors in the same community). Figure 1.1 shows the relative radiosensitivity of various taxa for both aquatic and terrestrial systems.

Participants at the DOE-sponsored workshop to evaluate the adequacy of existing effects data and approaches to radiation protection of aquatic and terrestrial organisms (Barnthouse 1995) concluded that existing data support the application of recommended dose limits to

representative rather than maximally exposed individuals within populations of plants and animals. Participants concluded that exposure below the recommended dose limits would not cause adverse effects at the population level, even though some individuals within the population might be adversely affected.

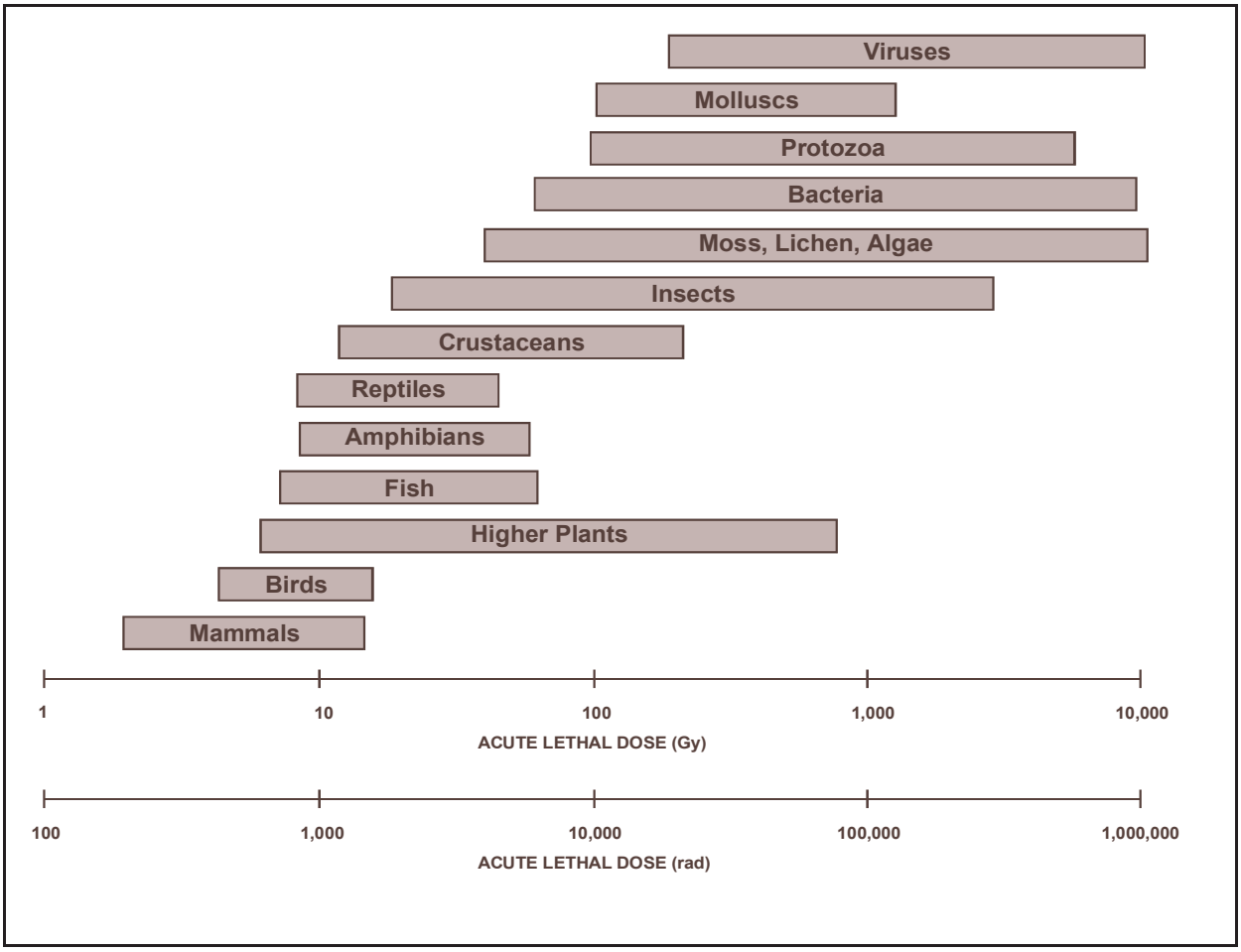


Figure 1.1 Approximate Acute Lethal Dose Ranges for Various Taxonomic Groups
Source: Whicker and Schulz 1982; UNSCEAR 1996.

2 Overview of the DOE Graded Approach

DOE's graded approach for evaluating radiation doses to aquatic and terrestrial biota consists of a three-step process which is designed to guide a user from an initial, conservative general screening to, if needed, a more rigorous analysis using site-specific information (Figure 2.1). The three-step process includes: (1) assembling radionuclide concentration data and knowledge of sources, receptors, and routes of exposure for the area to be evaluated; (2) applying an easy-to-use general screening methodology that provides limiting radionuclide concentration values (i.e., Biota Concentration Guides - BCGs) in soil, sediment, and water; and (3) if needed, conducting an analysis through site-specific screening, site-specific analysis, or an actual site-specific biota dose assessment conducted within an eco-risk. Any of the steps within the graded approach may be used at any time, but the general screening methodology will usually be the simplest, most cost-effective, and least time-consuming. Table 2.1 provides a summary of DOE's graded approach.

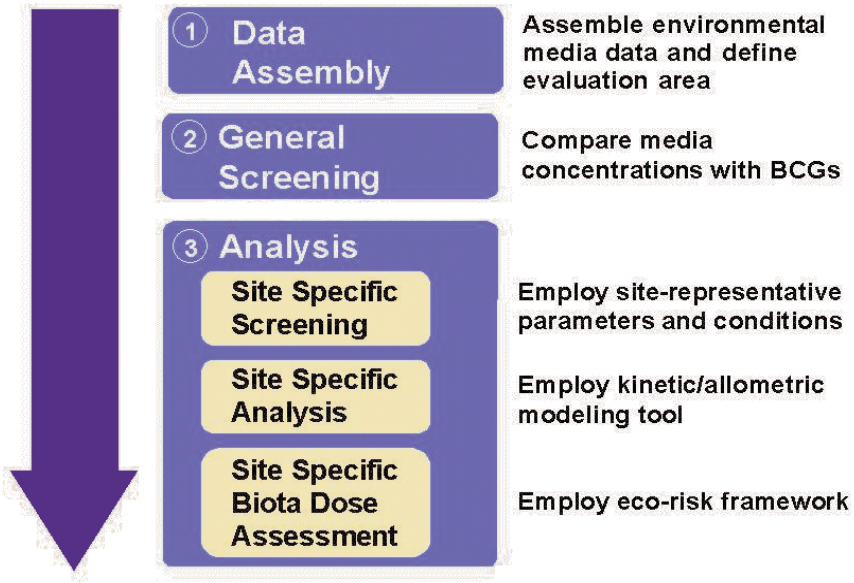


Figure 2.1 Overview of the DOE Graded Approach for Evaluating Radiation Doses to Aquatic and Terrestrial Biota

Table 2.1 Summary of DOE's Three-Step Process for Evaluating Radiation Doses to Aquatic and Terrestrial Biota

1. Data Assembly	Knowledge of sources, receptors, and routes of exposure for the area to be evaluated is summarized. Measured radionuclide concentrations in water, sediment, and soil are assembled for subsequent screening.
2. General Screening	Maximum measured radionuclide concentrations in an environmental medium (i.e., water, sediment, soil) are compared with a set of Biota Concentration Guides (BCGs). Each radionuclide-specific BCG represents the limiting radionuclide concentration in an environmental medium which would not result in recommended dose standards for biota to be exceeded.
3. Analysis	This phase consists of three increasingly more detailed steps of analysis.
(a) Site-Specific Screening	Site-specific screening, using more realistic site-representative lumped parameters (e.g., bioaccumulation factors) in place of conservative default parameters. Use of mean radionuclide concentrations in place of maximum values, taking into account time dependence and spatial extent of contamination, may be considered.
(b) Site-Specific Analysis	Site-specific analysis employing a kinetic modeling tool (applicable to riparian and terrestrial animal organism types) provided as part of the graded approach methodology. Multiple parameters which represent contributions to the organism's internal dose (e.g., body mass, consumption rate of food/soil, inhalation rate, lifespan, biological elimination rates) can be modified to represent site and organism-specific characteristics. The kinetic model employs allometric equations relating body mass to these internal dose parameters.
(c) Site-Specific Biota Dose Assessment	An actual site-specific biota dose assessment involving the collection and analysis of biota samples. The dose assessment would involve a problem formulation, analysis, and risk characterization protocol consistent with the widely-used ecological risk assessment paradigm.

2.1 Key Features of the Graded Approach

The graded approach was designed for flexibility and acceptability:

- It provides users with a tiered approach for demonstrating compliance with biota dose limits that is generally cost-effective and easy-to-implement.

- It allows for the use of measured radionuclide concentrations in environmental media typically collected as part of routine environmental surveillance programs.

- It is designed for multiple applications. The technical standard is applicable to demonstrations of compliance with biota dose limits and for use in ecological risk assessments of radiological impact.

- It provides a framework that supports the use of site-specific information.

- It incorporates ecological risk assessment concepts and provides guidance for site-specific biota dose assessments (where needed) employing the widely-used ecological risk assessment (ERA) paradigm.

- All of the equations and resulting BCGs contained in this technical standard have been encoded into a series of electronic spreadsheets. The spreadsheets were built using Microsoft Excel® and incorporate Visual Basic® commands to help guide and automate the user's progression through the biota dose evaluation process. Use of these spreadsheets, termed the "RAD-BCG Calculator," is described in Module 1, Sections 4-8. Refer to Module 1, Section 4 for an overview of the RAD-BCG Calculator and its contents for use as a companion tool to this technical standard.

- It provides users with "a place to start" and "an analysis path forward." The BCGs are not stand-alone. Exceedance of BCGs leads the user to the more-detailed tiers of analysis as needed in a stepwise manner. These linkages are an integral part of the graded approach framework and are built into the companion software tool, the RAD-BCG Calculator.

2.2 Key Points Regarding Methods Derivation

Internal and external sources of dose (and their contributing exposure pathways) are incorporated in the derivation of the graded approach methodology. Sufficient prudence has been exercised in the development of each of the assumptions and default parameter values to ensure that the resulting BCGs are appropriately conservative. In the event that an individual default parameter value is subsequently found to be an upper-end value but not the "most limiting" value for a unique site-specific exposure scenario, the other prudent assumptions and default parameter values will ensure that the BCGs (and resultant doses to biota) should continue to carry the appropriate degree of conservatism for screening purposes. Refer to Module 3 for a detailed description of the derivation of dose equations and default parameters used in the graded approach. Key assumptions used in deriving the BCGs that highlight the conservatism applied in the general screening phase are presented in Table 2.2. Exposure pathways for each of the reference organism types considered in the graded approach are presented in Figures 2.2 through 2.5. A summary of the general dose equation and approach used to derive the BCGs is provided in Table 2.3.

Table 2.2 Assumptions Regarding Sources, Receptors, and Routes of Exposure Applied in the General Screening Phase of the Graded Approach

Dose Limits	• BCGs were derived for aquatic animal, riparian animal, terrestrial plant, and terrestrial animal reference organisms. The dose rate limits used to derive the BCGs for each organism type are 1 rad/d, 0.1 rad/d, 1 rad/d, and 0.1 rad/d respectively. • While existing effects data support the application of these dose limits to representative individuals within populations of plants and animals, the assumptions and parameters applied in the derivation of the BCGs are based on a maximally exposed individual, representing a conservative approach for screening purposes.
External Sources of Radiation Exposure	• Estimates of the contribution to dose from external radioactive material were made assuming that all of the ionizing radiation was deposited in the organism (i.e., no pass-through and no self-shielding). This is conservative, and is tantamount to assuming that the radiosensitive tissues of concern (the reproductive tissues) lie on the surface of a very small organism. • For external exposure to contaminated soil, the source was presumed to be infinite in extent. In the case of external exposure to contaminated sediment and water, the source was presumed to be semi-infinite in extent. • The source medium to which the organisms are continuously exposed is assumed to contain uniform concentrations of radionuclides. • These assumptions provide for appropriately conservative estimates of energy deposition in the organism from external sources of radiation exposure.
Internal Sources of Radiation Exposure	• Estimates of the contribution to dose from internal radioactive material were conservatively made assuming that all of the decay energy is retained in the tissue of the organism, (i.e., 100% absorption). • Progeny of radionuclides and their decay chains are also included. This provides an over-estimate of internal exposure, as the lifetime of many of the biota of interest is generally short compared to the time for the build-up of progeny for certain radionuclides. • The radionuclides are presumed to be homogeneously distributed in the tissues of the receptor organism. This is unlikely to under-estimate the actual dose to the tissues of concern (i.e., reproductive organs). • A radiation weighing factor of 20 for alpha particles is used in calculating the BCGs for all organism types. This is conservative, especially if non-stochastic effects are most important in determining harm to biota. The true value may be a factor of 3 to 4 lower.

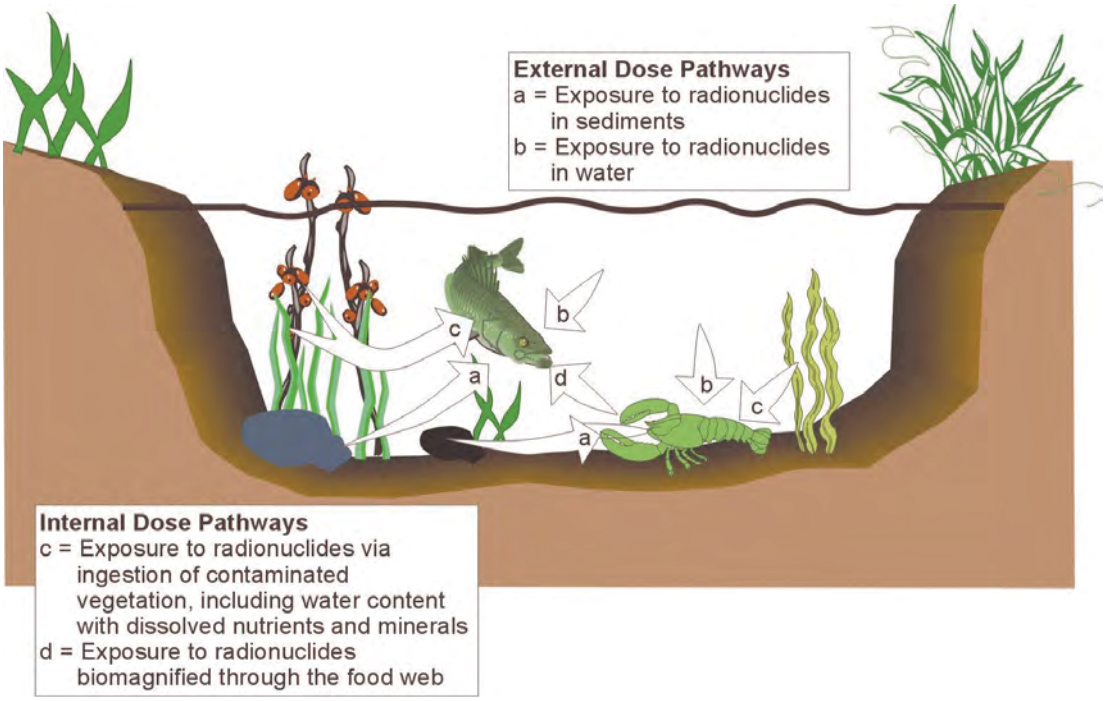


Figure 2.2 Exposure Pathways for Aquatic Animals

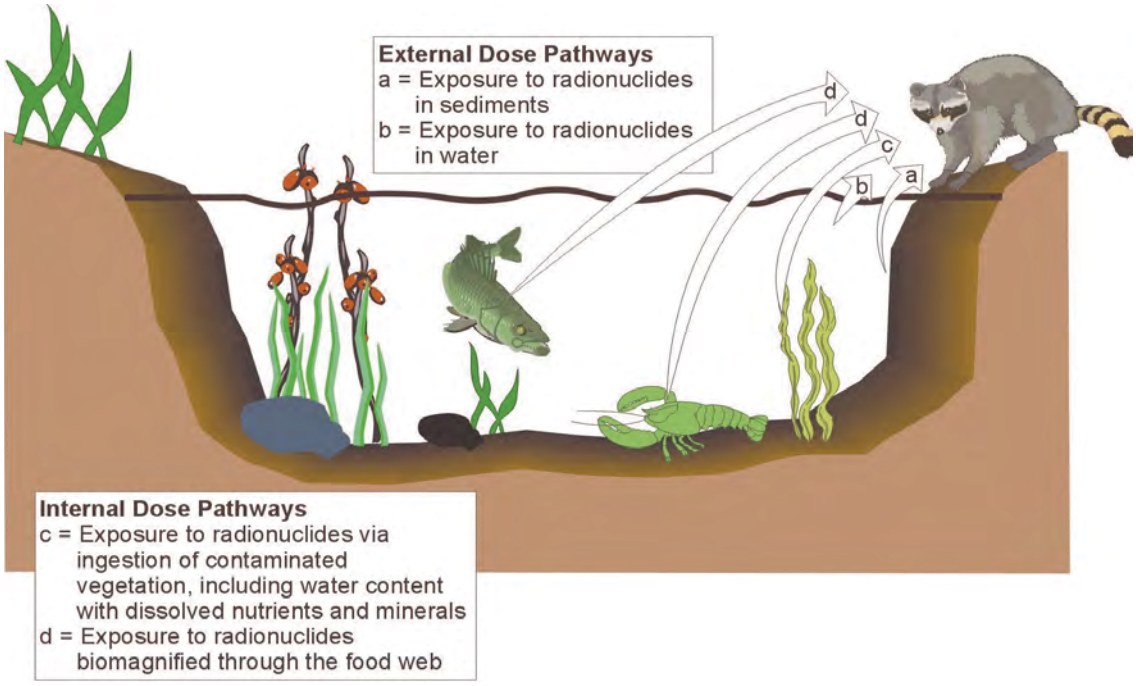


Figure 2.3 Exposure Pathways for Riparian Animals

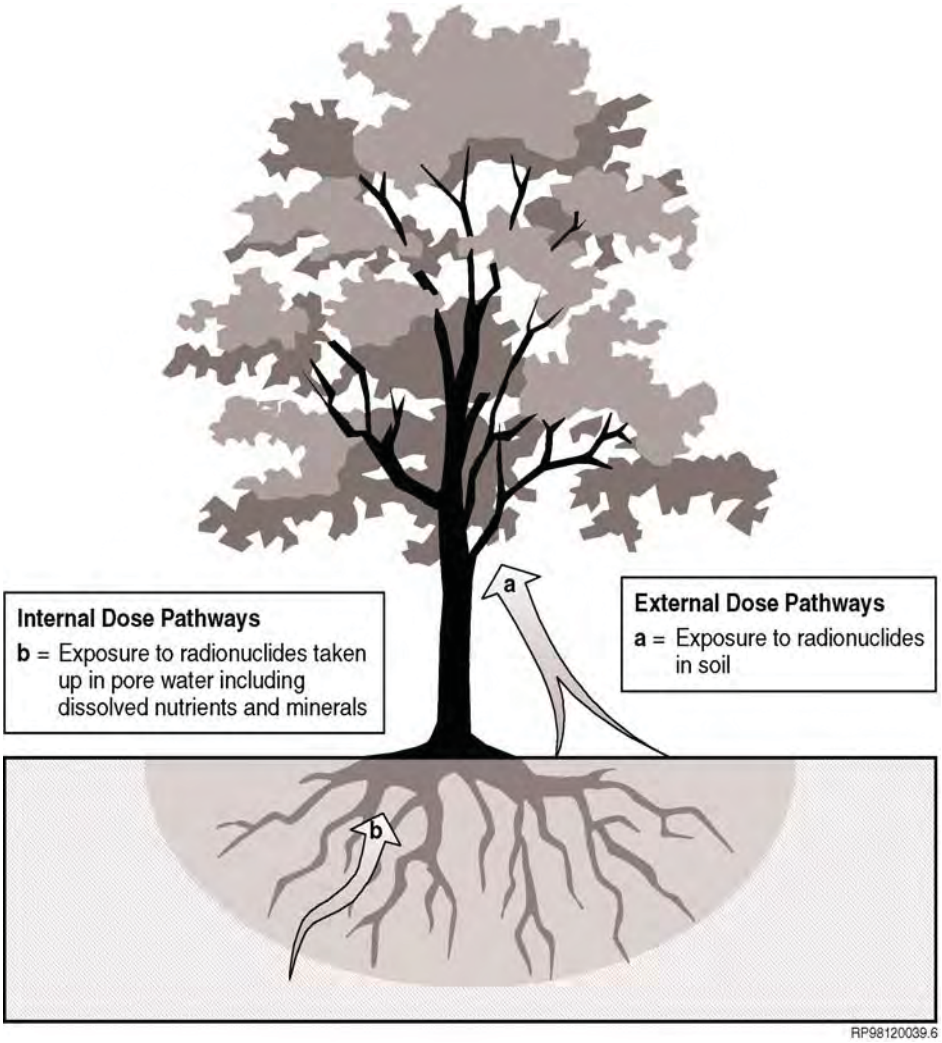


Figure 2.4 Exposure Pathways for Terrestrial Plants

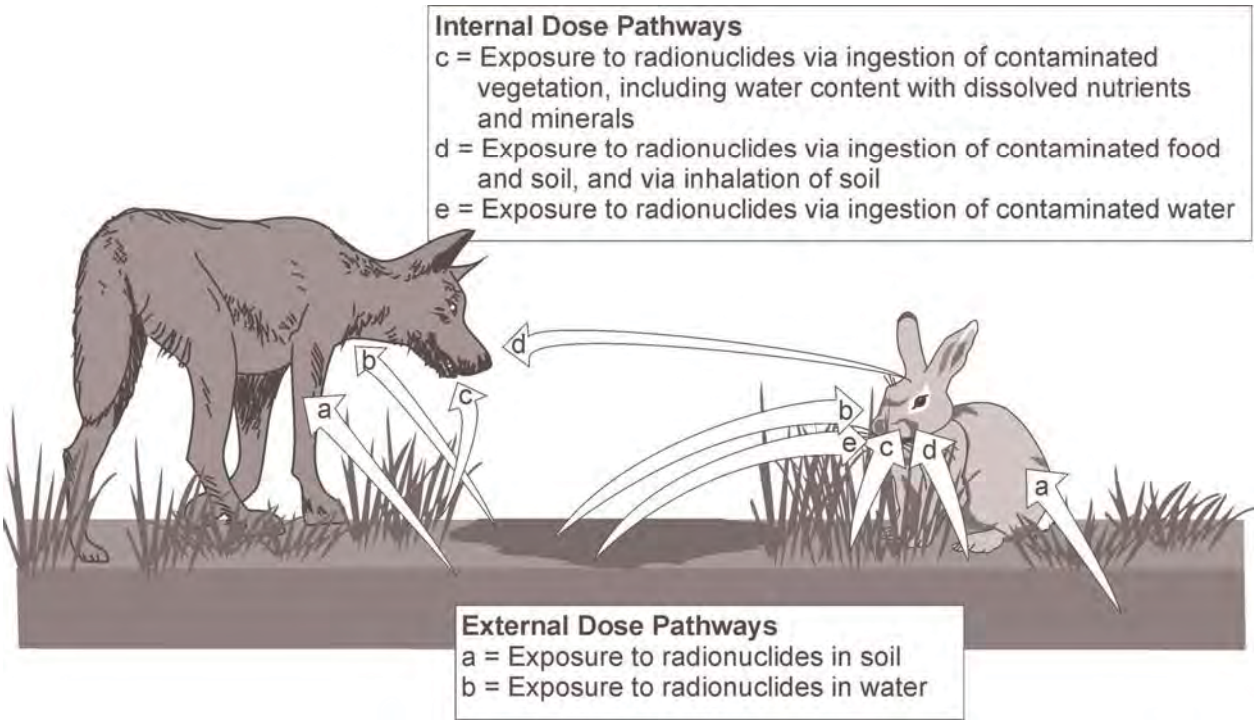


Figure 2.5 Exposure Pathways for Terrestrial Animals

Table 2.3 General Dose Equation and Approach Used to Derive BCGs

$$\text{Limiting Concentration} = \frac{\text{Dose Rate Limit}}{(\text{Internal Dose Rate}) + (\text{External Dose Rate}_{\text{soil/sed.}}) + (\text{External Dose Rate}_{\text{water}})}$$

- The limiting concentration in an environmental medium was calculated by first setting a target total dose (e.g., 1 rad/d for aquatic organisms and terrestrial plants, or 0.1 rad/d for riparian and terrestrial animals) and then back-calculating to the medium concentration (i.e., the BCG) necessary to produce the applicable dose from radionuclides in the organism (internal dose), plus the external dose components from radionuclides in the environment (external dose).
- The denominator of the generic equation represents the dose per unit media concentration and may be broken down into the base components of internal and external dose.
- Internal doses originate from radionuclides inside the organism's body. The internal dose is calculated as the product of the internal radionuclide concentration and internal dose conversion factor. External doses originate from radionuclides external to the organism and are calculated as the product of the radionuclide concentration in the environmental medium in which the organism resides and an appropriate dose conversion factor.

2.3 Relationship of the Graded Approach to Ecological Risk Assessment

The Graded Approach Is a Framework for Organizing Successively Rigorous Tiers of Assessment, with a Particular Emphasis on Ionizing Radiation.

The graded approach for evaluating radiation doses to aquatic and terrestrial biota is consistent with the standard ecological risk assessment (ERA) paradigm (EPA 1998). As in the standard ERA paradigm, the graded approach moves from a simple and relatively conservative screening evaluation to a more detailed and realistic assessment. Each step in the graded approach addresses, either explicitly or implicitly, the principal ERA components. That is, the graded approach is a framework for organizing the successively rigorous ERA tiers, but with a particular emphasis on ionizing radiation.

The graded approach for evaluating radiation doses to aquatic and terrestrial biota is consistent with the standard ecological risk assessment (ERA) paradigm (EPA 1998). The ERA structure provides a process for organizing and evaluating information to determine the nature, likelihood, and magnitude of potential impacts on environmental receptors (Suter 1993). The three major phases of an ERA are problem formulation, analysis of exposure and effects, and risk characterization. The ERA is typically done in successively rigorous tiers, each of which includes the three general ERA phases (Suter, Efroymson, Sample & Jones 2000). As in the widely-used ERA paradigm, the graded approach moves from a simple and relatively conservative screening evaluation to a more detailed and realistic assessment. Each step in the graded approach addresses, either explicitly or implicitly, all of the aforementioned ERA components. That is, the graded approach is a framework for organizing the successively rigorous ERA tiers, but with a particular emphasis on ionizing radiation.

The ERA process is general in nature and could be applied to the evaluation of radiation as a stressor, but not without some modifications and provision of additional guidance. There are some noteworthy technical issues concerning the evaluation of radiation that require further consideration and elaboration. Some issues are the same as for chemicals, but some are unique to radionuclides. In response to requests for guidance on this topic, Module 2, Section 1 provides a basic “primer” on technical issues that should be considered when evaluating radiation as a stressor to the environment, and draws on the experiences gained by BDAC members in developing the graded approach and conducting radiological ERAs. To our knowledge, standardized guidance on how to address these issues is not available elsewhere.

3 Application Considerations

The principal application of the graded approach is to demonstrate that routine DOE operations and activities are in compliance with the biota dose limits for protecting populations of plants and animals. In addition, the design of the graded approach (e.g., assumptions used; a multi-tiered screening and analysis approach; flexibility to allow use of site-specific information on sources, receptors, and routes of exposure) permits its application in ecological assessments of radiological impact and in other environmental assessment scenarios. Discussions on other intended or potential applications of the graded approach were first held in 1999 at a Biota Dose Assessment Committee (BDAC) Meeting (DOE 1999). Additional applications of the graded approach were identified by users and reviewers of an interim version of this technical standard that was made available for a trial use period beginning in July 2000 (DOE 2000a). Recommendations made by BDAC members and users on the intended and potential applications of the graded approach are summarized in an applications matrix (Table 3.1).

Data Quality Objectives

Data quality objectives (DQOs) shall be considered when determining the appropriateness of applying the DOE graded approach to other environmental assessment scenarios identified in Table 3.1.

Table 3.1 Applications Matrix Summarizing Intended and Potential Uses of the DOE Graded Approach

APPLICATIONS	INTENDED / POTENTIAL USE	CONSIDERATIONS
Types of Receptors		
Populations of plants and animals	This is the primary intended use.	
Individual plants and animals, including threatened and endangered species, and commercially or culturally valued species	Equations used within the graded approach are technically sound for application to individual organisms. Applying dose limits intended for the protection of populations to evaluations of individuals may require further consideration.	Use of effects endpoints/dose limits appropriate for protection of the individuals being evaluated; and/or application of safety factors, conservative exposure assumptions, and parameter values. Dose evaluations should be performed under the provisions of the applicable Federal and/or state statutes or regulations for rare and endangered species.

Table 3.1 (Continued) Applications Matrix Summarizing Intended and Potential Uses of the DOE Graded Approach

APPLICATIONS	INTENDED / POTENTIAL USE	CONSIDERATIONS
Types of Exposure		
Chronic	The methodology assumes chronic exposure and equilibrium conditions.	
Acute		The methodology is not intended to be used for assessing acute exposures. The models and assumptions used in the graded approach assume equilibrium conditions.
Accidents	Could be used to provide an indication of long-term "recovery" or health of the population over time following an accident. Equations and models used within the graded approach are technically sound for this application.	Accidents typically result in short-term, acute exposures for which the methodology is not intended. However, it can be applied for assessing long-term exposures due to accidents.
Types of Environments		
Fresh water, coastal, and marine environments	The methodology is intended to be applied to fresh water environments, and can be applied to coastal and marine environments.	Care must be taken when selecting parameter values (e.g., receptor lumped parameters; K_d values), as fresh water, coastal, and marine equilibrium chemistry differ considerably.
Terrestrial environments	The methodology is intended to be applied to terrestrial environments.	
Compliance / Impact Assessment		
Demonstration that DOE activities are in compliance with biota dose limits	This is a principal DOE application of the graded approach.	

Table 3.1 (Continued) Applications Matrix Summarizing Intended and Potential Uses of the DOE Graded Approach

APPLICATIONS	INTENDED / POTENTIAL USE	CONSIDERATIONS
Compliance / Impact Assessment (Continued)		
National Environmental Policy Act (NEPA)	The graded approach could be coupled with predictive dispersion codes that model a facility's effluents prior to construction, to estimate doses to biota in the Environmental Impact Statement. •• Comparison of alternatives •• Screen for issues needing analysis •• Defining significance criteria •• Mitigation action plan	Effects and assessment endpoints selected for use in the biota dose evaluation should be relevant to the management goals of the study.
Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA)	Screening for potential radiological impacts within an ecological risk assessment. •• Remedial Investigation/ Feasibility Study (RI/FS) •• Engineering Evaluation/ Cost Analysis (EE/CA)	Effects and assessment endpoints selected for use in the biota dose evaluation should be relevant to the management goals of the study.
Natural Resource Damage Assessments (NRDA)	Screening assessments.	Effects and assessment endpoints selected for use in the biota dose evaluation should be relevant to the management goals of the study.
Decommissioning	Could be used to evaluate doses to biota, and to predict future doses to biota, associated with pre- and post- site or facility decommissioning activities.	Effects and assessment endpoints selected for use in the biota dose evaluation should be relevant to the management goals of the study.
Resource Conservation and Recovery Act (RCRA)	•• Mixing zone definition •• Alternative concentration limits	Effects and assessment endpoints selected for use in the biota dose evaluation should be relevant to the management goals of the study.

Table 3.1 (Continued) Applications Matrix Summarizing Intended and Potential Uses of the DOE Graded Approach

APPLICATIONS	INTENDED / POTENTIAL USE	CONSIDERATIONS
Compliance / Impact Assessment (Continued)		
Clean Water Act	Mixing zone assessments.	Effects and assessment endpoints selected for use in the biota dose evaluation should be relevant to the management goals of the study.

As mentioned earlier, the principal driver and basis of need for developing the graded approach was to provide DOE field and program elements with methods for demonstrating compliance with DOE biota dose limits and recommendations for radiological protection of the environment. Thus, many of the decisions that are traditionally made when conducting a case-specific assessment (e.g., choice of indicator receptors; defining receptor exposure profiles; selection of effects endpoints) were made at a programmatic level and incorporated into the screening phase of the graded approach *a priori.* For example, the thresholds for adverse effects were set at the recommended limits for protection of natural populations of biota. Those are the appropriate effects levels for demonstrating protection with DOE requirements and recommendations for the protection of the environment from ionizing radiation (Module 1, Section 1.2). If the graded approach is used for other purposes (e.g., Table 3.1), then the programmatic objectives and the methods should be reviewed and discussed with the relevant decision makers and stakeholders, preferably via the Data Quality Objectives (DQO) process (Bilyard et al. 1997) to ensure that the results obtained through application of the graded approach will support the management goals and objectives of the environmental assessment.

3.1 Evaluating Doses to Individual Organisms

The equations and models used within the graded approach for estimating the dose per unit concentration of radionuclides in environmental media and for deriving the BCGs are also applicable to individual organisms. However, there are questions concerning the applicability of the biota dose limits to individual organisms. While the biota dose limits presented in Module 1, Section 1.1 were derived based on dose-response information for the most radiosensitive of all species studied, and taking into account the most radiosensitive life stages, the question of whether these dose limits can be applied to protection of individual members of a species, in contrast to protection of populations of species, requires further consideration. That is, for individual plants and animals, especially threatened and endangered species, the health effects of concern could be different from the effects of concern in protection of populations.

The application of safety factors to these dose limits is one approach that has been used in evaluating doses to individual organisms (e.g., for culturally valued species). Use of safety

factors, appropriate default parameter values, maximum radionuclide concentrations in environmental media, and 100 percent organism residence time and exposure are factors to consider in the application of the graded approach for evaluating doses to individuals. Refer to Module 2, Section 8 for a more detailed discussion on this issue. Specific cases where evaluation of individual organisms may be needed are discussed below.

3.1.1 Threatened and Endangered Species

Care must be taken by the user if the graded approach is applied in an evaluation of potential radiological impacts to endangered, threatened, rare, or otherwise sensitive species of plants and animals managed under the Federal Endangered Species Act or similar state laws or regulations pertaining to rare or endangered species (Endangered Species Act, 16 USC 1531 et seq.). It is the users responsibility to select effects and assessment endpoints, and the required input parameter values that reflect actual or expected exposure profiles, for the individuals being evaluated. Protection of endangered species should be performed under the provisions of the applicable Federal and/or state statutes or regulations for rare and endangered species.

3.1.2 Commercially and Culturally Valued Species

Care must be taken by the user if the graded approach is applied in an evaluation of potential radiological imacts to these categories of species. These would include species that are routinely harvested for their economic value (e.g., salmon) or their cultural value (e.g., medicinal plants used by Native Americans). One issue is whether or not these species should be evaluated at the individual or the population level. It is the users responsibility to select effects and assessment endpoints, and the required input parameter values that reflect actual or expected exposure profiles, for the individuals being evaluated.

3.2 Evaluating Doses to Aquatic Plants

Available information about the effects of ionizing radiation on aquatic plants does not appear to be adequate to characterize their sensitivity to ionizing radiation, or to establish defensible recommendations (i.e., in the form of dose standards or criteria) for allowable exposures of populations or individuals. However, regarding this technical standard, indirect means can provide a general qualitative indication of the effects to aquatic plants relative to effects on other organisms. In general, one would expect substantially lower radiosensitivity in higher plants in comparison to the most sensitive birds, fishes and mammals (Whicker and Schultz 1982; Whicker 1997). Therefore, an evaluation using this technical standard that demonstrates protection of aquatic and riparian animals should provide an indication that aquatic plants are also likely protected. Alternatively, appropriate bioaccumulation factors (B_{iv}s) for aquatic plants could be used in the appropriate aquatic system spreadsheets to calculate BCGs for aquatic plants. Refer to Module 2, Section 2.3, and Module 3, Section 3.2.1, for guidance in this area.

3.3 Experimental Facilities

The methods in this technical standard are not directly intended to be applied to properly permitted experimental facilities that expose biota to ionizing radiation without releasing materials to the environment (e.g., particle beam accelerators). Although the operation of such facilities may be considered to be "routine," any inadvertent exposure of biota as a result of such operations should have been addressed in the operating permit, precluding any need to apply the methods described herein. Additionally, any such exposures would be localized, and would thus be unlikely to affect substantial populations of any species that this technical standard addresses. Refer to Module 2, Section 2.4 for detailed considerations and methods for evaluating potential impacts to biota around accelerators or other sources of direct radiation.

3.4 Hazardous Chemicals and Industrial Hazards

The methods in this technical standard are not appropriate for evaluating potential impacts on biota from hazardous chemicals or industrial-type hazards, including noise and traffic.

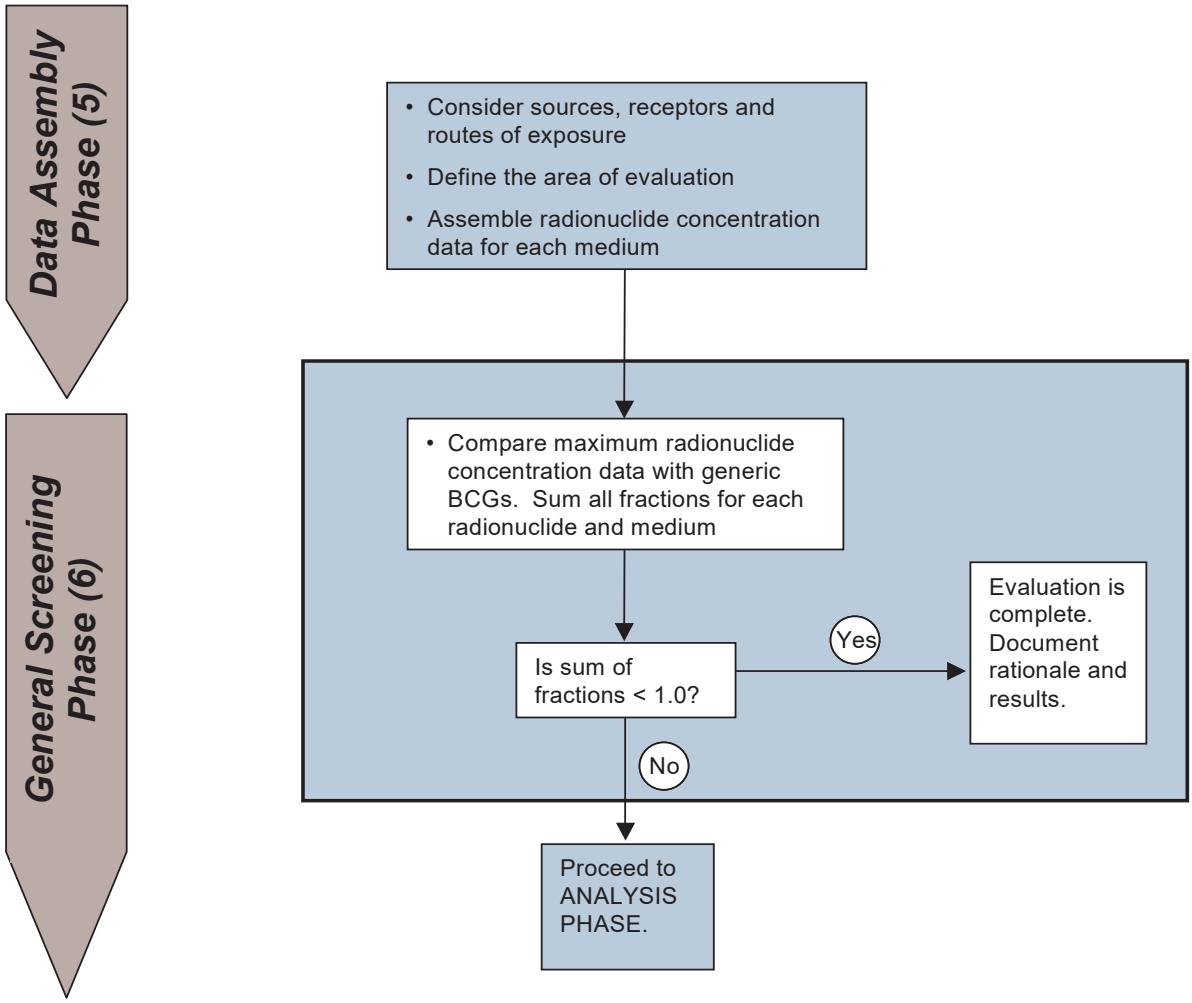
3.5 Frequency of Conducting Evaluations

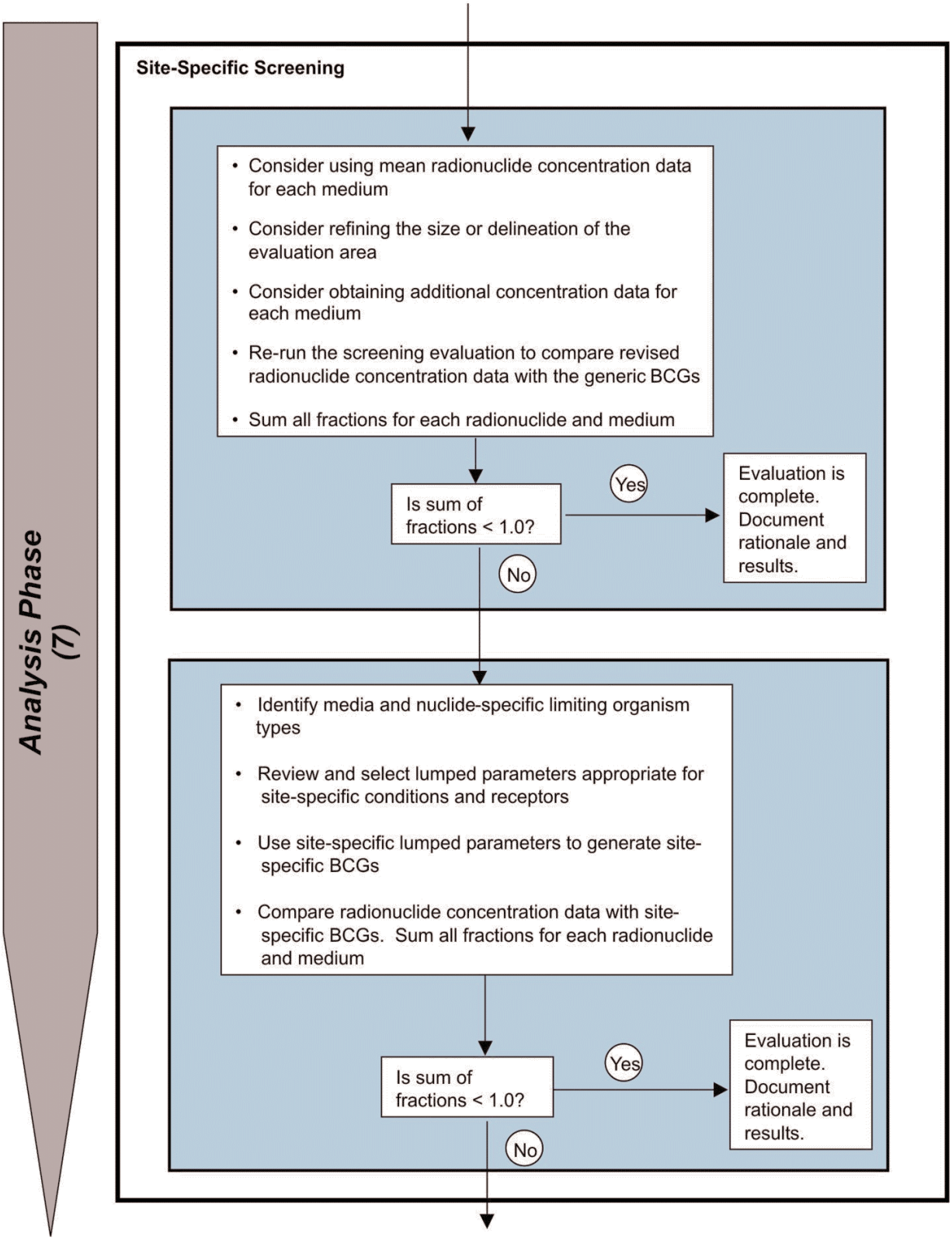
Dose evaluations for aquatic and terrestrial biota shall be conducted annually in conjunction with the preparation of annual site environmental reports that are required under DOE Orders 5400.1 and 5400.5. More frequent evaluations could be required at the direction of DOE's Office of Environment, Safety and Health (EH).

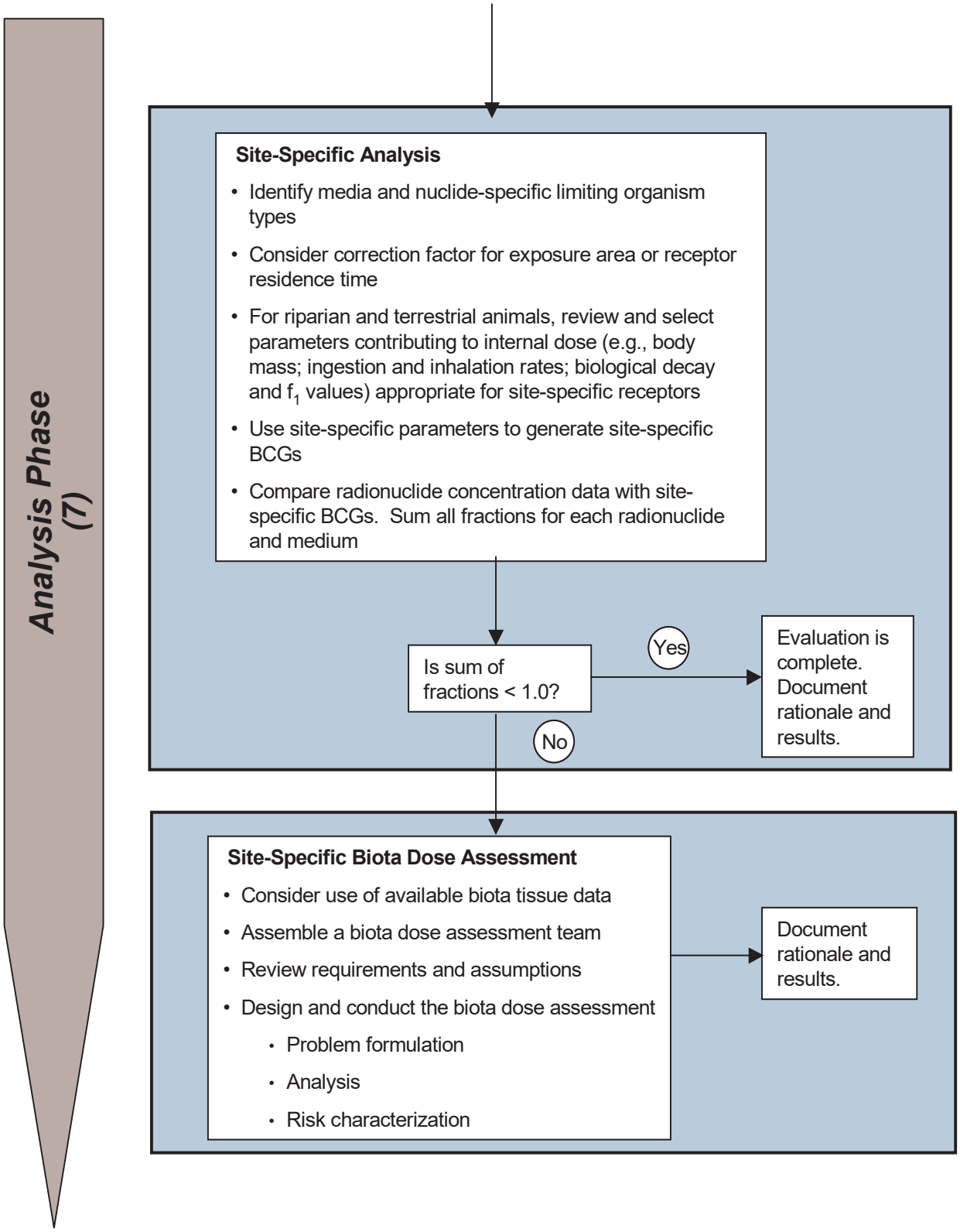
4 Step-by-Step Implementation of the Graded Approach

Here we present an overview of the complete process for implementing the graded approach. This section is provided to help orient you to the step-by-step guidance corresponding to each phase of the graded approach which follows in Sections 5 - 8 of this Module. A flowchart showing how to progress through each phase of the graded approach, and the components of each phase, is provided in Figure 4.1. Refer to this figure as you proceed through the step-by-step guidance presented in subsequent sections. References to more comprehensive guidance (presented in Module 2 of this technical standard) are provided throughout the step-by-step guidance. Example applications of the graded approach, using actual DOE site data, are presented in Section 9 of this Module.

Figure 4.1 Flowchart Illustrating Step-by-Step Guidance for Progressing Through the DOE Graded Approach. Section numbers within this technical standard corresponding to each phase are highlighted for reference.




Analysis Phase
(7)
Site-Specific Screening
• Consider using mean radionuclide concentration data for each medium
• Consider refining the size or delineation of the evaluation area
• Consider obtaining additional concentration data for each medium
• Re-run the screening evaluation to compare revised radionuclide concentration data with the generic BCGs
• Sum all fractions for each radionuclide and medium
Is sum of fractions < 1.0?
Yes
Evaluation is complete. Document rationale and results.
No
• Identify media and nuclide-specific limiting organism types
• Review and select lumped parameters appropriate for site-specific conditions and receptors
• Use site-specific lumped parameters to generate site-specific BCGs
• Compare radionuclide concentration data with site-specific BCGs. Sum all fractions for each radionuclide and medium
Is sum of fractions < 1.0?
Yes
Evaluation is complete. Document rationale and results.
No


Analysis Phase
(7)
Site-Specific Analysis
• Identify media and nuclide-specific limiting organism types
• Consider correction factor for exposure area or receptor residence time
• For riparian and terrestrial animals, review and select parameters contributing to internal dose (e.g., body mass; ingestion and inhalation rates; biological decay and f_1 values) appropriate for site-specific receptors
• Use site-specific parameters to generate site-specific BCGs
• Compare radionuclide concentration data with site-specific BCGs. Sum all fractions for each radionuclide and medium
Is sum of fractions < 1.0?
Yes
Evaluation is complete. Document rationale and results.
No
Site-Specific Biota Dose Assessment
• Consider use of available biota tissue data
• Assemble a biota dose assessment team
• Review requirements and assumptions
• Design and conduct the biota dose assessment
• Problem formulation
• Analysis
• Risk characterization
Document rationale and results.

4.1 Parameter Values that Can be Modified in the Graded Approach

DOE's three-phased approach is designed to guide you from an initial conservative evaluation using general screening to, if needed, a more rigorous analysis using site-specific information. The amount of effort required for your biota dose evaluation and the information needed on site-specific conditions and receptors increases as you progress through the three phases of the graded approach, particularly during the analysis phase. The result will be a set of less conservative, more realistic site-representative BCGs. Table 4.1 provides a general summary of parameter values that can be modified or applied corresponding to each phase of the graded approach. Use this table as a reference when progressing through the step-by-step guidance provided in subsequent sections of this Module.

Table 4.1 Summary of Parameter Values that Can, with Technical Justification, be Modified Corresponding to Each Phase of the Graded Approach

Phase	Parameters[1]
Data Assembly	• Size of evaluation area • Radionuclide concentrations in environmental media
General Screening	• Initial general screening using maximum radionuclide concentrations: No parameter modifications are allowed
Analysis: *Site-Specific Screening*	• Use of mean radionuclide concentrations, taking into account time dependence and spatial extent of contamination, may be considered • Site-specific lumped parameter values in place of default values used in the general screening phase • Sediment K_d values may be modified, with technical justification, for aquatic system evaluations where only water or only sediment concentration data are available for the screening process
Site-Specific Analysis	• A correction factor for exposure area or receptor residence time for all organism types may be considered • For riparian and terrestrial animals: - Food source B_{iv} value for riparian and terrestrial animals - Body mass - Uptake fraction of radionuclide ingested/absorbed (f_1) - Biological elimination rate constant of radionuclide exiting the organism (λ_{bio})

Table 4.1 (Continued) Summary of Parameter Values that Can, with Technical Justification, be Modified Corresponding to Each Phase of the Graded Approach

Phase	Parameters[1]
	- Food intake rate and supporting parameters - Soil intake rate and supporting parameters - Inhalation rate and supporting parameters - Soil inhalation rate and supporting parameters - Water consumption rate - Maximum life span - Allometric equations provided can be modified
Site-Specific Biota Dose Assessment	• Design, collection, and direct analysis of environmental media and biota

[1] The RAD-BCG Calculator provides the capabilities to modify the dose limits for aquatic and terrestrial organisms, to modify the RBE weighting factor for alpha emitters, and to de-select inclusion of energies for progeny of chain-decaying nuclides with regard to internal dose conversion factors. These default values shall be used in dose evaluations conducted for DOE sites. See Module 2, Section 7 for a detailed discussion on the selection of the RBE weighting factor for alpha emitters.

4.2 Use of the RAD-BCG Calculator

The RAD-BCG Calculator is a companion tool to the technical standard. It contains a series of electronic spreadsheets for use in:

- entering site data on radionuclide concentrations in soil, sediment, or water,
- comparing radionuclide-specific data with radionuclide-specific BCGs,
- determining if the sum of fractions for all radionuclide data/BCG comparisons is less than 1.0, and
- when technically justified, modifiying default parameters used in the general screening phase, and calculating site-specific BCGs using site-specific information representing the evaluation area and receptors.

A Table of Contents within the RAD-BCG Calculator provides a listing of the spreadsheets and information text screens, with a brief statement about their application. The contents of the RAD-BCG Calculator are also provided in Table 4.2.

Within these electronic spreadsheets, several fields (e.g., columns) of cells contain notes, viewed by placing the cursor over the cell, that provide additional information on the source of the number of parameter value cited in that cell. The equations used to derive the BCG calculations and to link values across different spreadsheets are presented in a separate

protected spreadsheet within the RAD-BCG Calculator. The equations and assumptions used to derive the BCGs are described in detail within Module 3 of this technical standard.

4.3 The Biota Dose Assessment Committee

The Biota Dose Assessment Committee (BDAC), chaired by DOE's Air, Water and Radiation Division (EH-412), is available as a resource to answer questions concerning the graded approach for evaluating radiation doses to biota. The BDAC is an approved technical standards topical committee organized under the DOE Technical Standards Program. As stated in its charter, the purpose of the BDAC is (a) to assist, consistent with DOE needs, in developing and promoting technical standards and associated guidance for DOE-wide applications in assessing radiation dose to biota, (b) to serve as a major forum within DOE for obtaining technical assistance, discussing technical issues, and sharing lessons learned regarding biota dose standards and assessment methods, and (c) to serve as a technical resource and advisory group for DOE program and field elements regarding site-specific biota dose assessments. The BDAC web site (http://homer.ornl.gov/oepa/public/bdac) provides internet access to guidance, methods, and related tools associated with this technical standard; links to related web sites also are provided. Specific questions concerning the guidance and methods contained in this technical standard, and requests for consultation with the BDAC Core Team, should be coordinated through EH-412 (contact Stephen Domotor, 202-586-0871, Stephen.Domotor@eh.doe.gov).

The BDAC is available as a resource to DOE program and field elements

The Department's Biota Dose Assessment Committee is available as a technical resource and advisory group concerning evaluation of radiation doses to biota. Questions concerning the application of the DOE graded approach should be coordinated through DOE's Air, Water and Radiation Division (EH-412).

Table 4.2 Contents of the RAD-BCG Calculator. A listing of the spreadsheets and information text screens, with a brief statement on their application and relationship to tables contained in this technical standard, is provided.

Spreadsheet Type	Spreadsheet Title	Content Description	Parameters That Can Be Modified
Information Text Screens	Front Page	Welcoming comments and a description of the purpose of the RAD-BCG Calculator, and its intended use. Provides a link to begin a semi-automated biota dose evaluation using the RAD-BCG Calculator.	
	Overview	Provides an overview of DOE's graded approach for evaluating radiation doses to aquatic and terrestrial biota. Summarizes the three phases (data assembly, general screening, analysis) of the graded approach. Provides a link to begin a semi-automated biota dose evaluation using the RAD-BCG Calculator.	
	Table of Contents	Lists all spreadsheets and information text screens included in the RAD-BCG Calculator.	
	Getting Started	Provides general considerations on the general site information required for defining the evaluation area and conducting a biota dose evaluation using the screening methods contained in the technical standard, along with general considerations when conducting an aquatic vs. terrestrial system evaluation. Provides a link for continuing on with a semi-automated evaluation.	
Principal Screening and Analysis Spreadsheets	Initial Conditions	Allows the user to select SI (e.g., Bq/kg) or special (e.g., pCi/g) units in the biota dose evaluation. Provides a feature to reset all parameters to their default values.	Restore initial default parameters; select units
	Aquatic and Terrestrial System Data Entry/BCG Worksheets	Provides the environmental system data entry/BCG worksheet for aquatic system evaluations and terrestrial system evaluations, respectively. Allows the user to enter data on radionuclide concentrations in soil, sediment and water. Lists the BCGs for each radionuclide. Calculates the sum of fractions for all radionuclide data/BCG comparisons and indicates if this sum of fractions is less than 1.0. Lists the limiting organism type responsible for the BCG cited, which references the organism type spreadsheet where default parameters can be modified with site-specific values.	Site radionuclide concentration data for soil, sediment, and water

Spreadsheet Type	Spreadsheet Title	Content Description	Parameters That Can Be Modified
Supporting Parameter and Reference Spreadsheets	Aquatic Animal	Contains the basic parameters used in the calculation of water and sediment BCGs for aquatic biota. Contains all of the same information and parameters as is presented in Module 1, Table 7.1.	B_{iv}; correction factor for exposure area and time
	Terrestrial Plant	Contains the basic parameters used in the calculation of water and soil BCGs for terrestrial plants. Contains all of the same information and parameters as is presented in Module 1, Table 7.3.	B_{iv}; correction factor for exposure area and time
	Riparian Animal	Contains the basic parameters used in the calculation of water and sediment BCGs for riparian animals. Contains all of the same information and parameters as is presented in Module 1, Tables 7.2, 7.5, and 7.6.	With "lumped BCGs" selected: lumped parameter; correction factors for exposure area and time; with "allometric BCGs" selected: correction factors for exposure area and time; fraction of intake retained; biological decay constant; all allometric parameters and equations
	Terrestrial Animal	Contains the basic parameters used in the calculation of water and soil BCGs for terrestrial animals. Contains all of the same information and parameters as is presented in Module 1, Tables 7.4, 7.7, and 7.8.	With "lumped BCGs" selected: lumped parameter; correction factors for exposure area and time; with "allometric BCGs" selected: correction factors for exposure area and time; fraction of intake retained; biological decay constant; all allometric parameters and equations
Supporting Spreadsheets	Dose Factors and Common Parameters	Contains internal dose conversion factors, and external dose conversion factors in soil, sediment and water for each radionuclide. Contains sediment and soil most probable K_d values used as default parameters, and provides their ranges. Contains all of the same information and parameters as is presented in Module 1, Tables 6.5 and 7.9.	Most probable K_d values; radiation weighting factor for alpha emitters; inclusion of energies for progeny of chain-decaying nuclides with regard to internal dose conversion factors
	Decay Chains	Contains decay chains (both with and without progeny) for each radionuclide. This spreadsheet is not provided in the technical standard.	

5 Data Assembly Phase

The DOE graded approach for evaluating radiation doses to aquatic and terrestrial biota was designed to minimize the need for additional data collection above and beyond environmental radionuclide concentration data typically available through routine environmental monitoring and surveillance programs. The data assembly phase encompasses three steps: (1) considering the sources of radioactivity, the key receptors, and the routes of exposure to these receptors; (2) defining the geographic area to be evaluated; and (3) assembling and organizing data on radionuclide concentrations in water, sediments, and soil for use in the general screening phase, and for use in the analysis phase, if needed. Each of the three steps are interdependent and should be considered collectively when implementing the data assembly phase.

5.1 Step 1: Consider the Sources, Receptors, and Routes of Exposure

It is expected that general knowledge concerning sources, receptors, and routes of exposure will be sufficient for defining the geographic area of evaluation when implementing the general screening phase of the graded approach. However, more detailed information regarding these elements may need to be considered as you progress through the graded approach. For example, if the BCGs for the general screening evaluation are exceeded, you may wish to refine your input data for site-specific screening (e.g., using mean radionuclide concentration data in place of maximum values; re-defining the geographic area of evaluation). Alternatively, you may wish to move to the site-specific analysis component of the graded approach, which may require consideration of internal dose parameters relating to site-specific receptors and routes of exposure. Detailed guidance on consideration of sources, receptors, and routes of exposure, for application in defining the area of evaluation and for use in the analysis phase, is provided in Module 2, Section 2.

5.2 Step 2: Define Your Area of Evaluation

It is necessary to determine the spatial extent over which the graded approach will be applied. The assumptions regarding sources, receptors, and routes of exposure used in the development of the graded approach provide for conservative BCGs. In the derivation of the screening approach, the source medium to which the organisms are exposed is assumed to be infinite in extent and to contain uniform concentrations of radionuclides. The organisms are also assumed to be resident in the contaminated area (e.g., exposed to contaminated media) 100 percent of the time. Given these

Three conditions should be present for a dose evaluation:

- *Radioactivity should be present or anticipated to be present in the environment as a result of DOE activities*
- *Receptors (i.e., plants and/or animals) should be present in the vicinity of those sources*
- *Routes of exposure should exist from those sources to the receptors*

assumptions, the first approach shall be to use maximum radionuclide concentration data applicable to your geographic area of interest (e.g., the entire site). A review of your effluent monitoring and environmental surveillance program design and resultant data should provide insights on sampling locations yielding the highest radionuclide concentrations.

5.3 Step 3: Assemble and Organize Data on Radionuclide Concentrations in Environmental Media

The next step is to collect and organize relevant data on radionuclide concentrations in environmental media. Radionuclide concentrations in surface water and/or sediment and in soil are needed for implementing the graded approach. Acceptable sources of data include but are not limited to: Annual Site Environmental Reports, effluent monitoring and environmental surveillance data, remediation data, and data from special site-specific studies (e.g., ecological studies conducted for other purposes). The data should be organized by location and medium, and be applicable to the geographic area of evaluation identified in Step 2 above. Locations may be defined by management and administrative characteristics (e.g., remediation sites; operations areas; operable units), physical characteristics (e.g., watershed; pond; stream), or ecological characteristics (e.g., corresponding to habitat types). Maximum radionuclide concentrations in environmental media shall be used in the initial application of the general screening phase to provide the most conservative evaluation.

5.3.1 Aquatic System Considerations

If you are conducting an aquatic system evaluation, note that use of radionuclide concentration data from co-located surface water and sediment samples is preferred and will result in a less conservative, more realistic evaluation. A mix of data from water and/or sediment samples collected from different locations within the vicinity of one another may be used, with justification. Note that where co-located samples are not available, only water or only sediment data may be used, but will result in a significantly more conservative evaluation. This is because the BCGs derived using individual water or sediment values involve the use of a conservative sediment distribution coefficient (K_d) to calculate the environmental media radionuclide concentration and dose contribution of either the missing water or sediment component.

5.3.2 Terrestrial System Considerations

If you are conducting a terrestrial system evaluation, you should consider the types of receptors resident in your area of evaluation and the appropriateness of your soil samples with regard to these receptors. For example, surface soil samples may not be representative of potential radionuclide exposure to deep-rooted plant receptors. Refer to Module 2, Section 5 for detailed guidance in this area. Also note that if you have a water body in your evaluation area, you must also conduct an aquatic system evaluation.

6 General Screening Phase

A major goal of the general screening phase is to provide a method that allows you to easily apply data on radionuclide concentrations in an environmental medium to evaluate compliance with the dose limits for biota. In the general screening phase, data on radionuclide concentrations in environmental media are compared with a set of generic BCGs. Each radionuclide-specific BCG represents the limiting radionuclide concentration in environmental media which would not result in DOE's established or recommended dose limits for biota to be exceeded. These limiting radionuclide concentrations, or BCGs, are presented in Tables 6.1 through 6.4. These "look-up" tables allow for quick, easy comparisons of radionuclide concentrations in environmental media with the BCGs. Guidance on using these look-up tables is provided below.

6.1 Step 1: Compare Data on Radionuclide Concentrations in Environmental Media with Generic BCGs Contained in Look-up Tables

A sum of fractions approach is used in comparing data on measured radionuclide concentrations in environmental media with the BCGs contained in the look-up tables. That is, when multiple radionuclides are present in multiple environmental media, the sum of fractions rule shall be applied to account for all sources of exposure. Hence, the sum of the ratios of the measured concentration of each radionuclide to its corresponding BCG for each medium shall then be summed across media, and the total sum of fractions shall not exceed 1.0.

Sum of Fractions Rule

When multiple radionuclides are present in multiple environmental media, the sum of fractions rule shall be applied to account for all sources of exposure.

For each environmental medium, for radionuclides A, B, ... N, with concentrations C_A, C_B ...C_N, and corresponding screening BCG values BCG_A, BCG_B, ... BCG_N, this relationship for aquatic and terrestrial system evaluations is as follows:

- Aquatic System Evaluation:

$$\left[\frac{C_A}{BCG_A} + \frac{C_B}{BCG_B} + \ldots + \frac{C_N}{BCG_N}\right]_{\text{water}} + \left[\frac{C_A}{BCG_A} + \frac{C_B}{BCG_B} + \ldots + \frac{C_N}{BCG_N}\right]_{\text{sediment}} < 1.0$$

- Terrestrial System Evaluation:

$$\left[\frac{C_A}{BCG_A} + \frac{C_B}{BCG_B} + \ldots + \frac{C_N}{BCG_N}\right]_{\text{water}} + \left[\frac{C_A}{BCG_A} + \frac{C_B}{BCG_B} + \ldots + \frac{C_N}{BCG_N}\right]_{\text{soil}} < 1.0$$

If the sum of fractions (the summed ratios between the radionuclide concentrations in environmental media and the radionuclide-specific BCGs) is less than 1.0, the dose to an aquatic or terrestrial receptor is below the biota dose limit, and you have passed the general screening evaluation. Proceed to Section 8, Documenting Your Biota Dose Evaluation Results. If the sum is greater than 1.0, further investigation is required (e.g., initiating site-specific screening or analysis).

Getting Started with the RAD-BCG Calculator

Enable Macros. Click on "Enable Macros" when prompted.

Select your units. You may work in either SI Units (e.g., Bq/kg) or Special Units (e.g., pCi/g). Select your units in the "Initial Conditions" spreadsheet of the RAD-BCG calculator.

Enter your data. The RAD-BCG Calculator contains aquatic and terrestrial system data entry/BCG worksheets. These environmental data/BCG worksheets allow you to enter your data on radionuclide concentrations in environmental media, automatically calculate the sum of fractions, and determine whether the sum of fractions is greater or less than 1.0.

When entering data for an aquatic system evaluation, be sure to select "water," "sediment," or "both," corresponding to the data you are working with.

The terrestrial system data entry/BCG worksheet provides a feature that allows you to import water data used in the aquatic evaluation, as appropriate.

Prepare for General Screening. To prepare for general screening, be sure that the "lumped BCGs" button is selected within the riparian and terrestrial animal spreadsheets.

Using the Sum of Fractions Rule: Terrestrial System Evaluation

Maximum radionuclide concentrations for water and soil collected within the evaluation area and available through the existing site environmental surveillance program were summarized. Maximum radionuclide concentrations for Cs-137 and Sr-90 in soil were 1.21 and 1.30 pCi/g, respectively. Maximum radionuclide concentrations for Cs-137 and Sr-90 in water were 49.6 and 84.5 pCi/L, respectively. Applying the sum of fractions rule, and using the BCG values listed in Table 6.4, one obtains the following:

soil: $\frac{1.21}{20} + \frac{1.30}{20} = 1.2E-01$

water: $\frac{49.6}{6E+05} + \frac{84.5}{5E+04} = 1.63E-03$

1.2E-01 + 1.77E-03 = 0.12
(soil sum of fractions) (water sum of fractions) (total sum of fractions)

Conclusion: Because 0.12 is less than 1.0, the dose to a terrestrial receptor does not exceed the recommended dose limits for protection of populations of terrestrial plants and animals. Note that the soil medium provides most of the contribution to dose.

Using the Sum of Fractions Rule: **Aquatic System Evaluation**

Maximum radionuclide concentrations for co-located water and sediment samples collected within the evaluation area and available through the existing site environmental surveillance program were summarized. Maximum radionuclide concentrations for water and sediment are:

	Sr-90	Cs-137
water (pCi/L)	1.5E-03	ND
sediment (pCi/g)	3.8	7.9

Applying the sum of fractions rule, and using the BCG values listed in Table 6.2, one obtains the following:

$$\frac{1.5E\bullet 03}{3E\bullet 02} \bullet\bullet \frac{0}{4E\bullet 01} \bullet\bullet 5.0E\bullet 06$$ (sum of fractions for radionuclides in water)

$$\frac{3.8}{6E\bullet 02} \bullet\bullet \frac{7.9}{3E\bullet 03} \bullet\bullet 8.96E\bullet 03$$ (sum of fractions for radionuclides in sediment)

$$5.0E\bullet 06 \bullet\bullet 8.96E\bullet 03 \bullet\bullet 8.96E\bullet 03$$ (total sum of fractions for radionuclides in water and sediment)

Conclusion: Dose to an aquatic receptor does not exceed the recommended dose limits for aquatic or riparian animals.

6.1.1 Aquatic System Considerations

In situations where co-located water and sediment data are not available, in the general screening phase you must estimate the missing radionuclide concentration data through the use of "most probable" radionuclide-specific K_d values. Radionuclide-specific most probable K_d values are provided in Table 6.5 of this Module and in the Dose Factors and Common Parameters spreadsheet of the RAD-BCG Calculator. The radionuclide concentration data estimated for the missing water or sediment medium is then used along

Estimating Radionuclide Concentration Data in Situations where Co-Located Water and Sediment Data are not Available

The RAD-BCG Calculator uses a "most probable" default K_d value to automatically calculate the missing radionuclide concentration, and then automatically enters it into the aquatic system data entry/BCG worksheet.

with the radionuclide concentration data for the available medium in the sum of fractions calculation as described previously.

Judgement should be applied in determining if measured radionuclide concentration data for water and sediment media can be considered as originating from co-located water and sediment samples. If measured radionuclide concentration data for water and sediment media are only available from separate locations, you should calculate the missing radionuclide concentration data for each missing medium, and apply the approach that results in the highest (e.g., most conservative) sum of fractions in your biota dose evaluation. Equations for estimating radionuclide concentration data in situations where co-located water and sediment data are not available are provided in Module 3, Section 3.2.3. If the sum of fractions is less than 1.0, the dose to an aquatic receptor is below the biota dose limit, and you have passed the general screening evaluation. Proceed to Section 8, Documenting Your Biota Dose Evaluation Results. If the sum is greater than 1.0, further investigation is required (e.g., initiating site-specific screening or analysis).

6.1.2 Dealing with High Background Levels of Naturally Occurring Radionuclides

Radiation dose rates at local background reference sites can be used to ensure that the site-related dose rates represent an actual increase in exposure. If the evaluation area is suspected or has been documented to have high background levels of naturally occurring radionuclides, these background levels may be taken into account when determining compliance of DOE activities with the biota dose limits. For example, this may be a consideration for the two isotopes of radium (see BCGs for Ra-226 and Ra-228, Tables 6.1 - 6.4). Background levels for environmental media should be estimated based on data for the same or similar media types in uncontaminated areas. If the sum of fractions for measured radionuclide concentrations in media from the contaminated area exceeds 1.0, this sum should be compared with the sum of fractions calculated using measured radionuclide concentrations in media from the background area. If the sum of fractions from the contaminated area does not exceed that from the background area, the contaminated area has passed the screening evaluation. Proceed to Module 1, Section 8 and document the results of the comparison. If it does exceed the background sum of fractions, proceed to the next phases of the graded approach. Refer to Module 2, Section 3.3.1, and Module 2, Section 6.3.1.5 for related guidance on this topic.

Table 6.1 Biota Concentration Guides (BCGs) for Water and Sediment (in SI Units) for Use in Aquatic System Evaluations. For use with radionuclide concentrations from co-located water and sediment.

Nuclide	BCG (water), Bq/m^3	Organism Responsible for Limiting Dose in Water	BCG (sediment), Bq/kg	Organism Responsible for Limiting Dose in Sediment
^{241}Am	2E+04	Aquatic Animal	2E+05	Riparian Animal
^{144}Ce	6E+04	Aquatic Animal	1E+05	Riparian Animal
^{135}Cs	2E+04	Riparian Animal	2E+06	Riparian Animal
^{137}Cs	2E+03	Riparian Animal	1E+05	Riparian Animal
^{60}Co	1E+05	Aquatic Animal	5E+04	Riparian Animal
^{154}Eu	8E+05	Aquatic Animal	1E+05	Riparian Animal
^{155}Eu	1E+07	Aquatic Animal	1E+06	Riparian Animal
^{3}H	1E+10	Riparian Animal	1E+07	Riparian Animal
^{129}I	1E+06	Riparian Animal	1E+06	Riparian Animal
^{131}I	5E+05	Riparian Animal	2E+05	Riparian Animal
^{239}Pu	7E+03	Aquatic Animal	2E+05	Riparian Animal
^{226}Ra	2E+02	Riparian Animal	4E+03	Riparian Animal
^{228}Ra	1E+02	Riparian Animal	3E+03	Riparian Animal
^{125}Sb	1E+07	Aquatic Animal	3E+05	Riparian Animal
^{90}Sr	1E+04	Riparian Animal	2E+04	Riparian Animal
^{99}Tc	2E+07	Riparian Animal	2E+06	Riparian Animal
^{232}Th	1E+04	Aquatic Animal	5E+04	Riparian Animal
^{233}U	7E+03	Aquatic Animal	2E+05	Riparian Animal
^{234}U	7E+03	Aquatic Animal	2E+05	Riparian Animal
^{235}U	8E+03	Aquatic Animal	1E+05	Riparian Animal
^{238}U	8E+03	Aquatic Animal	9E+04	Riparian Animal
^{65}Zn	5E+02	Riparian Animal	5E+04	Riparian Animal
^{95}Zr	3E+05	Aquatic Animal	9E+04	Riparian Animal

Table 6.2 Biota Concentration Guides (BCGs) for Water and Sediment (in Special Units) for Use in Aquatic System Evaluations. For use with measured radionuclide concentrations from co-located water and sediment.

Nuclide	BCG (water), pCi/L	Organism Responsible for Limiting Dose in Water	BCG (sediment), pCi/g	Organism Responsible for Limiting Dose in Sediment
^{241}Am	4E+02	Aquatic Animal	5E+03	Riparian Animal
^{144}Ce	2E+03	Aquatic Animal	3E+03	Riparian Animal
^{135}Cs	5E+02	Riparian Animal	4E+04	Riparian Animal
^{137}Cs	4E+01	Riparian Animal	3E+03	Riparian Animal
^{60}Co	4E+03	Aquatic Animal	1E+03	Riparian Animal
^{154}Eu	2E+04	Aquatic Animal	3E+03	Riparian Animal
^{155}Eu	3E+05	Aquatic Animal	3E+04	Riparian Animal
^{3}H	3E+08	Riparian Animal	4E+05	Riparian Animal
^{129}I	4E+04	Riparian Animal	3E+04	Riparian Animal
^{131}I	1E+04	Riparian Animal	5E+03	Riparian Animal
^{239}Pu	2E+02	Aquatic Animal	6E+03	Riparian Animal
^{226}Ra	4E+00	Riparian Animal	1E+02	Riparian Animal
^{228}Ra	3E+00	Riparian Animal	9E+01	Riparian Animal
^{125}Sb	4E+05	Aquatic Animal	7E+03	Riparian Animal
^{90}Sr	3E+02	Riparian Animal	6E+02	Riparian Animal
^{99}Tc	7E+05	Riparian Animal	4E+04	Riparian Animal
^{232}Th	3E+02	Aquatic Animal	1E+03	Riparian Animal
^{233}U	2E+02	Aquatic Animal	5E+03	Riparian Animal
^{234}U	2E+02	Aquatic Animal	5E+03	Riparian Animal
^{235}U	2E+02	Aquatic Animal	4E+03	Riparian Animal
^{238}U	2E+02	Aquatic Animal	2E+03	Riparian Animal
^{65}Zn	1E+01	Riparian Animal	1E+03	Riparian Animal
^{95}Zr	7E+03	Aquatic Animal	2E+03	Riparian Animal

Table 6.3 Biota Concentration Guides (BCGs) for Water and Soil (in SI Units) for Use in Terrestrial System Evaluations.

Nuclide	BCG (water), Bq/m^3	Organism Responsible for Limiting Dose in Water	BCG (soil), Bq/kg	Organism Responsible for Limiting Dose in Soil
^{241}Am	7E+06	Terrestrial Animal	1E+05	Terrestrial Animal
^{144}Ce	1E+08	Terrestrial Animal	5E+04	Terrestrial Animal
^{135}Cs	3E+08	Terrestrial Animal	1E+04	Terrestrial Animal
^{137}Cs	2E+07	Terrestrial Animal	8E+02	Terrestrial Animal
^{60}Co	4E+07	Terrestrial Animal	3E+04	Terrestrial Animal
^{154}Eu	8E+07	Terrestrial Animal	5E+04	Terrestrial Animal
^{155}Eu	1E+09	Terrestrial Animal	6E+05	Terrestrial Animal
^{3}H	9E+09	Terrestrial Animal	6E+06	Terrestrial Animal
^{129}I	2E+08	Terrestrial Animal	2E+05	Terrestrial Animal
^{131}I	7E+07	Terrestrial Animal	3E+04	Terrestrial Animal
^{239}Pu	7E+06	Terrestrial Animal	2E+05	Terrestrial Animal
^{226}Ra	3E+05	Terrestrial Animal	2E+03	Terrestrial Animal
^{228}Ra	3E+05	Terrestrial Animal	2E+03	Terrestrial Animal
^{125}Sb	3E+08	Terrestrial Animal	1E+05	Terrestrial Animal
^{90}Sr	2E+06	Terrestrial Animal	8E+02	Terrestrial Animal
^{99}Tc	6E+08	Terrestrial Animal	2E+05	Terrestrial Animal
^{232}Th	2E+06	Terrestrial Animal	6E+04	Terrestrial Animal
^{233}U	1E+07	Terrestrial Animal	2E+05	Terrestrial Animal
^{234}U	1E+07	Terrestrial Animal	2E+05	Terrestrial Animal
^{235}U	2E+07	Terrestrial Animal	1E+05	Terrestrial Animal
^{238}U	2E+07	Terrestrial Animal	6E+04	Terrestrial Animal
^{65}Zn	6E+06	Terrestrial Animal	2E+04	Terrestrial Animal
^{95}Zr	8E+07	Terrestrial Animal	4E+04	Terrestrial Animal

Table 6.4 Biota Concentration Guides (BCGs) for Water and Soil (in Special Units) for Use in Terrestrial System Evaluations.

Nuclide	BCG (water), pCi/L	Organism Responsible for Limiting Dose in Water	BCG (soil), pCi/g	Organism Responsible for Limiting Dose in Soil
^{241}Am	2E+05	Terrestrial Animal	4E+03	Terrestrial Animal
^{144}Ce	3E+06	Terrestrial Animal	1E+03	Terrestrial Animal
^{135}Cs	8E+06	Terrestrial Animal	3E+02	Terrestrial Animal
^{137}Cs	6E+05	Terrestrial Animal	2E+01	Terrestrial Animal
^{60}Co	1E+06	Terrestrial Animal	7E+02	Terrestrial Animal
^{154}Eu	2E+06	Terrestrial Animal	1E+03	Terrestrial Animal
^{155}Eu	3E+07	Terrestrial Animal	2E+04	Terrestrial Animal
^{3}H	2E+08	Terrestrial Animal	2E+05	Terrestrial Animal
^{129}I	6E+06	Terrestrial Animal	6E+03	Terrestrial Animal
^{131}I	2E+06	Terrestrial Animal	9E+02	Terrestrial Animal
^{239}Pu	2E+05	Terrestrial Animal	6E+03	Terrestrial Animal
^{226}Ra	8E+03	Terrestrial Animal	5E+01	Terrestrial Animal
^{228}Ra	7E+03	Terrestrial Animal	4E+01	Terrestrial Animal
^{125}Sb	7E+06	Terrestrial Animal	3E+03	Terrestrial Animal
^{90}Sr	5E+04	Terrestrial Animal	2E+01	Terrestrial Animal
^{99}Tc	2E+07	Terrestrial Animal	4E+03	Terrestrial Animal
^{232}Th	5E+04	Terrestrial Animal	2E+03	Terrestrial Animal
^{233}U	4E+05	Terrestrial Animal	5E+03	Terrestrial Animal
^{234}U	4E+05	Terrestrial Animal	5E+03	Terrestrial Animal
^{235}U	4E+05	Terrestrial Animal	3E+03	Terrestrial Animal
^{238}U	4E+05	Terrestrial Animal	2E+03	Terrestrial Animal
^{65}Zn	2E+05	Terrestrial Animal	4E+02	Terrestrial Animal
^{95}Zr	2E+06	Terrestrial Animal	1E+03	Terrestrial Animal

Table 6.5 Part 1 of Dose Factors and Common Parameters Spreadsheet. Most Probable K_d values for use in calculating generic BCGs for water and sediment in situations where co-located water and sediment samples are unavailable.

	Distribution Coefficients, K_d					
Nuclide	Maximum Value L/kg (mL/g)	Reference $K_{d,max}$	Minimum Value L/kg (mL/g)	Reference $K_{d,min}$	Most Probable Value [(a)] L/kg (mL/g)	Reference $K_{d,mp}$
^{241}Am	6.5E+05	T&M	8.5E+01	T&M	5.0E+03	T&M
^{144}Ce	1.0E+07	T&M	1.0E+02	RESRAD	1.0E+03	RESRAD
^{135}Cs	8.0E+04	T&M	1.7E+01	T&M	5.0E+02	RESRAD
^{137}Cs	8.0E+04	T&M	1.7E+01	T&M	5.0E+02	RESRAD
^{60}Co	3.0E+05	T&M	1.0E+02	RESRAD	1.0E+03	RESRAD
^{154}Eu	1.3E+05	T&M	2.0E+02	T&M	5.0E+02	T&M
^{155}Eu	1.3E+05	T&M	2.0E+02	T&M	5.0E+02	T&M
^{3}H	1.0E-04	KAH	1.0E-05	KAH	1.0E-03	KAH
^{129}I	1.0E+02	T&M	1.0E-05	KAH	1.0E+01	T&M
^{131}I	1.0E+02	T&M	1.0E-05	KAH	1.0E+01	T&M
^{239}Pu	1.0E+07	T&M	1.0E+02	T&M	2.0E+03	RESRAD
^{226}Ra	1.0E+03	T&M	1.0E-01	RESRAD	7.0E+01	RESRAD
^{228}Ra	1.0E+03	T&M	1.0E-01	RESRAD	7.0E+01	RESRAD
^{125}Sb	1.0E+03	KAH	1.0E-03	KAH	1.0E+00	KAH
^{90}Sr	4.0E+03	T&M	1.0E-01	RESRAD	3.0E+01	RESRAD
^{99}Tc	1.0E+02	T&M	1.0E-05	KAH	5.0E+00	T&M
^{232}Th	1.0E+06	T&M	1.2E+00	RESRAD	6.0E+04	RESRAD
^{233}U	5.0E+01	RESRAD	1.0E-01	RESRAD	5.0E+01	RESRAD
^{234}U	5.0E+01	RESRAD	1.0E-01	RESRAD	5.0E+01	RESRAD
^{235}U	5.0E+01	RESRAD	1.0E-01	RESRAD	5.0E+01	RESRAD
^{238}U	5.0E+01	RESRAD	1.0E-01	RESRAD	5.0E+01	RESRAD
^{65}Zn	1.0E+04	T&M	2.0E+00	RESRAD	2.0E+01	RESRAD
^{95}Zr	1.0E+05	T&M	1.0E+02	RESRAD	1.0E+03	RESRAD

T&M = Till and Meyer 1983; RESRAD = Yu et al. 1993; KAH = Estimation by K. A. Higley, Oregon State University.
(a) = "Most Probable" values shall be used to generate the generic BCGs for use in general screening in a case where only water or sediment data are available. This value may be modified using a site-representative K_d value in the analysis phase of the graded approach.

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7 Analysis Phase

The analysis phase of the graded approach contains three increasingly more detailed components of analysis for evaluating doses to biota: site-specific screening, site-specific analysis, and site-specific biota dose assessment. In the analysis phase, you are also increasingly moving away from the default parameters and assumptions used in the general screening phase of the graded approach. The amount of effort required for your biota dose evaluation and the information needed about site-specific conditions and receptors increase as you progress through the three components of the analysis phase. The amount of specialized assistance (e.g., in health physics, radioecology, and eco-risk assessment) that might be needed also increases as you progress through the components of the analysis phase. In return, the result will be a set of less conservative, more realistic and site-representative BCGs. **The rationale for selection of site-specific parameters applied in this phase shall be sufficiently documented when reporting your biota dose evaluation results**. Each of the three analysis components is described below.

7.1 Analysis Phase - Site-Specific Screening

Site-specific screening allows you to apply knowledge of site-specific conditions and receptors in your biota dose evaluation in place of the default parameter values and assumptions used in the general screening phase of the graded approach. For example, use of mean radionuclide concentrations in place of maximum values, taking into account time dependence and spatial extent of contamination, may be considered. Parameters representative of site-specific receptors also may be considered. These considerations and their application in site-specific screening are discussed below.

Questions to Consider in Determining Your Path Forward in Site-Specific Screening:

Can I use mean radionuclide concentrations rather than maximum values?

Does it make sense to adjust or re-define my evaluation area, using knowledge of the spatio-temporal extent of my contamination with respect to receptor habitats?

Are the "limiting organism types" corresponding to my media and radionuclides expected to be present in my evaluation area?

Do I have site-representative parameters (e.g., lumped parameters; B_{iv}s; K_ds) that can be used in place of default values?

7.1.1 Step 1: Assess the Representativeness of Your Input Data on Radionuclide Concentrations in Environmental Media and the Delineation of Your Evaluation Area

Spatial and temporal variability relative to the distribution of contamination in the evaluation area can be taken into account when evaluating doses to biota. Each of the elements presented below should be considered collectively as you proceed through this step.

7.1.1.1 Consider Using Mean Radionuclide Concentrations

Determine if mean radionuclide concentrations can be used in place of maximum concentrations. For example, use of mean values is appropriate and permitted in situations where time-series data are available and of sufficient quality. Spatial variability in the distribution of contamination can also be taken into account. Note that depending on the purpose of your application of the graded approach, you may be requested (e.g., by regulators or stakeholders) to use only maximum radionuclide concentration data rather than mean values. Detailed guidance on applying spatio-temporal considerations in determining mean radionuclide concentrations for use in the graded approach is provided in Module 2, Section 3.

7.1.1.2 Consider Refining the Evaluation Area

It may be useful to re-assess your rationale for delineating the evaluation area (e.g., breaking one large area into several smaller areas) through consideration of the quality and spatio-temporal distribution of radionuclide concentration data, the ecological susceptibility and habitats of the receptors, and the spatial distribution of contaminants with respect to these habitats. Refer to Module 2, Section 4 for detailed guidance in this area.

7.1.1.3 Consider Obtaining Additional Radionuclide Concentration Data

Consider collecting additional radionuclide concentration data. For an aquatic system evaluation, consider using co-located water and sediment data if you have not already done so.

7.1.2 Step 2: Re-Run the Screening Evaluation Using Revised Radionuclide Concentration Data and/or Evaluation Area

Here you are comparing your refined data on measured radionuclide concentrations corresponding to your original or re-defined evaluation area, with the generic BCGs. This is done by re-entering these revised data into the appropriate environmental data/BCG worksheet in the RAD-BCG Calculator. It is important to note that in this step you have not modified the initial, generic BCG values. They are the same generic BCGs that are used in the general screening phase of the graded approach. This step is considered a site-specific screen in that you are now making site-specific judgements relative to your measured radionuclide concentration data and your evaluation area. If the sum of fractions is less than 1.0, then you

have passed the site-specific screening evaluation. Proceed to Section 8, Documenting Your Biota Dose Evaluation Results. If the sum of fractions is greater than 1.0, then continue to progress through the graded approach.

7.1.3 Step 3: Assess the Representativeness of Default Parameters and Assumptions Used in Deriving the Generic BCGs; Select Site-Specific Parameters and Generate Site-Specific BCGs

This step allows you to replace default parameters used in the general screening phase with site-representative parameters for use in site-specific screening. Each of the elements presented below should be considered collectively as you proceed through this step.

7.1.3.1 Identify Radionuclide-Specific Limiting Medium and Organism Type

Review the radionuclide-specific BCGs used in the general screening phase of the graded approach. First, identify the environmental medium and individual radionuclides from your evaluation that provide the greatest contribution to potential dose (e.g., medium concentration: BCG ratios that represent the largest contributors to the sum of fractions). Then, for each of these radionuclides, identify the limiting organism type from which the generic BCGs were derived. Limiting organism types corresponding to generic BCGs are listed for each radionuclide in Tables 6.1 - 6.4 and in the corresponding RAD-BCG Calculator spreadsheets. If you did not conduct a general screen prior to site-specific screening, go to the organism type table or spreadsheet that corresponds to the site-specific receptor you have chosen to use in your analysis. The site-specific receptor you select should be important to the structure and function of the community, in that protection of this organism within your evaluation area assures that all other organisms in your evaluation area are also protected. Some examples of receptors that could serve as good indicators of radiological impact are provided for your reference in Module 2, Section 2.1.3.

Selecting A Site-Specific Receptor

The receptor should be important to the structure and function of the community. It should: (1) be expected to receive a comparatively high degree of exposure (e.g., expected to receive a radiation dose to reproductive tissues which is relatively high per unit of radionuclide present in the ecosystem, in comparison to other receptors in the same community); (2) have a comparably high degree of radiosensitivity (e.g., radiation effects of concern occur at relatively low doses, in comparison with other receptors in the same community); and (3) exhibit a high degree of bioaccumulation.

7.1.3.2 Review and Select Site-Specific Lumped Parameters

The general screening phase uses a conservative default “lumped parameter” in the estimation of internal dose to an organism. The lumped parameter is based largely on empirical

measurements of radionuclides in biological tissues of organisms collected in contaminated habitats. In cases where empirical measurements are unavailable or few in number, the lumped parameter is based on a conservative value derived using uncertainty analysis on the kinetic/allometric method (see Module 3, Section 3.5). The lumped parameter serves as a "natural integrator" of internal contamination in that it inherently reflects all pathways of intake by an organism. Here, in site-specific screening, lumped parameters representative of site-specific conditions and receptors are used to generate site-specific BCGs in place of the default lumped parameters that were used in generating the generic BCGs. This site-specific screening results in a less conservative, more realistic evaluation of potential doses to biota for your area of evaluation.

The initial values of the lumped parameters were specifically chosen to produce conservative (e.g., highly protective) BCGs. It is recognized that actual lumped parameters for a single radionuclide may range over several orders of magnitude, depending upon biotic and abiotic features of the environment. In step 3 you review the default lumped parameters used in deriving the BCGs for the appropriate organism type. The default lumped parameter values (and other input parameters) are contained in a set of organism type tables (Tables 7.1 - 7.4). The RAD-BCG Calculator contains similar tables which can be easily located (see Module 1, Section 4). Review and select lumped parameters representative of site-specific conditions and receptors you have selected for your evaluation area. These site-specific lumped parameters are entered into the appropriate organism type spreadsheet in the RAD-BCG Calculator and used to generate site-specific BCGs. Sources for lumped parameter values representative of your site-specific conditions and receptors include: (1) your own site-derived lumped parameters (e.g., B_{iv}s) for site-specific receptors; (2) values published in the scientific literature or in site-specific technical reports (e.g., from specialized ecological studies) for receptors that are comparable to site-specific receptors in your evaluation area; and (3) databases such as the pilot version of the Biota Dose Assessment Database of Environmental Parameters (BDAD), which is accessible via the Internet through the BDAC web site (http://homer.ornl.gov/oepa/public/bdac).

7.1.3.3 Review and Select Site-Representative K_ds

For aquatic system evaluations where co-located water and sediment samples are not available, recall that in the general screening phase a most probable K_d is used to calculate the environmental media radionuclide concentration and dose contribution of either the missing water or sediment component. Site-specific screening allows you to consider the use of a site-representative K_d value in place of the default most probable value that was used in the general screening phase. Minimum, maximum, and most probable K_d values for each radionuclide are provided in Table 6.5. Sources for K_d values representative of your site specific conditions include: (1) your own site-derived K_d values; (2) values published in the scientific literature or in site-specific technical reports; and (3) databases such as the pilot version of the BDAD, which is accessible via the Internet (see above). Site-representative K_d values are entered into the

Dose Factors and Common Parameters spreadsheet within the RAD-BCG Calculator and used in generating site-specific BCGs.

7.1.4 Step 4: Re-Run the Screening Evaluation and Compare Data on Radionuclide Concentrations in Environmental Media with Newly-Generated Site-Specific BCGs

The use of lumped parameters appropriate for site-specific conditions or receptors should result in more realistic, site-representative BCGs. When using the RAD-BCG Calculator, the generic BCGs listed in the aquatic and terrestrial system data entry/BCG worksheets are automatically updated with the newly generated BCGs, allowing for easy evaluation. If the sum of fractions (the summed ratios between the radionuclide concentrations in environmental media and the radionuclide-specific BCGs) is less than 1.0, the dose to the aquatic or terrestrial receptor is below the biota dose limit. Refer to Section 8, Reporting Your Biota Dose Evaluation Results. If the sum is greater than 1.0, further analysis is required. Proceed to Section 7.2, Site-Specific Analysis.

Entering Site-Specific Information into the RAD-BCG Calculator to Calculate Site-Specific BCGs

Lumped parameters may be modified in each of the organism type spreadsheets contained in the RAD-BCG Calculator. When working in the riparian or terrestrial animal spreadsheets, click on the "Lumped BCGs" button to allow these parameters to be modified. A "user supplied value" message will appear for each lumped parameter modified. Reset buttons for returning all values to their defaults are also featured.

Site-specific K_d values may be used by entering these values in place of the "most probable" values in the Dose Factors and Common Parameters spreadsheet.

The site-specific BCGs derived using these new parameters will show up in the organism-type spreadsheet, and also in the environmental data entry/BCG worksheets, allowing for easy comparison with site radionuclide concentration data previously entered.

7.2 Analysis Phase - Site-Specific Analysis

In site-specific analysis, a kinetic/allometric model is employed to conduct a more rigorous analysis of riparian animal and terrestrial animal organism types. Here you are conducting a very site-specific evaluation (essentially estimating an upper-bound dose) to a site-specific riparian or terrestrial animal of known characteristics (e.g., body mass, behavior, internal exposure pathways, and parameters). Recall that the general and site-specific screening approaches use a lumped parameter in the estimation of internal dose to an organism. The lumped parameter serves as a "natural integrator" of internal contamination in that it inherently

reflects all pathways of intake by an organism. In site-specific analysis, simplistic, first-order kinetic modeling is used to examine the internal pathways of exposure for riparian animal and terrestrial animal receptors in greater detail. Appropriate parameters representing individual mechanisms (e.g., ingestion; inhalation) that contribute to internal dose are applied in place of the lumped parameter (one value which reflects all mechanisms contributing to internal dose). Appropriate values (e.g., organism body mass; ingestion rate; inhalation rate; biological uptake and elimination rates) representative of site-specific conditions and receptors are used in the estimation of internal dose and generation of site-specific BCGs. Allometric equations relating body size to many of these parameters (e.g., ingestion rate; inhalation rate; life span) are used in the estimation of internal dose. Alternatively, you can enter your own values in place of allometrically derived parameters. A correction factor for exposure area or organism residence time may also be applied for all organism types in site-specific analysis.

7.2.1 Step 5: Assess the Representativeness of Default Parameters and Assumptions Employed in Kinetic/Allometric Models; Select Site-Specific Parameters and Generate Site-Specific BCGs

This step allows you to examine and replace default parameters, assumptions, and allometric relationships used in kinetic/allometric models to derive BCGs for riparian animals and terrestrial animals. A correction factor for exposure area or organism residence time may also be applied for all organism types. Each of the elements presented below should be considered collectively when implementing this step.

7.2.1.1 Identify Radionuclide-Specific Limiting Medium and Organism Type

Review the radionuclide-specific BCGs used in the general or site-specific screening portions of the graded approach. First, identify the environmental medium and individual radionuclides from your evaluation that provide the greatest contribution to potential dose (e.g., medium concentration:BCG ratios that represent the largest contributors to the sum of fractions). Then, for each of these radionuclides, identify the limiting organism type from which the general or site-specific BCGs were derived. Limiting organism types corresponding to general BCGs are listed for each radionuclide in Tables 6.1 - 6.4, and in the corresponding RAD-BCG Calculator spreadsheets. If the riparian animal or terrestrial animal organism types are listed, then you may consider the guidance in Sections 7.2.1.2 - 7.2.1.4. If riparian or terrestrial animals are not listed as the limiting organism types, then you need only consider Section 7.2.1.2 below. If you did not conduct a general or site-specific screen prior to site-specific analysis, the proceeding statement applies to the site-specific receptor you have chosen to use in your analysis.

7.2.1.2 Consider Correction Factor for Exposure Area or Receptor Residence Time

A correction factor for exposure area or receptor residence time should be among the first parameters that you consider in site-specific analysis. Temporal and spatial variability can be taken into account when evaluating doses to biota. For example: (1) radionuclides will typically

be distributed non-uniformly in the environment; and (2) organisms are typically distributed non-uniformly within the environment such that exposure may vary among individuals in an affected population (e.g., organisms may migrate into and out of areas of greater and lesser contamination). The general and site-specific screening portions of the graded approach assume for conservative purposes that an organism's residence time in the evaluation area is 100 percent and that the contaminated media are available 100 percent of the time to provide a source of exposure. These assumptions can be modified in site-specific analysis.

Using a Correction Factor for Exposure Area or Receptor Residence Time in the RAD-BCG Calculator

A correction factor for exposure area or receptor residence time, located in each of the organism-type spreadsheets, may be applied. Site-specific BCGs derived using these correction factors will appear in the organism-type spreadsheets, and also in the environmental data entry/BCG worksheet, allowing for easy comparison with site radionuclide concentration data previously entered.

Note that in cases where a riparian or terrestrial animal was indicated as the limiting organism in general or site-specific screening, it is possible that "scaling down" the correction factor to reflect a very small percentage of time an organism spends in the contaminated area may result in triggering the identification of a new limiting organism type (e.g., aquatic animal; terrestrial plant).

Correction Factor for Receptor Residence Time. The term "residence time" as used in the graded approach refers to the fraction of time that an organism resides in a radioactively contaminated area. In site-specific analysis, a correction factor for residence time (e.g., as a percentage of time) may be applied to take into account a specific receptor's home range, movements, and behavior relative to the evaluation area. This

Using the Kinetic/Allometric Method for Riparian and Terrestrial Animals: Entering Site-Representative Parameters into the Riparian Animal and Terrestrial Animal Spreadsheets contained in the RAD-BCG Calculator.

First, click on the "Allometric BCGs" button to allow these parameters to be modified.

Individual parameters (e.g., body mass; ingestion rate; inhalation rate; radionuclide uptake and retention factors) related to mechanisms providing an internal dose may be modified.

Changing the radionuclide-specific food source (B_{iv}) values in the aquatic animal and terrestrial plant spreadsheets will automatically change the BCG values in the riparian animal and terrestrial animal spreadsheets, respectively.

Site-specific BCGs derived using these new parameter values will show up in the riparian and terrestrial animal spreadsheets, and also in the environmental data entry/BCG worksheets, allowing for easy comparison to site radionuclide concentration data previously entered.

correction factor is entered into the appropriate organism type spreadsheet within the RAD-BCG Calculator and used in generating site-specific BCGs.

Correction Factor for Exposure Area. Radionuclides will typically be distributed non-uniformly in the environment. In site-specific analysis, a correction factor for contaminated area (e.g., as a percentage of time) can be applied to take into account an intermittent source of exposure to all receptors in the evaluation area. This correction factor is entered into the appropriate organism type spreadsheet within the RAD-BCG Calculator and used in generating site-specific BCGs.

7.2.1.3 Riparian and Terrestrial Animals: Review and Select Parameters Representative of Site-specific Conditions and Receptors

In site-specific analysis you can also modify the individual parameters that relate to internal exposure pathways for site-specific conditions and receptors. The RAD-BCG Calculator is designed for easy modification of these parameters and subsequent generation of site-specific BCGs that are derived using these new parameter values. Refer back to Table 4.1 for a complete list of parameters that can be modified when conducting a site-specific analysis.

7.2.1.4 Riparian and Terrestrial Animals: Review and Select Food Source Parameter Values Representative of Site-Specific Receptors

The kinetic/allometric method for deriving riparian and terrestrial animal BCGs uses a radionuclide-specific food source parameter in calculating the internal dose contribution for these organism types. The method uses radionuclide-specific default B_{iv}s for aquatic animals (listed in Table 7.1) and terrestrial plants (listed in Table 7.3) as the default food source parameter values for riparian and terrestrial animals respectively. You may review the appropriateness of these default food source parameter values (i.e., the B_{iv}s and their source organisms) and replace these with food source parameter values (B_{iv}s) corresponding to organisms which are more representative of the expected food sources for the riparian or terrestrial animal you have selected to use in your site-specific analysis. When using the RAD-BCG Calculator, changing the radionuclide-specific B_{iv} values in the aquatic animal and terrestrial plant spreadsheets will automatically change the BCG values in the riparian animal and terrestrial animal spreadsheets respectively. These new site-specific BCGs will also show up in the environmental system data entry/BCG worksheets, allowing for easy comparisons with previously entered radionuclide concentration data.

7.2.2 Step 6: Re-Run the RAD-BCG Calculator and Compare Data on Radionuclide Concentrations in Environmental Media with Newly-Generated Site-Specific BCGs

The use of parameter values and a correction factor appropriate for site-specific conditions or receptors should result in more realistic, site-representative BCGs. If the sum of fractions (the

summed ratios between the radionuclide concentrations in environmental media and the radionuclide-specific BCGs) is less than 1.0, the dose to the aquatic or terrestrial receptor organism is below the biota dose limit. Refer to Section 8, Documenting Your Biota Dose Assessment Results. If the sum is greater than 1.0, further analysis is required.

7.3. Analysis Phase - Conducting a Site-Specific Biota Dose Assessment

7.3.1 Determine if Additional Analysis is Warranted

While the majority of the graded approach centers on the use of measured radionuclide concentrations in environmental media for comparison with BCGs, the site-specific biota dose assessment component of the analysis phase centers on the actual collection and analysis of biota from the evaluation area. This is so that measured concentrations of radionuclides in the tissues of biota can then be used to more realistically estimate the internal dose contribution to a site-specific receptor.

Should Additional Analysis or Remedial Action be Considered?

Factors to consider if initial general screening, site-specific screening, and site-specific analysis elements of the graded approach indicate a potential radiological impact to populations of biota within the evaluation area:

- The geographical extent of the contamination
- The magnitude of potential or observed effects of the contamination relative to the level of biological organization affected
- The likelihood that these effects could occur or will continue to occur
- The presence of genetically-isolated populations
- The ecological relationship of the affected area to the surrounding habitat
- The preservation of threatened or endangered species, or commercially or culturally valued species
- The recovery potential of the affected ecological resources and expected persistence of the radionuclides of concern under present site conditions
- The short- and long-term effects of the remedial alternatives on the habitat and the surrounding ecosystem
- Information obtained through a “lines of evidence” approach

Additional analysis may be warranted if biota dose evaluations using the screening and analysis methods described to this point continue to indicate that there is a potential adverse impact from radiation as a stressor to populations of biota (i.e., the BCGs are exceeded). An important point is that exceeding the BCGs should not force a mandatory decision regarding remediation of the evaluation area, but rather is an indication that further investigation is likely necessary. There are many factors that should be considered when deciding how to respond following a determination that the BCGs are exceeded (e.g., ecological relevance and susceptibility of the affected population; size of the contaminated area and persistence of contaminants; impacts of remediation alternatives).

If radionuclide concentrations in environmental media exceed the BCGs, two courses of action may be taken. On the one hand, it may be desirable to perform detailed dose assessments for relevant receptors. But given the potentially large expense that such a site-specific assessment could incur, removing the sources of ionizing radiation by reducing or eliminating discharges, or remediating existing environmental contamination, should also be considered. Site-specific conditions, especially the cost of eliminating discharges and/or remediating contaminated areas, will determine which approach is the more desirable.

The discussion below provides basic guidance on how to conduct a site-specific biota dose assessment.

7.3.2 An Important Note Concerning the Use of Available Biota Tissue Data

It is important to note that the use of measured concentrations of radionuclides in tissues of plants and animals in estimating internal dose is a reasonable and acceptable approach if adequate data are available. That is, if it can be justified that the available tissue data (1) are representative of species within the evaluation area that are capable of receiving the highest dose, and (2) reflect a representative sampling of the population within the evaluation area. These considerations are especially important in cases where biota tissue data becomes available as a result of opportunistic sampling (e.g., road kills; hunting). Detailed guidance regarding the selection of representative receptor species, and representative population and exposure considerations, is provided in Module 2, Section 6. If available biota tissue data is determined to be inadequate, then collection and analysis of biota from the evaluation area will be required. The internal dose conversion factors for biota, and external dose conversion factors for water, sediment and soil used to derive the generic BCGs in the graded approach are provided in Table 7.9. These values, together with your measured radionuclide concentrations in water, sediment and soil, and biota tissue data, can be used to estimate an upper-bound dose to a receptor.

7.3.3 Step 1: Assemble a Biota Dose Assessment Team

The composition of the biota dose assessment team is critical to designing and conducting a technically sound dose assessment. Together, team members must have a complete set of the

relevant skills necessary to do the work. Necessary skills will vary somewhat by site, but should include ecology, health physics, radioecology, and specialists in fate and transport of contaminants for the environmental media of interest. Depending on the regulatory compliance agreements and monitoring program requirements that exist at the site, it may also be desirable to have a regulatory specialist participate in the assessment. Other site-specific conditions will dictate the need for other related skills within the team or the need for direct stakeholder participation at this level.

7.3.4 Step 2: Review Requirements

To perform a detailed dose assessment, it will usually be necessary to design and conduct a relatively comprehensive environmental study of the sources of ionizing radiation and the potential receptors (e.g., to involve collection and analysis of site-specific organisms within the evaluation area). Such a study should be consistent with the requirements of applicable DOE Orders and guidance, Federal regulations, and State regulations. Particularly important are the following DOE Orders:

- Order 5400.1, *General Environmental Protection Program*
- Order 5400.5, *Radiation Protection of the Public and the Environment*
- Order 414.1A, *Quality Assurance*

These Orders, and the Federal legislation and Executive Orders cited therein, applicable State regulations, and applicable DOE site-specific requirements should be consulted during the design and conduct of field and laboratory studies to support dose assessments.

7.3.5 Step 3: Review Assumptions

Two assumptions will most likely be implicit in the dose assessment:

- Because it will be impossible to assess dose to all potential receptor populations in the area of contamination, one (to several) receptor species must serve as surrogates for all potentially exposed populations. Therefore, species selected for dose assessment should be among those that are most sensitive to the effects of ionizing radiation, helping to ensure that all populations are protected.
- The population of the receptor species for which doses are assessed is defined as those individuals living within the contaminated area. This assumption is consistent with the EPA definition of “population.” This assumption is conservative to the extent that individuals move in and out of the contaminated area.

Any deviations from the above assumptions when designing or conducting the dose assessment should be documented.

7.3.6 Recommended Approaches to Designing and Conducting the Dose Assessment

It is strongly recommended that all dose assessments be designed and conducted following the *Guidelines for Ecological Risk Assessment* (EPA 1998). Use of these guidelines will help ensure that the resulting dose assessments are technically sound. In addition, some of the steps in the ecological risk process (e.g., development of a site conceptual model) will be useful for assessing toxicological risks associated with some radionuclides (e.g., uranium isotopes) as well as the ecological risks from other co-occurring substances or stressors within the contaminated area (e.g., hazardous chemicals). The site conceptual model will also be useful for understanding the large-scale distribution of contaminants and the sources of ecological risk to the populations within and beyond the study area. *Guidelines for Ecological Risk Assessment* can be downloaded from the DOE EH-41 Dose and Risk Assessment web site (http://www.eh.doe.gov/oepa/risk). An electronic tool for developing a site conceptual model is also available at this web site. If multiple stressors are present and need to be evaluated, then appropriate guidance concerning cumulative risk assessment should be considered (e.g., see EPA 1997b).

In addition to the references found in EPA's *Guidelines for Ecological Risk Assessment*, the following references and materials should be useful, many of which are also available on the EH-41 Dose and Risk Assessment web site: (http:/www.eh.doe.gov/oepa/risk).

- G.R. Bilyard, H. Beckert, J.J. Bascietto, C.W. Abrams, S.A. Dyer, and L.A. Haselow. 1997. *Using the Data Quality Objectives Process During the Design and Conduct of Ecological Risk Assessments.* DOE/EH-0544, prepared for U.S. Department of Energy, Office of Environmental Policy and Assistance by Pacific Northwest National Laboratory, Richland, Washington.

- B.E. Sample, M.S. Aplin, R.A. Efroymson, G.W. Suter II, and C.J.E. Welsh. 1997. *Methods and Tools for Estimation of the Exposure of Terrestrial Wildlife to Contaminants.* ORNL/TM-13391, prepared for U.S. Department of Energy, Office of Environmental Policy and Assistance by Oak Ridge National Laboratory, Oak Ridge, Tennessee.

- U.S. Department of Energy. 1991. *Environmental Regulatory Guide for Radiological Effluent Monitoring and Environmental Surveillance.* DOE/EH-0173T, Assistant Secretary for Environment, Safety and Health, U.S. Department of Energy, Washington, D.C.

•• U.S. Department of Energy. 1998. *Compendium of EPA-Approved Analytical Methods for Measuring Radionuclides in Drinking Water.* Office of Environmental Policy and Assistance, Assistant Secretary for Environment, Safety and Health, U.S. Department of Energy, Washington, D.C.

•• U.S. Environmental Protection Agency (EPA). 1997. *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments.* EPA 540-R-97-006 (Interim Final June 5, 1997), U.S. EPA, Washington, D.C.

7.3.7 Designing and Conducting the Dose Assessment

The *Guidelines for Ecological Risk Assessment* (EPA 1998) provide a flexible framework for assessing ecological risks. The framework consists of three major phases of activity: problem formulation, analysis, and risk characterization. Activities within each of these phases can be summarized as follows:

In *problem formulation,* risk assessors evaluate goals and select assessment endpoints, prepare the conceptual model, and develop an analysis plan. During the *analysis phase*, assessors evaluate exposure to stressors and the relationship between stressor levels and ecological effects. In the third phase, *risk characterization*, assessors estimate risk (or dose) through integration of exposure and stressor-response profiles, describe risk by discussing lines of evidence and determining ecological adversity, and prepare a report. A more detailed "primer" on how to evaluate doses to biota through the ecological risk assessment process is provided in Module 2, Section 1.

Assessment Endpoint

An explicit expression of the environmental value that is to be protected, operationally defined by an ecological entity and its attributes. For example, salmon are valued ecological entities; reproduction and age class structure are some of their important attributes. Together "salmon reproduction and age class structure" form an assessment endpoint.

The dose assessment team has considerable latitude over how activities should be conducted within each phase of the assessment. The dose limits recommended in Module 1, Section 1.1 do not compromise this flexibility, but provide a major advantage for the dose assessment team because they define doses below which risks to populations are assumed not to occur. This definition simplifies those steps in the ecological risk assessment process that involve assessing the relationship between stressor levels and ecological effects, characterizing, estimating, and assessing risks. Caution should be exercised if more restrictive limits are selected, to ensure that the supporting effects data are of high quality, reproducible, and clearly relevant to protection of natural populations. In cases where evaluating dose to individual organisms is needed, you should consider the guidance provided in Module 2, Section 8. The following brief overview of the

ecological risk assessment process emphasizes how the recommended dose limits simplify the risk assessment process for the dose assessment team.

Problem Formulation. In this first phase, the purpose of the dose assessment is clearly defined, the problem is clearly stated, and a plan for analyzing and characterizing risks is developed. As seen in Figure 7.1, available information is integrated to develop a site conceptual model and define assessment endpoints. The analysis plan is derived from the assessment endpoints and conceptual model. As the risk assessment proceeds, assessment endpoints and/or the site conceptual model may be refined, requiring subsequent revisions to the analysis plan.

In the problem formulation phase, the dose assessment team will perform the above steps in much the same way as would an ecological risk-assessment team. For this reason, the dose assessment team should coordinate its activities with other ecological risk assessment efforts

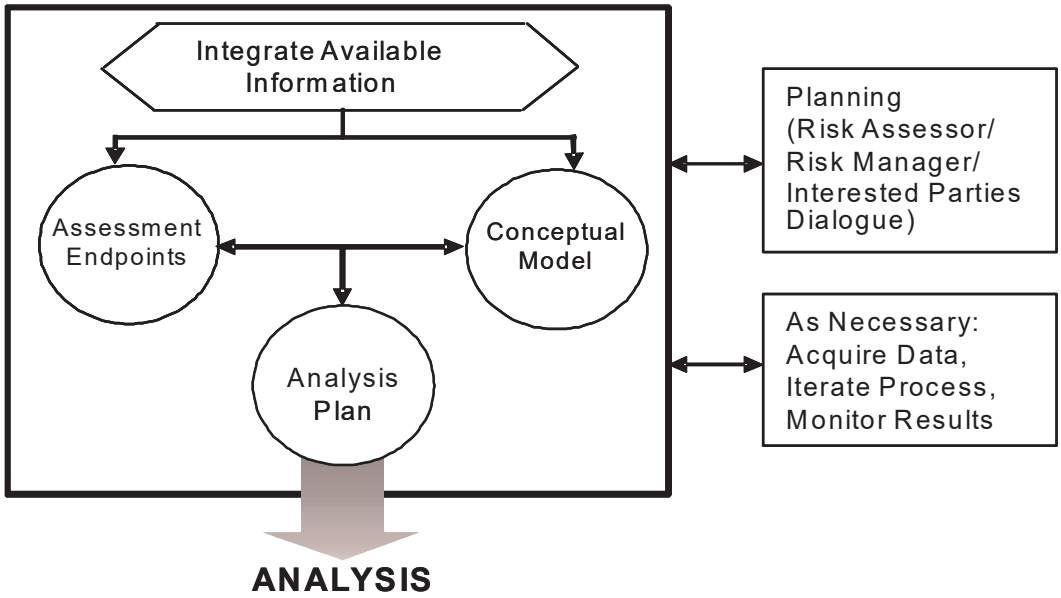

Figure 7.1 Problem Formulation, Phase 1 of Dose Assessment (from EPA 1998)

so that the identification of assessment endpoints and the development of site conceptual models are coordinated. The dose assessment team will, however, need to consider two factors that an ecological risk-assessment team might not. First, the analysis plan should select receptor species resident at the specific site that are known to be radiosensitive. Second, certain considerations are important to collecting biological samples for dosimetric assessments. Collection of biological samples is done to provide more realistic estimates of internal dose to organisms. Considerations for collecting biological samples are reviewed in detail in Module 2, Section 6. Additional considerations for both dose assessments and ecological risk assessments are the movement of receptors into and out of the contaminated area and the distribution of receptors relative to the contaminated area. These considerations

are particularly relevant to motile species, small “hot spots” of contamination, and areas where the concentrations of contaminants vary spatially. In such cases, it may be expedient to better

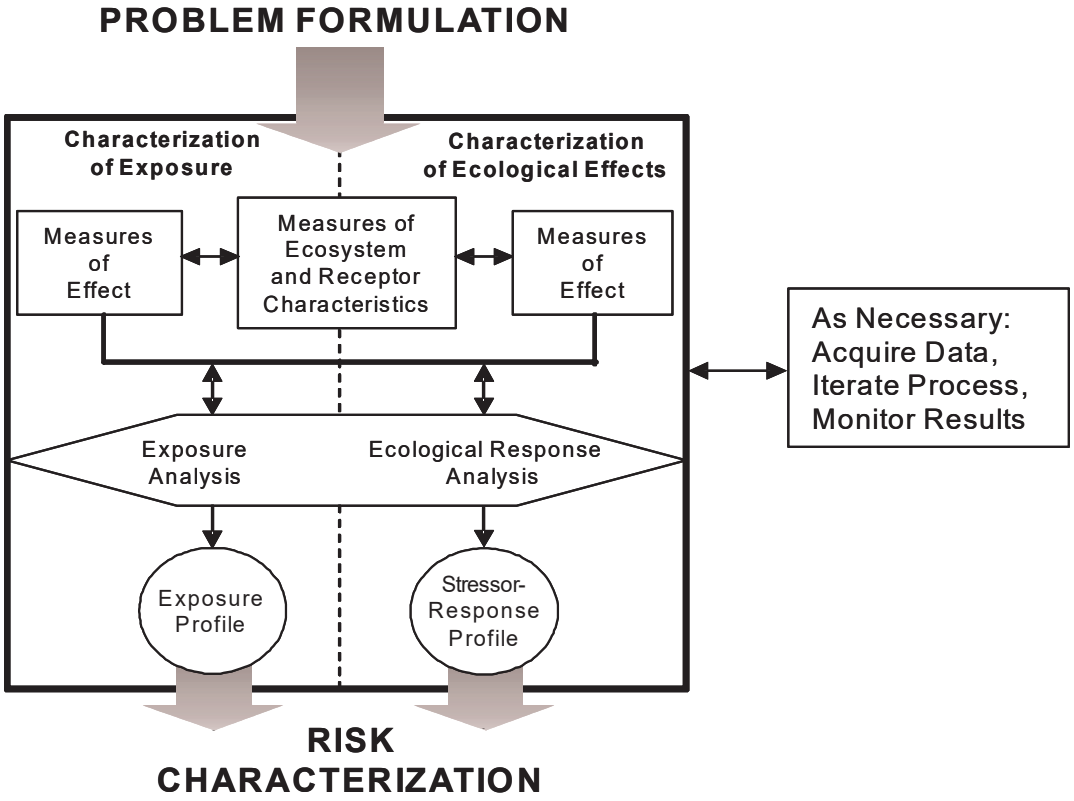

Figure 7.2 Analysis, Phase 2 of Dose Assessment (from EPA 1998)

define the distribution of organisms in time and/or space relative to the contaminated area. For example, individuals of a species may reside year-round within the region but move into and out of the contaminated area, necessitating the collection of data on duration of exposure. Or, ecologically significant species of plants may be located in only one part of the contaminated area and may be exposed to radionuclide concentrations that are above or below mean values for the area. Refer to Module 2, Sections 2 through 5 for detailed guidance in these areas.

Analysis Phase. The exposure profile and stressor-response profile (i.e., ecological effects profile) are estimated during this phase (see Figure 7.2). The dose assessment team should focus on the exposure side of the analysis phase because deleterious effects on receptor populations are assumed not to occur below the recommended limits of 0.1 rad/d or 1.0 rad/d, as appropriate.

In this phase, the dose assessment team should focus on identifying exposure pathways and quantifying exposure. The site conceptual model is the basis for identifying exposure pathways. Quantifying exposure is achieved by assessing the strengths and limitations of the

existing site-specific environmental data on radionuclide contamination, collecting additional supplemental data as needed, and quantitatively analyzing exposure. If supplemental data are needed, the analysis plan may also need to be revised.

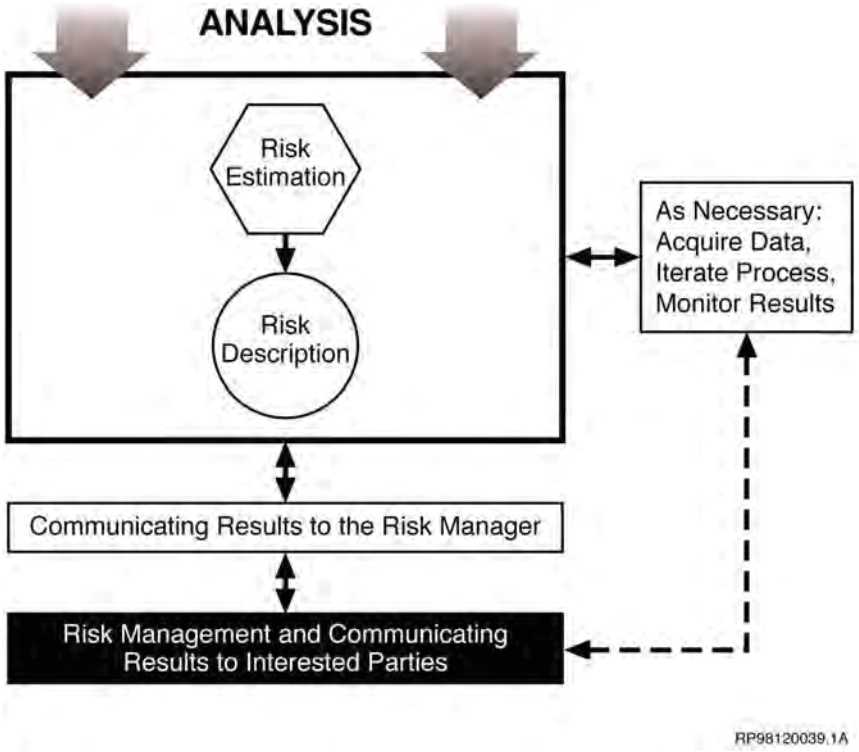

Figure 7.3 Risk Characterization, Phase 3 of Dose Assessment (from EPA 1998)

Risk Characterization. In this phase, doses are estimated and described (see Figure 7.3). The recommended limits again simplify this process since adverse effects on receptor populations are assumed not to occur at exposures below the recommended limits. Plants and animals may also be simultaneously exposed to other stressors, such as noise and hazardous chemicals. At present, no consensus exists within the scientific community about what the cumulative impacts are of simultaneous exposure to ionizing radiation and other anthropogenic stressors, or how to measure them. This factor should be considered when estimating and describing the risks associated with doses of ionizing radiation, if only qualitatively. In cases where exposure of biota to ionizing radiation exceeds the biota dose limits, a consideration of cumulative impacts from radiation and other stressors present may be warranted. Refer to *EPA's Guidance on Cumulative Risk Assessment*, Part 1, Planning and Scoping (EPA 1997b) for an introduction to this topic.

Table 7.1 Aquatic Animal Biota Concentration Guide Spreadsheet. BCGs are for use with radionuclide concentrations from co-located water and sediment. The default lumped parameter values (B_{iv}s) listed here were used to derive the generic BCGs for the general screening phase. These lumped parameter values may be replaced with site-representative values in the site-specific screening component of the analysis phase.

	Derived Concentrations		Bioaccumulation Factor	
Nuclide	BCG (sediment) Bq/kg	BCG (water) Bq/m^3	B_{iv}, Organism to Water (L/kg) Fresh Mass	Water B_{iv} Reference[(a)]
^{241}Am	3E+07	2E+04	400	CRITR
^{144}Ce	1E+06	6E+04	9000	T&M, Table 5.41
^{135}Cs	3E+07	5E+05	22000	T&M, Table 5.41
^{137}Cs	2E+06	4E+04	22000	T&M, Table 5.41
^{60}Co	6E+05	1E+05	2000	T&M, Table 5.41
^{154}Eu	1E+06	8E+05	600	GENII
^{155}Eu	1E+07	1E+07	600	GENII
^{3}H	3E+08	2E+11	0.2	CRITR
^{129}I	2E+07	4E+07	220	T&M, Table 5.41
^{131}I	3E+06	6E+06	220	T&M, Table 5.41
^{239}Pu	3E+08	7E+03	1000	T&M, Table 5.41
^{226}Ra	5E+05	4E+02	3200	T&M, Table 5.41
^{228}Ra	1E+06	3E+02	3200	Based on ^{226}Ra
^{125}Sb	3E+06	1E+07	100	T&M, Table 5.41
^{90}Sr	1E+06	2E+06	320	T&M, Table 5.41
^{99}Tc	2E+07	9E+07	78	T&M, Table 5.41
^{232}Th	1E+08	1E+04	80	T&M, Table 5.41
^{233}U	4E+08	7E+03	1000	T&M, Table 5.41
^{234}U	1E+08	7E+03	1000	T&M, Table 5.41
^{235}U	4E+06	8E+03	1000	T&M, Table 5.41
^{238}U	2E+06	8E+03	1000	T&M, Table 5.41
^{65}Zn	2E+06	7E+04	17000	T&M, Table 5.41
^{95}Zr	9E+05	3E+05	1600	T&M, Table 5.41

(a) T&M = Till and Meyer 1983; GENII = Napier et al. 1988; CRITR = Baker and Soldat 1992

Table 7.2 Part 1 of the Riparian Animal Biota Concentration Guide Spreadsheet. BCGs are for use with radionuclide concentrations from co-located water and sediment. The default lumped parameter values were used to derive the generic BCGs for the general screening phase. These lumped parameter values may be replaced with site-representative values in the site-specific screening component of the analysis phase.

Nuclide	BCG (sediment) Bq/kg	BCG (water) Bq/m^3	Lumped Parameter, Bq/kg (animal - wet weight)	
			per Bq/kg sediment	per Bq/L water
^{241}Am	2E+05	5E+04	3.E-03	1.E+01
^{144}Ce	1E+05	2E+06	5.E-04	3.E+01
^{135}Cs	2E+06	2E+04	3.E-01	5.E+04
^{137}Cs	1E+05	2E+03	3.E-01	5.E+04
^{60}Co	5E+04	2E+05	1.E-02	2.E+02
^{154}Eu	1E+05	2E+06	4.E-03	2.E+01
^{155}Eu	1E+06	3E+07	3.E-03	2.E+01
^{3}H	1E+07	1E+10	4.E-01	8.E-01
^{129}I	1E+06	1E+06	3.E-01	6.E+02
^{131}I	2E+05	5E+05	1.E-01	2.E+02
^{239}Pu	2E+05	2E+04	3.E-03	3.E+01
^{226}Ra	4E+03	2E+02	3.E-02	8.E+02
^{228}Ra	3E+03	1E+02	3.E-02	8.E+02
^{125}Sb	3E+05	2E+08	4.E-04	3.E-01
^{90}Sr	2E+04	1E+04	2.E+00	6.E+03
^{99}Tc	2E+06	2E+07	5.E-02	3.E+01
^{232}Th	5E+04	6E+04	2.E-03	1.E+00
^{233}U	2E+05	3E+04	4.E-03	3.E+01
^{234}U	2E+05	3E+04	4.E-03	3.E+01
^{235}U	1E+05	3E+04	4.E-03	3.E+01
^{238}U	9E+04	3E+04	4.E-03	3.E+01
^{65}Zn	5E+04	5E+02	2.E+00	2.E+05
^{95}Zr	9E+04	1E+06	3.E-03	4.E+01

(a) CRITR = Baker and Soldat 1992; T&M = Till and Meyer 1983; GENII = Napier et al. 1988; W&S = Whicker and Schultz 1982; KAH = K. A. Higley, Oregon State University.

Table 7.3 Terrestrial Plant Biota Concentration Guide Spreadsheet. The default lumped parameter values (B_{iv}s) listed here were used to derive the generic BCGs for the general screening phase. These lumped parameter values may be replaced with site-representative values in the site-specific screening component of the analysis phase.

Nuclide	BCG (soil) Bq/kg	BCG (water) Bq/m^3	Bioaccumulation Factor, B_{iv}	
			Bq/kg (plant - wet weight) per Bq/kg (soil)	Reference[(a)]
^{241}Am	8E+05	3E+10	8E-03	T&M, Table 5.16, Table 5.18
^{144}Ce	5E+05	1E+09	4E-02	T&M, Table 5.16, Table 5.17
^{135}Cs	1E+06	3E+10	1E+01	T&M, Table 5.16, Table 5.17
^{137}Cs	8E+04	2E+09	1E+01	T&M, Table 5.16, Table 5.17
^{60}Co	2E+05	6E+08	2E-01	T&M, Table 5.16, Table 5.17
^{154}Eu	5E+05	1E+09	4E-02	Estimated from Ce value by KAH
^{155}Eu	6E+06	1E+10	4E-02	Estimated from Ce value by KAH
^{3}H	6E+07	3E+11	1E+00	Estimated from NUREG 1.109
^{129}I	6E+06	2E+10	4E-01	T&M, Table 5.16, Table 5.17
^{131}I	9E+05	3E+09	4E-01	T&M, Table 5.16, Table 5.17 for ^{129}I
^{239}Pu	5E+05	3E+11	1E-02	T&M, Table 5.16, Table 5.18
^{226}Ra	1E+04	5E+08	1E-01	T&M, Table 5.16, Table 5.18
^{228}Ra	9E+03	1E+09	1E-01	T&M, Table 5.16, Table 5.18 from ^{226}Ra
^{125}Sb	1E+06	3E+09	1E-02	From GENII, Food Transfer Library
^{90}Sr	1E+05	1E+09	4E+00	T&M, Table 5.16, Table 5.17
^{99}Tc	8E+05	2E+10	8E+00	From GENII, Food Transfer Library
^{232}Th	9E+05	1E+11	1E-03	T&M, Table 5.16, Table 5.18
^{233}U	2E+06	4E+11	4E-03	T&M, Table 5.16, Table 5.18 from ^{238}U
^{234}U	2E+06	1E+11	4E-03	T&M, Table 5.16, Table 5.18 from ^{238}U
^{235}U	1E+06	4E+09	4E-03	T&M, Table 5.16, Table 5.18 from ^{238}U
^{238}U	6E+05	2E+09	4E-03	T&M, Table 5.16, Table 5.18
^{65}Zn	9E+05	2E+09	3E-01	T&M, Table 5.16, Table 5.17
^{95}Zr	4E+05	9E+08	3E-02	T&M, Table 5.16, Table 5.17

(a) T&M = Till and Meyer 1983; GENII = Napier et al. 1988; KAH = K. A. Higley, Oregon State University.

Table 7.4 Part 1 of the Terrestrial Animal Biota Concentration Guide Spreadsheet. The default lumped parameter values listed here were used to derive the generic BCGs for the general screening phase. These lumped parameter values may be replaced with site-representative values in the site-specific screening component of the analysis phase.

Nuclide	BCG (soil) Bq/kg	BCG (water) Bq/m^3	Lumped Parameters	
			Terrestrial Animal to Soil Bq/kg (animal - wet wt.) / Bq/kg (soil)	Terrestrial Animal to Water Bq/kg (animal - wet wt.)/Bq/L (water)
^{241}Am	1E+05	7E+06	4E-03	9E-02
^{144}Ce	5E+04	1E+08	6E-03	8E-03
^{135}Cs	1E+04	3E+08	1E+02	3E+00
^{137}Cs	8E+02	2E+07	1E+02	3E+00
^{60}Co	3E+04	4E+07	8E-02	1E-01
^{154}Eu	5E+04	8E+07	5E-03	1E-01
^{155}Eu	6E+05	1E+09	4E-03	9E-02
^{3}H	6E+06	9E+09	1E+00	1E+00
^{129}I	2E+05	2E+08	3E+00	3E+00
^{131}I	3E+04	7E+07	3E+00	1E+00
^{239}Pu	2E+05	7E+06	3E-03	9E-02
^{226}Ra	2E+03	3E+05	6E-02	4E-01
^{228}Ra	2E+03	3E+05	6E-02	4E-01
^{125}Sb	1E+05	3E+08	4E-04	5E-03
^{90}Sr	8E+02	2E+06	8E+01	3E+01
^{99}Tc	2E+05	6E+08	3E+00	8E-01
^{232}Th	6E+04	2E+06	2E-03	5E-02
^{233}U	2E+05	1E+07	4E-03	5E-02
^{234}U	2E+05	1E+07	4E-03	5E-02
^{235}U	1E+05	2E+07	4E-03	5E-02
^{238}U	6E+04	2E+07	4E-03	5E-02
^{65}Zn	2E+04	6E+06	7E+00	2E+01
^{95}Zr	4E+04	8E+07	4E-03	3E-02

Table 7.5 Part 2 of the Riparian Animal Biota Concentration Guide Spreadsheet. The default parameter values listed here relating to the uptake, retention and biological decay rates of radionuclides for a riparian animal receptor may be replaced with site-representative values in the site-specific analysis component of the analysis phase.

Nuclide	Fraction of Intake Retained f_1 (unitless)	Biological Decay Constant λ_{bio} (d^{-1})	$\lambda bio = ln(2)/(aW^b)$		λ_{bio} (d^{-1}) Reference[(a)]
			a (constant)	b (constant)	
^{241}Am	1E-03	5.53E-04	0.8	0.81	ICRP 30, Part 4
^{144}Ce	3E-04	3.46E-04	1.4	0.8	ICRP 30, Part 1
^{135}Cs	1E+00	2.24E-02	3.5	0.24	W&S
^{137}Cs	1E+00	2.24E-02	3.5	0.24	W&S
^{60}Co	5E-02	3.01E-02	2.6	0.24	W&S
^{154}Eu	1E-03	3.46E-04	1.4	0.8	ICRP 30, Part 3
^{155}Eu	1E-03	3.46E-04	1.4	0.8	ICRP 30, Part 3
^{3}H	1E+00	5.72E-03	0.82	0.55	W&S
^{129}I	1E+00	3.13E-02	6.8	0.13	W&S
^{131}I	1E+00	3.13E-02	6.8	0.13	W&S
^{239}Pu	1E-03	5.53E-04	0.8	0.81	ICRP 30, Part 4
^{226}Ra	2E-01	3.58E-02	2	0.25	Estimated by KAH
^{228}Ra	2E-01	3.58E-02	2	0.25	Estimated by KAH
^{125}Sb	1E-02	1.43E-01	0.5	0.25	ICRP 30, Part 3
^{90}Sr	3E-01	6.11E-04	107	0.26	W&S
^{99}Tc	8E-01	6.11E-02	0.3	0.4	ICRP 30, Part 2
^{232}Th	2E-04	1.34E-04	3.3	0.81	ICRP 30, Part 1
^{233}U	5E-02	6.81E-02	0.8	0.28	ICRP 30, Part 1
^{234}U	5E-02	6.81E-02	0.8	0.28	ICRP 30, Part 1
^{235}U	5E-02	6.81E-02	0.8	0.28	ICRP 30, Part 1
^{238}U	5E-02	6.81E-02	0.8	0.28	ICRP 30, Part 1
^{65}Zn	5E-01	7.16E-04	100	0.25	ICRP 30, Part 2
^{95}Zr	2E-03	7.16E-04	100	0.25	ICRP 30, Part 1

(a) CRITR = Baker and Soldat 1992; T&M = Till and Meyer 1983; GENII = Napier et al. 1988; W&S = Whicker and Schultz 1982; KAH = K. A. Higley, Oregon State University.

Table 7.6 Part 3 of the Riparian Animal Biota Concentration Guide Spreadsheet. Riparian animal default kinetic/allometric relationship parameter values listed here may be replaced with site-representative values in the site-specific analysis component of the analysis phase.

Parameter	Equation	Descriptions	Value(s)	Reference
W		Body mass (g)	8800	default for raccoon or river otter
r	$r = \frac{a}{dc}\ 70\ M^{0.75}$	Food intake rate (g/d)	325.1377223	W&S, Vol. II, p. 43, equation 78
		a: ratio of active to basal metabolic rate	2	
		70: constant	70	
		d: fraction of energy ingested that is assimilated or oxidized	0.44	
		c: caloric value of food, kcal/g	5	
		M: body mass in kg (=W*0.001)	8.8	
		0.75: exponent in calculation	0.75	
$r_{sediment}$	$r_{sediment} = 0.1\ r$	Sediment Intake Rate (g/d)	32.5137723	EPA Wildlife Exposure Factor Handbook, Vol. 1, p. 4-22
		r: food intake rate, g/d	325.1377223	
		0.1: fraction of sediment in diet, expressed as % of food diet, dry wt.	0.1	
T_{ls}	$T_{ls,\max} = 1.02 M^{0.30}$	Maximum Lifespan	1.958596497	Calder, p. 316, Table 11-5
		1.02: constant in equation	1.02	
		M = body mass in kg	8.8	
		0.30: exponent in calculation	0.30	
r_b	$r_b = 0.481 M^{0.76}$	Inhalation rate (m^3/d)	2.511608286	Pedley, p. 15, Table V., adjusted to provide units of m^3/d
		0.481: constant in calculation to give m^3/d	0.481	
		M = body mass in kg	8.8	
		0.76: exponent in equation	0.76	
$r_{inhalation}$	$r_{inhalation} = x\ r_b$	Sediment inhalation rate (g/d)	0.000251161	derived
		x: airborne dust loading, g/m^3	0.0001	
		r_b: inhalation rate (see above)	2.511608286	
I_w	$I_w = 0.099 M^{0.90}$	Water consumption rate (L/d)	0.700921852	EPA Wildlife Exposure Factor Handbook, Vol. 1, p. 3-10, equation 3-17
		0.099: constant in equation	0.099	
		see above equation, M: body mass in kg (=W*0.001)	8.8	
		0.9: exponent in calculation	0.9	

Table 7.7 Part 2 of the Terrestrial Animal Biota Concentration Guide Spreadsheet. The default parameter values listed here relating to the uptake, retention, and biological decay rates of radionuclides for a terrestrial animal receptor may be replaced with site-representative values in the site-specific analysis component of the analysis phase.

Nuclide	Fraction of Intake Retained f_1 (unitless)	Biological Decay Constant λ_{bio}, d^{-1}	$\lambda_{bio} = \ln(2) / (aW^b)$		λ_{bio} (d^{-1}) Reference [a]	PT/IT [b]
			a (constant)	b (exponent)		
^{241}Am	1E-03	7.09E-02	0.8	0.81	ICRP 30 Part 4	250
^{144}Ce	3E-04	4.18E-02	1.4	0.8	ICRP 30 Part 1	16
^{135}Cs	1E+00	9.43E-02	3.5	0.24	Whicker & Schultz	0.8
^{137}Cs	1E+00	9.43E-02	3.5	0.24	Whicker & Schultz	0.8
^{60}Co	5E-02	1.27E-01	2.6	0.24	Whicker & Schultz	7
^{154}Eu	1E-03	4.18E-02	1.4	0.8	ICRP 30 Part3	30
^{155}Eu	1E-03	4.18E-02	1.4	0.8	ICRP 30 Part 3	30
^{3}H	1E+00	1.54E-01	0.82	0.55	Whicker & Schultz	1
^{129}I	1E+00	6.82E-02	6.8	0.13	Whicker & Schultz	0.7
^{131}I	1E+00	6.82E-02	6.8	0.13	Whicker & Schultz	0.7
^{239}Pu	1E-03	7.09E-02	0.8	0.81	ICRP 30 Part 3	4000
^{226}Ra	2E-01	1.6E-01	2	0.25	Estimated by KAH	3
^{228}Ra	2E-01	1.6E-01	2	0.25	Estimated by KAH	3
^{125}Sb	1E-02	6.40E-01	0.5	0.25	ICRP 30 Part 3	3.5
^{90}Sr	3E-01	2.90E-03	107	0.26	Whicker & Schultz	200
^{99}Tc	8E-01	6.71E-01	0.3	0.4	ICRP 30 Part 2	5
^{232}Th	2E-04	1.72E-02	3.3	0.81	ICRP 30 Part 1	750
^{233}U	5E-02	3.65E-01	0.8	0.28	ICRP 30 Part 1	7000
^{234}U	5E-02	3.65E-01	0.8	0.28	ICRP 30 Part 1	7000
^{235}U	5E-02	3.65E-01	0.8	0.28	ICRP 30 Part 1	3500
^{238}U	5E-02	3.65E-01	0.8	0.28	ICRP 30 Part 1	4000
^{65}Zn	5E-01	3.20E-03	100	0.25	ICRP 30 Part 2	1
^{95}Zr	2E-03	3.20E-03	100	0.25	ICRP 30 Part 1	10

(a) KAH = K.A. Higley, Oregon State University; Whicker and Schultz, 1982; (b) PT/IT = Factor used in assessing the relative contribution to internal dose from animal inhalation versus ingestion.

Table 7.8 Part 3 of Terrestrial Animal Biota Concentration Guide Spreadsheet. Terrestrial animal default kinetic/allometric relationship parameter values listed here may be replaced with site-representative values in the site-specific analysis component of the analysis phase.

Parameter	Equation	Descriptions	Value(s)	Reference
W		Body mass (g)	22	default for deer mouse
r	$r = \frac{a}{dc}\ 70\ M^{0.75}$	Food intake rate (g/d)	3.635150245	W&S, Vol. II, p. 43, equation 78
		a: ratio of active to basal metabolic rate	2	
		70: constant	70	
		d: fraction of energy ingested that is assimilated or oxidized	0.44	
		c: caloric value of food, kcal/g	5	
		M: body mass in kg (=W*0.001)	0.022	
		0.75: exponent in calculation	0.75	
r_{soil}	$r_{soil} = 0.1\ r$	Soil Intake Rate (g/d)	0.363515025	EPA Wildlife Exposure Factor Handbook, Vol. 1, p. 4-22
		r: food intake rate, g/d	3.635150245	
		0.1: amount of soil in diet, expressed as fraction of food diet, dry wt.	0.1	
T_{ls}	$T_{ls,\max,} = 1.02M^{0.30}$	Maximum Lifespan	.324583901	Calder, p. 316, Table 11-5
		1.02: constant in equation	1.02	
		see above equation, M: body mass in kg (=W*0.001)	0.022	
		0.30: exponent in calculation	0.30	
r_b	$r_b = 0.481M^{0.76}$	Inhalation rate (m^3/d)	0.026447603	Pedley, p. 15, Table V., adjusted to provide units of m^3/d
		0.481: constant in calculation to give m^3/d	0.481	
		M = body mass in kg	0.022	
		0.76: exponent in equation	0.76	
$r_{inhalation}$	$r_{inhalation} = x\ r_b$	Soil inhalation rate (g/d)	2.64476E-06	derived
		x: airborne dust loading, g/m^3	0.0001	
		r_b: inhalation rate (see above)	0.026447603	
I_w	$I_w = 0.099M^{0.90}$	Water consumption rate (L/d)	0.003190183	EPA Wildlife Exposure Factor Handbook, Vol. 1, p. 3-10, equation 3-17
		0.099: constant in equation	0.099	
		see above equation, M: body mass in kg (=W*0.001)	0.022	
		0.9: exponent in calculation	0.9	

Table 7.9 Part 2 of Dose Factors and Common Parameters Spreadsheet. Provides a reference source for all default internal dose conversion factors, and external dose conversion factors for water, sediment, and soil used in deriving the generic BCGs. These values, together with measured radionuclide concentrations in water, sediment, soil, and biota tissue data, can be used to estimate a dose to a receptor.

Nuclide	Dose Conversion Factors			
	Internal Dose	External Dose		
	Gy/y per Bq/kg (wet)	Water Gy/y per Bq/m^3	Sediment Gy/y per Bq/kg (dry)	Soil Gy/y per Bq/kg (dry)
^{241}Am	5.6E-04	1.4E-10	1.4E-07	2.9E-07
^{144}Ce	6.8E-06	3.4E-09	3.4E-06	6.8E-06
^{135}Cs	3.4E-07	1.4E-10	1.4E-07	2.8E-07
^{137}Cs	4.3E-06	2.0E-09	2.0E-06	4.0E-06
^{60}Co	1.3E-05	6.6E-09	6.6E-06	1.3E-05
^{154}Eu	7.6E-06	3.8E-09	3.8E-06	7.7E-06
^{155}Eu	6.2E-07	3.1E-10	3.1E-07	6.2E-07
^{3}H	2.9E-08	1.4E-11	1.4E-08	2.9E-08
^{129}I	4.5E-07	2.0E-10	2.0E-07	4.0E-07
^{131}I	2.9E-06	1.4E-09	1.4E-06	2.9E-06
^{239}Pu	5.3E-04	1.4E-11	1.4E-08	2.8E-08
^{226}Ra	3.0E-03	6.8E-09	6.8E-06	1.4E-05
^{228}Ra	3.6E-03	3.4E-09	3.4E-06	6.9E-06
^{125}Sb	2.7E-06	1.4E-09	1.4E-06	2.9E-06
^{90}Sr	5.7E-06	2.8E-09	2.8E-06	5.7E-06
^{99}Tc	5.1E-07	2.1E-10	2.1E-07	4.3E-07
^{232}Th	4.1E-03	3.0E-11	3.0E-08	6.1E-08
^{233}U	4.9E-04	9.3E-12	9.3E-09	1.9E-08
^{234}U	4.9E-04	3.2E-11	3.2E-08	6.5E-08
^{235}U	4.5E-04	9.4E-10	9.4E-07	1.8E-06
^{238}U	4.4E-04	2.3E-09	2.3E-06	4.6E-06
^{65}Zn	3.0E-06	1.5E-09	1.5E-06	3.0E-06
^{95}Zr	8.4E-06	4.2E-09	4.2E-06	8.4E-06

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8 Documenting Your Biota Dose Evaluation Results

At a minimum, your results shall be documented in your Annual Site Environmental Report (DOE 2000b). The following information shall be summarized in the Annual Site Environmental Report, and described in more detail within a report retained on file for future reference:

- Specify the biota dose limits being complied with (e.g., 1 rad/d for aquatic animals; DOE Order 5400.5).

- Identify the methods used to demonstrate compliance with these limits. Cite the method used (e.g., this technical standard). Describe the process used (e.g., general screening phase, site-specific analysis, actual biota dose assessment involving the collection and analysis of biota).

Printing the Results of Your Biota Dose Evaluation using the RAD-BCG Calculator

Clicking on the "Set Print Area for Report" button at the bottom of the Aquatic or Terrestrial System Data Entry/BCG Worksheets, then pressing the printer icon in the toolbar, will print out a record of your biota dose evaluation. Sum of fraction totals, limiting organism types, and any changes you made to default parameters will be included.

- Describe the area(s) of evaluation, sources of exposure, organism types, media types, and radionuclide data used in the evaluation.

- Summarize the results (e.g., sum of fractions for media and radionuclides are less than 1; doses calculated are less than biota dose limits) for the site area(s) of evaluation; and conclusions.

- Summarize why the evaluation was conducted, and how the results will be used (e.g., to demonstrate compliance with DOE dose limits, for use in outreach activities, in response to stakeholder or regulator requests, or for use in an eco-risk assessment.)

- All detailed information used in calculations (e.g., site-specific parameters selected and the rationale for their use) shall be described and retained on file for future reference.

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9 Example Applications of the Graded Approach

9.1 Generic Example of an Aquatic System Evaluation

This example was prepared using actual measured radionuclide concentration data from a DOE site. However, the data is used within a hypothetical context for a generic site (e.g., Poplar Springs Site, a hypothetical site). Two cases are provided, drawing from the same data set of measured radionuclide concentrations from surface water samples. The first case considers the entire Poplar Springs Site as the evaluation area, and options for proceeding when the Site fails a general screening evaluation. The second case begins with the goal of assessing several evaluation areas independently within the boundary of the Poplar Springs Site. The cases are intended only to highlight key steps and concepts of the graded approach, and to highlight several alternatives within each that would also result in a determination of protection relative to Dose Rate Guidelines.

Purpose:

The purpose of the evaluation was to demonstrate that the Poplar Springs Site (PSS) is in compliance with DOE's biota dose limit for aquatic animals pursuant to DOE Order 5400.5 II 3.a.(c)(5): "to protect native animal aquatic organisms, the absorbed does to these organisms shall not exceed 1 rad/d from exposure to the radioactive material in liquid wastes discharged to natural waterways."

1. Data Assembly *(Phase 1 of the Graded Approach)*:

A. Verify Data is Appropriate for a Biota Dose Evaluation

Surface water samples are collected and analyzed to assess the impact of past and current DOE operations on the quality of local surface water. Sampling locations include streams within the main plant area and at downstream locations from Poplar Springs Site (PSS) facilities; all are within the PSS boundary. These sampling stations are located within the Blue Falls Creek Watershed (main plant and down stream locations) and within other smaller watersheds, all of which flow into the Darlington River. Surface water data (via the surface water surveillance program) are collected throughout the year. The sampling frequency is dependant on historical data and the processes or legacy activities nearby or upstream from these locations. Therefore, sampling occurs at different locations monthly, bimonthly, quarterly, or semiannually. The sampling locations are presented in Table 1.

Table 1 Surface Water Sampling Locations for the Poplar Springs Site

Watershed	Sampling Locations
Blue Falls Creek	
Main Plant–On-site Stream Locations:	Two Falls Creek TFCK 0.5
	Broad Creek BRCK
	Northwest Tributary NWTK 0.5
Downstream Locations:	Muddy Branch MB 0.6
	Blue Falls Creek BFCK 3.0
	Blue Falls Creek at Blue Falls Dam BFCK 1.4
Other Watersheds Entering the Darlington River	Taylor's Creek TCK 1.0
	Beaver Creek BVCK 2.3

B. Request Sampling Data, to Include Maximum and Mean Water and Sediment Radionuclide Concentrations (co-located if possible) Collected for the Environmental Monitoring and Surveillance Program at Poplar Springs Site

Environmental surveillance surface water monitoring results were available. However, no on-site sediment data (co-located with water sampling stations) were available. The data were organized by collection location and summarized in a table for future use (Table 2). It was determined that the sampling locations indicated in Table 2 were each representative of individual evaluation areas within the larger Poplar Springs Site. Each of the evaluation areas were identified because they provide a good indication of potential impacts to biota in natural waterways within the Poplar Springs Site.

Table 2 Measured Radionuclide Concentrations (pCi/L) in Surface Water Collected from the Poplar Springs Site. Maximum, minimum, and average values are summarized. The maximum measured radionuclide concentrations observed for the Poplar Springs Site (i.e., across all sampling locations) are indicated by an (*).

Sampling Location	Radionuclide	Maximum	Minimum	Average
Main Plant: On-site station locations:				
Two Falls Creek (TFCK 0.5)	H-3	530	430	480
	Sr	15	15	15
Broad Creek (BRCK)	H-3	360	110	240
	Sr	290	59	170
	*U-234	36	7.7	22
	U-235	0.048	0	0.024
	U-238	0.52	0.28	0.40
Northwest Tributary (NWTK 0.5)	H-3	160	110	140
	Sr	71	1.8	36
Downstream Locations:				
Muddy Branch (MB 0.6)	*Co-60	4.6	-2.8	2.0
	Cs-137	3.0	0.0050	1.5
	*H-3	760,000	39,000	460,000
	*Sr	460	84	250
	U-234	0.52	0.15	0.33
	U-238	0.50	0.15	0.37
Blue Falls Creek (BFCK 3.0)	Co-60	1.5	0.034	0.79
	*Cs-137	67	12	37
	H-3	36,000	3,300	17,000
	Sr	330	28	100
	U-234	4.8	1.2	3.5
	*U-235	0.075	0	0.024
	*U-238	2.1	0.24	0.98
Blue Falls Creek at Blue Falls Dam (BFCK 1.4)	Co-60	3.9	0.58	2.5
	Cs-137	40	8.5	12
	H-3	140,000	32,000	71,000
	Sr	140	54	100
	U-234	8.2	1.6	5.0
	U-235	0.065	0	0.029
	U-238	1.6	0.41	0.95
Other watersheds entering the Darlington River:				
Taylor's Creek (TCK 1.0)	Co-60	3.2	0.64	1.9
Beaver Creek (BVCK 2.3)	Co-60	1.8	1.6	1.7
	H-3	330	180	260
	Sr	43	4.8	24

CASE 1. Use of Maximum Measured Radionuclide Concentrations for the Entire Poplar Springs Site

1. General Screening Evaluation *(Phase 2 of the Graded Approach)*

A. Enter Data into the RAD-BCG Calculator

Maximum measured radionuclide concentration data for surface water detected for the entire Poplar Springs Site (i.e., the radionuclide-specific maximum values detected across the entire Site) were entered into the Aquatic System Data Entry/BCG Worksheet within the RAD-BCG Calculator. The RAD-BCG Calculator automatically calculated the missing sediment radionuclide concentration data (e.g., by using the "most probable" radionuclide-specific K_d values) and entered the calculated radionuclide concentrations into the appropriate fields.

B. Compare Measured Radionuclide Concentrations in Environmental Media with BCGs

The RAD-BCG Calculator automatically calculated the radionuclide-specific partial sum of fractions for water and sediment, then calculated the total sum of fractions. A summary of the comparisons for each medium and radionuclide (which is similar in presentation to what you would see in the Aquatic System Data Entry/BCG Worksheet) is provided in Table 3. Note that this comparison could also be done manually by using Tables 6.1 - 6.2 and associated guidance contained in Module 1 of the DOE technical standard. The results indicated that the Poplar Springs Site failed the general screening evaluation using maximum radionuclide concentration data. Results also indicated that the water medium appears to be limiting (see partial sum of fractions for water and sediment, respectively, in Table 3). In addition, Cs-137 and Sr-90 were the radionuclides that provided the greatest contribution to the total sum of fractions (i.e., they were the most limiting radionuclides, providing the greatest contribution to potential dose). A riparian animal was indicated as the limiting organism type for these radionuclides.

Table 3 Aquatic System Evaluation: General Screening Results for Poplar Springs Site using Maximum Measured Radionuclide Concentrations in Surface Water Across the Entire Site

Radionuclide	Maximum Measured Radionuclide Concentrations (pCi/L)	Water Sum of Fractions	Sediment Sum of Fractions
H-3	760,000	2.9E-03	2.03E-06
Sr-90	460	1.70	2.37E-02
U-234	36	1.8E-01	3.42E-04
U-235	0.075	3.4E-04	1.01E-06
U-238	2.1	9.4E-03	4.22E-05
Co-60	4.6	1.2E-03	3.14E-03
Cs-137	67	1.6	1.07E-02
Total of partial sum of fractions for each medium		3.42	3.80E-02
Total sum of fractions for all radionuclides and media			3.45

2. Site-Specific Screening using Mean Radionuclide Concentrations in Place of Maximum Values *(Phase 3 of the Graded Approach: Analysis Phase, Site-Specific Screening)*

It was determined through consultation with site environmental surveillance program personnel that the quality and quantity of data allowed for averaging of measured radionuclide concentration data by individual sampling location for the Poplar Springs Site, but not across the entire Site. Guidance provided in Module 2, Section 3 of the DOE technical standard concerning spatio-temporal averaging, and guidance provided in Module 2, Section 4 concerning the definition of an evaluation area was reviewed. It was determined that - although the habitats and presence of the limiting organism type (in this case a riparian animal) were similar across all sampling locations, radionuclide data could not be averaged across the entire Poplar Springs Site because: (1) the site was too large for such an averaging scheme to be sensible, and (2) the contamination profiles (e.g., the radionuclides detected and their levels) for Main Plant - on-site locations, downstream locations, and other streams that enter the Darlington River were too different from one another (see Table 2). However, it was determined that within the downstream locations, data from Blue Falls Creek (BFCK 3.0) and Blue Falls Creek at Blue Falls Dam (BFCK 1.4) station locations could be averaged over space and time, because of their proximity to each other (e.g., both stations are in the same water system), and because the contamination profiles, habitats, and limiting organism type (riparian animal) were determined to be similar across the areas represented by these sampling locations. Therefore, measured radionuclide concentrations for these two locations were averaged for subsequent use in site-specific screening. Measured radionuclide concentrations for each of the remaining sampling locations were averaged by location, consistent with advice from the Site environmental surveillance program personnel.

A. Enter Data into the RAD-BCG Calculator

The averaging scheme presented above resulted in the need for seven separate evaluations: one for each of the six individual sampling locations, and one for the combined Blue Falls Creek / Blue Falls Creek at Blue Falls Dam locations. For each evaluation, mean measured radionuclide concentration data for surface water were entered into the Aquatic System Data Entry/BCG Worksheet within the RAD-BCG Calculator. The RAD-BCG Calculator automatically calculated the missing sediment radionuclide concentration data (e.g., by using the “most probable” radionuclide-specific K_d values) and entered the calculated radionuclide concentrations into the appropriate fields.

B. Compare Measured Radionuclide Concentrations in Environmental Media with BCGs

The RAD-BCG Calculator automatically calculated the radionuclide-specific partial sum of fractions for water and sediment, then calculated the total sum of fractions. A summary of the comparisons for each location is provided in Table 4. The results indicated that all of the sampling locations, each representing an individual evaluation area, passed the site-specific screening.

Table 4 Aquatic System Evaluation: Site-Specific Screening Results using Mean Radionuclide Concentrations in Surface Water for Each Evaluation Area

Sampling Location	Average Concentrations Sum of Fractions < 1.0 (Pass/Fail)?	Water Sum of Fractions	Sediment Sum of Fractions	Total Sum of Fractions
Main Plant - On-site Locations:				
Two Falls Creek (TFCK 0.5)	passed	5.39E-02	7.73E-04	0.055
Broad Creek (BRCK)	passed	7.21E-01	8.98E-03	0.73
Northwest Tributary (NWTK 0.5)	passed	1.29E-01	1.86E-03	0.13
Downstream Locations:				
Muddy Branch (MB 0.6)	passed	9.38E-01	1.45E-02	0.95
Blue Falls Creek (BFCK 3.0) and Blue Falls Creek at Blue Falls Dam Station (BFCK 1.4) (combined)	passed	0.96	1.03E-02	0.97
Other Streams that enter Darlington River:				
Taylor's Creek (TCK 1.0)	passed	5.05E-04	1.3E-03	0.002
Beaver Creek (BVCK 2.3)	passed	8.66E-02	2.4E-03	0.089

3. Documentation of Results

The results of the biota dose evaluation were summarized. A summary report which contains computer screen printouts of the spreadsheet results from the RAD-BCG Calculator were retained on file for future reference. The rationale for using average radionuclide concentration values in place of maximum values was documented. As required by EH, a summary of the evaluation was included in the Poplar Springs Site's Annual Site Environmental Report.

4. Lessons Learned

- All of the downstream station locations corresponding to individual evaluation areas provided the greatest total sums of fractions. These are clearly good indicator locations for future biota dose evaluations.

- All of the evaluation areas passed. However, because the total sum of fractions for each of the downstream locations was very near 1.0, we could consider conducting additional analysis on these evaluation areas using the analysis phase of the graded approach (refer to the example provided in CASE 2).

- Possible future activities could include: (1) assessing the need for additional sampling locations; (2) collecting co-located sediment and water samples for these and other locations; (3) collecting representative receptors and analyzing tissue data to permit a direct and more realistic dose evaluation.

CASE 2. Evaluation of Several Evaluation Areas Using Maximum Measured Radionuclide Concentration Data

1. General Screening Evaluation *(Phase 2 of the Graded Approach)*

A. Enter Data into the RAD-BCG Calculator

Maximum measured radionuclide concentration data for surface water for each sampling location (each representative of individual evaluation areas) were entered into the Aquatic System Data Entry/BCG Worksheet within the RAD-BCG Calculator (i.e., in this case, eight individual evaluations, one for each sampling location representative of an evaluation area, were conducted). The RAD-BCG Calculator automatically calculated the missing sediment radionuclide concentration data (e.g., by using the "most probable" radionuclide-specific K_d values) and entered the calculated radionuclide concentrations into the appropriate fields.

B. Compare Measured Radionuclide Concentrations in Environmental Media with BCGs

The RAD-BCG Calculator automatically calculated the radionuclide-specific partial sum of fractions for water and sediment, then calculated the total sum of fractions. A summary of the comparisons for each location is provided in Table 5. The results indicated that four of the locations evaluated (Broad Creek, Muddy Branch, Blue Falls Creek, and Blue Falls Creek at Blue Falls Dam) failed the general screening evaluation using maximum radionuclide concentration data. Results also indicated that the water medium is limiting (see partial sum of fractions for water and sediment, respectively, in Table 5). It was also determined that Cs-137 and Sr-90 were the radionuclides that provided the greatest contribution to the total sum of fractions (i.e., they were the most limiting radionuclides, providing the greatest contribution to potential dose). A riparian animal was the limiting organism type for these radionuclides.

Table 5 Aquatic System Evaluation: General Screening Results for Poplar Springs Site Using Maximum Measured Radionuclide Concentrations in Surface Water

Sampling Locations	Sum of Fractions < 1.0 (Pass/Fail?) Using Maximum Concentrations	Water Sum of Fractions	Sediment Sum of Fractions	Total Sum of Fractions
Main Plant--On-site Locations:				
Two Falls Creek (TFCK 0.5)	passed	5.39E-02	7.7E-04	0.05
Broad Creek (BRCK)	failed	1.22	1.53E-02	1.24
Northwest Tributary (NWTK 0.1)	passed	2.55E-01	3.66E-03	0.26
Downstream Locations:				
Muddy Branch (MB 0.6)	failed	1.73	2.73E-02	1.76
Blue Falls Creek (BFCK 3.0)	failed	2.79	2.88E-02	2.82
Blue Falls Creek at Blue Falls Dam (BFCK 1.4)	failed	1.49	1.64E-02	1.51
Other Streams that enter Darlington River:				
Taylor's Creek (TCK 1.0)	passed	8.51E-04	2.19E-03	0.003
Beaver Creek (BVCK 2.3)	passed	1.55E-01	3.45E-03	0.16

2. Site-Specific Screening using Mean Radionuclide Concentrations in Place of Maximum Values ***(Phase 3 of the Graded Approach: Analysis Phase, Site-Specific Screening)***

It was determined through consultation with Site environmental surveillance program personnel that the quality and quantity of data provided for time averaging of measured radionuclide concentration data for each individual evaluation area. Guidance provided in Module 2, Section 2 of the DOE technical standard concerning spatio-temporal averaging was also consulted.

A. Enter Data into the RAD-BCG Calculator

Mean radionuclide concentration data for surface water from each of the four sampling locations which failed the general screening phase were entered into the Aquatic System Data Entry/BCG Worksheet within the RAD-BCG Calculator (i.e., four separate evaluations were conducted). The RAD-BCG Calculator automatically calculated the missing sediment radionuclide concentration data (e.g., by using the "most probable" radionuclide-specific K_d values) and entered the calculated sediment radionuclide concentrations into the appropriate fields.

B. Compare Measured Radionuclide Concentrations in Environmental Media with BCGs

The RAD-BCG Calculator automatically calculated the radionuclide-specific partial sum of fractions for water and sediment, then calculated the total sum of fractions. A summary of the comparisons for each location is provided in Table 6. The results indicated that of the four locations evaluated (Broad Creek, Muddy Branch, Blue Falls Creek, and Blue Falls Creek at Blue Falls Dam), Broad Creek, Muddy Branch, and Blue Falls Creek at Blue Falls Dam passed the site-specific screening evaluation using mean radionuclide concentration data. Results also indicated that for the remaining location (Blue Falls Creek - which did not pass the screen), the water medium is limiting (see partial sum of fractions for water and sediment, respectively, in Table 6). It was also determined that Cs-137 and Sr-90 were the radionuclides that provided the greatest contribution to the total sum of fractions (i.e., they were the most limiting radionuclides, providing the greatest contribution to potential dose).

Table 6 Aquatic System Evaluation: Site-Specific Screening Results for the Poplar Springs Site using Mean Radionuclide Concentrations in Surface Water

Sampling Location	Average Concentrations Sum of Fractions < 1.0 (Pass/Fail?)	Water Sum of Fractions	Sediment Sum of Fractions	Total Sum of Fractions
Main Plant--On-site Locations:				
Two Falls Creek (TFCK 0.5)	(passed in general screen)			--
Broad Creek (BRCK)	passed	7.21E-01	8.98E-03	0.73
Northwest Tributary (NWTK 0.5)	(passed in general screen)			--
Downstream Locations:				
Muddy Branch (MB 0.6)	passed	9.38E-01	1.45E-02	0.975
Blue Falls Creek (BFCK 3.0)	failed	1.25	1.17E-02	1.26
Blue Falls Creek at Blue Falls Dam (BFCK 1.4)	passed	6.70E-01	8.85E-03	0.68
Other Streams that enter Darlington River:				
Taylor's Creek (TCK 1.0)	(passed in general screen)			--
Beaver Creek (BVCK 2.3)	(passed in general screen)			--

3. Site-Specific Screening using Site-Representative Parameter Values in Place of Default Values *(Phase 3 of the Graded Approach, Site-Specific Screening)*

Further efforts were directed at modifying some of the default parameters used in the site-specific screening portion of the graded approach, replacing them with more site-representative values.

A. Review of Data and Parameters for Blue Falls Creek (BFCK 3.0)

Because both maximum and average surface water concentrations collected at Blue Falls Creek exceeded the BCGs in general screening and site-specific screening, respectively, it was necessary to review the data used, limiting organism type responsible for the BCGs, limiting media, and area of evaluation. A summary of this review is provided in Table 7.

Table 7 Review of Radionuclide Concentration Data and Limiting Organism Type to Determine Path Forward in the Biota Dose Evaluation

Review the Following:	Comment
Sampling/Data Frequency -- adequate?	Surface water samples were collected and analyzed bimonthly (Jan, March, May, Jul, Sep, Nov): considered to be adequate. Possible Future Activities: * Consider possible need to increase sampling frequency (contact appropriate personnel) * Consider collection of co-located sediment samples (see below)
Radionuclides of concern?	Cs-137 and Sr-90 are the limiting radionuclides contributing the most to the total sum of fractions at this location. Water is the limiting medium; sediment contributes to dose but is not the limiting medium. Maximum and average concentrations detected in surface water for this location: Cs-137: Maximum: 67; Average: 37 pCi/L Sr-90: Maximum: 330; Average: 100 pCi/L
Are the limiting organism types used to derive BCGs reasonable?	Riparian animal -- yes, this receptor is feasible for the evaluation area. Known to be resident.
Consider re-defining or modifying the evaluation area?	Radionuclide data was already time-averaged to generate mean concentrations which are representative of the evaluation area. The location from which the radionuclide concentrations were detected is considered to be a representative indicator for site impacts on natural waterways. No additional modifications to the delineation of the evaluation area will be conducted.

B. Consider Replacing Default Lumped Parameter Values with Site-Representative Values

The major issues for this evaluation were Cs-137 and Sr-90 surface water concentrations. Therefore, the focus was on the radionuclide-specific default lumped parameters used to derive the BCGs for these two radionuclides.

The Riparian Animal Spreadsheet contained in the RAD-BCG Calculator (and contained in Module 1 Table 7.2 of the DOE technical standard) was reviewed to identify the default lumped parameter values (see Table 8 below for a summary). Available site data was reviewed for site-representative lumped parameter values for riparian animals (the limiting organism type for Cs-137 and Sr-90). After making some preliminary inquiries with site personnel, it was determined that there were no easily-accessible site-specific lumped parameter data for riparian animals. A more extensive search could have been performed (e.g., making contact with other DOE site representatives; conducting a literature search), but it was decided to move on to the site-specific analysis component of the graded approach, focusing on reviewing and potentially modifying additional default parameters and assumptions used in the analysis phase.

Table 8 Default Lumped Parameter Values Used to Derive Generic Water BCGs for Riparian Animals

Radionuclide	Lumped Parameter Bq/kg (animal-wet weight) per Bq/L(water)	Comment
Cs-137	50,000	A preliminary search at the Site indicated no known or easily accessible site-specific data for estimating site-specific lumped parameters for riparian animals.
Sr-90	6,000	A preliminary search at the Site indicated no known or easily accessible site-specific data for estimating site-specific lumped parameters for riparian animals.

4. Site-Specific Analysis Using Site-Representative Parameter Values and Assumptions in Place of Default Values *(Phase 3 of the Graded Approach, Site-Specific Analysis)*

A. Review Default Parameter Values and Consider Replacing with Site-Representative Values

A number of default parameters which are used in estimating a riparian animal's internal dose can be considered for modification in site-specific analysis. The default parameters for a riparian animal were reviewed by accessing the Riparian Animal Spreadsheet in the RAD-BCG Calculator (also contained in Module 1, Tables 7.5 and 7.6 of the DOE technical standard). These parameters are summarized in Table 9 below.

Table 9 Review of Default Parameter Values for Possible Modification Using Site-Representative Values

Parameter	Default Value	Site-Specific Values?
Appropriate Riparian Receptor?	Raccoon	Default organism is known to be resident at the site.
Fraction of intake retained Cs-137 Sr-90	 1 0.3	No known site specific evaluations to conclude otherwise. Default values were used to be conservative.
Biological Decay Constant Cs-137 Sr-90	 2.24E-02 6.11E-04	No known site specific evaluations to conclude otherwise. Default values were used to be conservative.
Correction Factor for Area or Time	1.0	No known site specific evaluations to conclude otherwise. The organism would be expected to be resident in the evaluation area 100% of the time.
Dose Limits for Riparian Animals	0.1 rad/d	Default dose limit used for riparian animals. Can not be changed without DOE-EH-41 approval.
Body Mass	8800 g	Default value. Default value was used to be conservative.
Other Kinetic/Allometric Relationship Parameters	Allometric equations and related input parameters representing mechanisms to internal dose to a riparian animal.	A cursory review of the default values for these parameters was made. It was decided to use the default values and equations rather than to obtain more site-representative values for use in the kinetic/allometric models employed in the analysis phase of the graded approach. However, the aquatic animal food source B_{iv} value used as the default food source to the riparian animal was reviewed (in the Aquatic Animal Spreadsheet) and subsequently modified.

Each of the contributing parameters could have been reviewed in detail, with the objective of identifying values more representative of site-specific receptors. It was determined through contact with aquatic biologists and radioecologists at the Poplar Springs Site that a reasonable amount of data relating to bioaccumulation factors (B_{iv}s) for fish was available at relevant Poplar Springs Site locations for the Blue Falls Creek evaluation area. Data exists for fish at or near Blue Falls Creek (BFCK 3.0) for Cs-137 and there is some data for Sr-90 in whole fish collected on-site in nearby waterways having similar water chemistry. It was determined that these fish were representative of the expected food sources to a riparian animal at the evaluation area, and that their B_{iv}s would provide more representative food source values to a site-specific riparian animal, in place of the default values used. With the assistance of the aquatic specialists, site-specific Cs-137 and Sr-90 concentrations measured in fish and in surface water were used to estimate B_{iv}s applicable to the Blue Falls Creek evaluation area. The data and resulting B_{iv}s are shown in Tables 10 and 11.

Table 10 Site-Specific Bioaccumulation Information for Cesium-137

Species	Water Concentration (Bq/L)	Tissue Concentration (Bq/kg)[1]	Bioaccumulation Factor (L/kg)[2]	Reference
Bluegill	1.52 Bq/L	BFCK 2.9 (N=7): 7900 ± 3400 Bq/kg dw BFCK 2.3 (N=5): 4600 ± 752 Bq/kg dw	1040 605	PSS/TM-11295 - Third Report of the PSS BMAP for Blue Falls Creek Watershed and the Darlington River (Tables 8.2-water and 8.11-fish)
Sunfish (includes bluegill and redbreast sunfish)	5.2 Bq/L	BFCK 3.5 (N=8): 21600 ± 2200 Bq/kg dw BFCK 2.9 (N=8) 29800 ± 9100 Bq/kg dw BFCK 2.3 (N=8): 13600 ± 8400 Bq/kg dw	830 1150 520	PSS/TM-10804 - Second Report of the PSS BMAP for Blue Falls Creek Watershed and the Darlington River (Table 8.23) Water Data Table 5.2.26 Environmental Surveillance of the PSS and Surrounding Environs (ES/ESH-1/V2)
Redbreast Sunfish	1.52 Bq/L	BFCK 2.9 (N=5): 7600 ± 1300 Bq/kg dw	1000	PSS/TM-11295- Third Report of the PSS BMAP for Blue Falls Creek Watershed and the Darlington River (Tables 8.2-water and 8.11-fish)

[1] Tissue concentrations were measured in fish fillets. It is assumed that the tissue concentrations in fillets are representative of whole body concentrations. This is appropriate, given that Cs-137 is known to concentrate in muscle tissues.

[2] It is assumed that fish are about 80% water; therefore, the dry weight of fish is multiplied by 0.2 to convert dry weight to wet weight.

Table 11 Site-Specific Bioaccumulation Information for Strontium-90

Species	Water Concentration (Bq/L)	Tissue Concentration (Bq/kg)	Bioaccumulation Factor (L/kg)	Reference
Bluegill	4.8 Bq/L	520 ± 140 Bq/kg ww (1987) (Whole body) N=5	110	PSS/TM-10804 - Second Report of the PSS BMAP for Blue Falls Creek Watershed and the Darlington River (Table 8.1) Blue Falls Creek Water Data Table 2.2.1 Environmental Surveillance of the PSS and Surrounding Environs (ES/ESH-4/V2).
Gizzard Shad	4.8 Bq//L	370 ± 360 Bq/kg ww (1987) (Whole body) N=5	80	PSS/TM-10804 - Second Report of the PSS BMAP for Blue Falls Creek Watershed and the Darlington River (Table 8.1) Blue Falls Creek Water Data Table 2.2.1 Environmental Surveillance of the PSS and Surrounding Environs (ES/ESH-4/V2)
Largemouth Bass	4.8 Bq/L	230 ± 120 Bq/kg ww (1987) (Whole body) N=5	50	PSS/TM-10804 - Second Report of the PSS BMAP for Blue Falls Creek Watershed and the Darlington River (Table 8.1) Blue Falls Creek Water Data Table 2.2.1 Environmental Surveillance of the PSS and Surrounding Environs (ES/ESH-4/V2)

B. Modification of Default B_{iv} Values for Organisms Consumed by the Limiting Organism

The Aquatic Animal Spreadsheet within the RAD-BCG Calculator was accessed and the default B_{iv} values for Cs-137 and Sr-90 were reviewed. Based on literature reviews, calculated values (Table 10 and Table 11), and consultations with the aquatic specialists**,** the following site-specific B_{iv}s for fish were selected:

Cs-137: 1150 (L/kg). Most conservative estimated bioaccumulation factor for fish collected at or near the sampling location (BFCK 2.9).

Sr-90: 110 (L/kg). Most conservative estimated bioaccumulation factor for fish collected on the Poplar Springs Site.

Enter Site-Representative Parameter Values into the RAD-BCG Calculator

First, the "allometric BCGs" button on the Riparian Animal Spreadsheet of the RAD-BCG Calculator was selected. This selection allowed the calculation of BCGs using the kinetic/allometric method. Then, the Aquatic Animal Spreadsheet of the RAD-BCG Calculator was accessed, and the default B_{iv} values for Cs-137 and Sr-90 were replaced by entering the site-specific B_{iv} values listed above. A "user supplied value" message appeared in the Aquatic Animal Spreadsheet to provide a reminder that default values had been modified. The BCGs for Cs-137 and Sr-90 were automatically updated within the RAD-BCG Calculator to reflect these site-specific input values. The site-specific BCGs for these two radionuclides were shown in the Riparian Animal Spreadsheet, and in the Aquatic System Data Entry/BCG Worksheet - where our mean measured radionuclide concentration data was previously entered. A new partial and total sum of fractions were automatically calculated by the RAD-BCG Calculator.

Compare Measured Radionuclide Concentrations in Environmental Media with BCGs

Due to the adjustment of the Cesium-137 B_{iv} to 1150 and the Sr-90 B_{iv} to 110, the total sum of fractions for Blue Falls Creek was less than 1.0, indicating that we passed the site-specific analysis.

It is also noteworthy that - had we used the site-specific food source B_{iv} values compared with maximum measured radionuclide concentration data rather than mean values, the total sum of fractions for our riparian animal would also have passed. This would be a useful approach if we were required by regulators or stakeholders to use only maximum measured radionuclide concentrations in our evaluation. This point highlights one example regarding the flexibility of the graded approach.

5. Documentation of Results

The results of the biota dose evaluation were summarized. A summary report containing computer screen printouts of the spreadsheets from the RAD-BCG Calculator were retained on file for future reference. The rationale for selecting site-representative B_{iv}s as a food source value to a riparian animal was documented. As required by EH, a summary of the evaluation was included in the Poplar Springs Site's Annual Site Environmental Report.

6. Lessons Learned

- Possible future activities could include: (1) assessing the need for additional sampling locations; (2) collecting co-located sediment and water samples for these and other locations; (3) collecting representative receptors and analyzing tissue data to permit a direct and more realistic dose evaluation.

A Graded Approach for Evaluating Radiation Doses to Aquatic and Terrestrial Biota

MODULE 2

DETAILED GUIDANCE

MODULE 2: DETAILED GUIDANCE

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1 The Graded Approach, Ecological Risk Assessment, and Guidance on Their Implementation in Evaluating Radiation Doses to Biota

The graded approach was made available to DOE field and program elements and to external users for a trial use period beginning in July 2000 as an interim version of this technical standard. The purpose of the trial period was to give users an opportunity to become familiar with and implement the graded approach at their sites, and to have an opportunity to provide suggestions and lessons learned to the BDAC regarding any refinements and associated guidance that needed to be incorporated into the graded approach prior to finalizing the technical standard. During this trial period the graded approach received strong interest and requests from many national and international organizations. Some of these organizations had an interest in applying the graded approach in support of additional types of environmental assessments.

1.1 Purpose of this Section

This section of the technical standard was added to be responsive to those individuals who, during the trial use period of the graded approach:

- requested guidance on the relationship between the graded approach and the Ecological Risk Assessment (ERA) framework typically used for the evaluation of chemical stressors to the environment;

- requested guidance on how to utilize the graded approach in support of other types of environmental assessments; and

- requested guidance on the technical elements and issues inherent in evaluating radiation as a stressor to the environment which are different from those encountered when evaluating chemical stressors to the environment. The individuals requesting this guidance indicated that they had experience in working with the ERA framework for chemicals but little experience in working with radiological risk assessment.

This section also provides a general orientation and “roadmap” to the remaining Sections of Module 2 containing detailed guidance on specific biota dose evaluation issues that may be encountered when implementing the graded approach. This guidance is also applicable to radiological ERAs.

1.2 Relationship of the Graded Approach and the Ecological Risk Assessment (ERA) Framework

Ecological Risk Assessment (ERA) is a process for logically organizing and evaluating information to determine the nature, likelihood, and magnitude of potential impacts on environmental receptors (Suter 1993). The ERA framework consists of three general steps: problem formulation, analysis of exposure and effects, and risk characterization. ERAs are

typically done in successively rigorous tiers, each of which includes the three general ERA steps (Suter et al. 2000). The first and simplest tier is a scoping assessment, which establishes the need for an ERA. The second tier consists of a screening ERA, which is relatively simple and conservative in its application and assumptions. The third tier is a definitive ERA, which provides a relatively detailed and realistic assessment of the nature and magnitude of risks. The ERA framework is general in nature and has been widely applied in the evaluation of chemical stressors to the environment. The ERA framework can be applied to the evaluation of radiation as a stressor to the environment, but not without some modifications and provision of additional guidance. Some issues are the same as for chemicals, but some are unique to radionuclides.

The graded approach for evaluating radiation doses to aquatic and terrestrial biota is consistent with the standard ecological risk assessment (ERA) paradigm (EPA 1998). As in the standard ERA paradigm, the graded approach provides several tiers that move from a simple and relatively conservative screening evaluation to a more detailed and realistic assessment. Each step in the graded approach addresses, either explicitly or implicitly, the principal ERA components. That is, the graded approach is a framework for organizing the successively rigorous ERA tiers, but with a particular emphasis on radioecological issues.

1.3 Principal and Alternative Uses of the Graded Approach

The principal driver and basis of need for developing the graded approach was to provide DOE field and program elements with methods for demonstrating compliance with DOE biota dose limits and recommendations for radiological protection of the environment. Thus, many of the decisions that are traditionally made when conducting a case-specific ERA (e.g., choice of indicator receptors; defining receptor exposure profiles; selection of effects endpoints) were made at a programmatic level and incorporated into the screening phase of the graded approach *a priori.* For example, the thresholds for adverse effects were set at the recommended limits for protection of natural populations of biota. Those are the appropriate effects levels for demonstrating compliance with DOE requirements and recommendations for the protection of the environment from ionizing radiation (Module 1, Section 1.2).

The graded approach and BCGs can be used in support of other types of environmental assessments, provided that the user ensures that issues specific to the alternative application are appropriately addressed. Examples of other types of environmental assessments that the graded approach could potentially support include: ERAs at hazardous waste sites (i.e., Superfund sites), assessments for waste disposal and other facilities, and assessments at various stages of the Natural Resource Damage Assessment (NRDA) process. These typically include retrospective assessments of previously contaminated areas. These could also include prospective assessments of migrating contaminants (e.g., groundwater plumes) and planned releases (e.g., NEPA alternatives analysis).

If the graded approach is used for these or other purposes, then the programmatic objectives and the methods and model assumptions should be re-evaluated and discussed with the relevant decision makers and stakeholders, preferably via the Data Quality Objectives (DQO) process (Bilyard et al. 1997) or comparable processes to ensure that the results obtained through application of the graded approach will support the management goals and objectives of the environmental assessment. Module 1, Section 3 provides additional information on principal and potential applications of the graded approach along with specific application considerations.

1.4 Technical Issues to be Considered when Evaluating Radiation as a Stressor to the Environment

As mentioned earlier, the ERA framework is general in nature and can be applied to the assessment of radiation doses to biota. However, there are some noteworthy technical issues concerning the evaluation of radiation as a stressor that require further consideration and elaboration. To our knowledge, standardized guidance on how to address these issues is not available elsewhere.

In response to requests for guidance on this topic, Section 1.4 serves as a basic “primer” on technical issues that should be considered when evaluating radiation as a stressor to the environment, and draws on the experiences gained by BDAC members in developing the graded approach and conducting radiological ERAs. It focuses on key biota dose assessment issues identified in the graded approach. To facilitate communication of guidance on this topic, this section was intentionally written and organized with an orientation to those familiar with the ERA framework for chemicals. The issues, and an explanation of how they are addressed in the graded approach, are described below within the context of the ERA framework.

1.4.1 Problem Formulation

The first step of an ERA involves a formulation of the problem, in which the purpose of the assessment is clearly defined, the problem is clearly stated, and a plan for analyzing and characterizing risks is developed (Figure 1.1). This entails identifying the spatial and temporal bounds of the assessment, identifying the potential stressors and receptors, selecting assessment endpoints, developing a site conceptual model, selecting appropriate measures of exposure and effects, and developing an analysis plan (EPA 1998; Suter et al. 2000).

1.4.1.1 Scope of the Assessment

One of the first steps in problem formulation is to define the spatial and temporal scope of the assessment. The proposed spatial bounds of the assessment will determine which of the potential assessment endpoints are of an appropriate scale for the site. Conversely, identification of specific endpoints of concern can be used to set the spatial scale of the

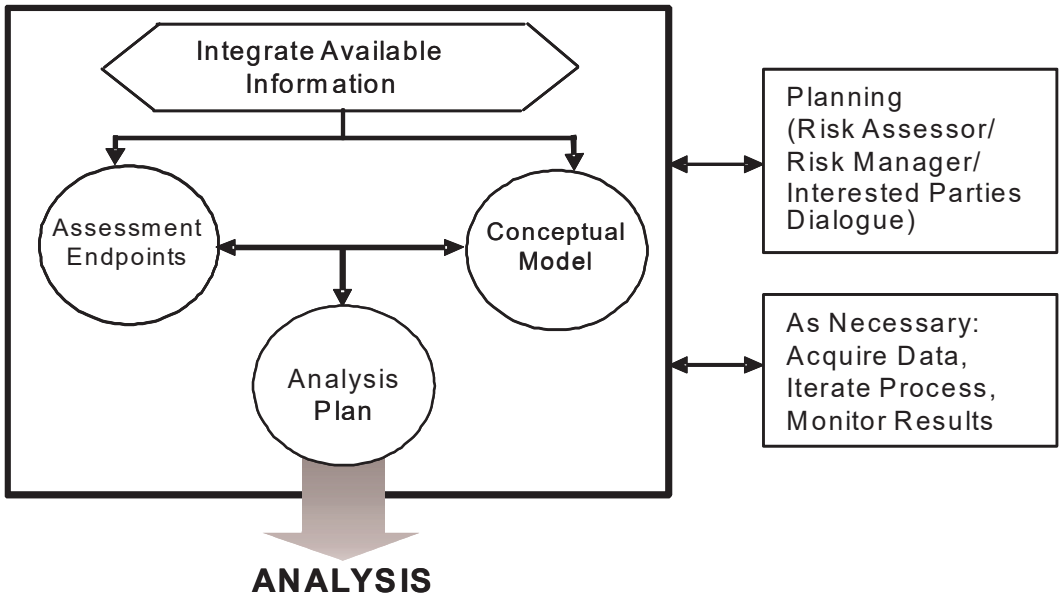

Figure 1.1 Problem Formulation, Phase 1 of Dose Assessment (from EPA 1998)

assessment. Establishing the physical scope of an assessment is addressed in more detail in Module 2, Section 4 and elsewhere (Suter et al. 2000).

The temporal scope of the assessment is determined by the types of exposures and effects that are anticipated. With the exception of rare accidents (e.g., Chernobyl), radiological ERAs are concerned with long-term, low-level exposures that are appropriately evaluated as chronic exposures. Thus, the temporal scope is generally not less than a week and more frequently on the order of months to a year. Aggregation of data across time and space is addressed in Module 2, Section 3.

1.4.1.2 Stressor Characteristics

Unlike standard ERAs, radiological ERAs are by definition focused on one stressor, ionizing radiation resulting from the decay of unstable isotopes that have been released to the environment. Many of the stressor characteristics that must be considered when developing a conceptual model and selecting endpoints are the same for radionuclides as for non-radioactive chemicals, because fate and transport of radionuclides in the environment is generally determined by elemental properties, rather than isotopic properties. For example, biological uptake and partitioning among ambient media will be similar for ^{235}U and stable uranium.

However, there are also several radiation-specific characteristics that must be considered when developing the conceptual model and analysis plan. These include: (1) variation in penetrating power and damage potential of the radiations of primary concern in radioactive decay (i.e., alpha particles, electrons, and photons); (2) additivity of exposure when Radiation Weighting

Factors (RWFs) are used; (3) external exposure; and (4) exposure from radioactive decay products (progeny), the environmental fate of which is often different from the parent radionuclide. These issues are discussed below.

1.4.1.3 Assessment Endpoints

Assessment endpoints are an explicit expression of the environmental value that is to be protected, operationally defined by an ecological entity and its attributes (EPA 1998). For example, the fish community is a possible assessment endpoint entity and reduced reproduction is a possible assessment endpoint attribute. Of the recommended criteria for selecting and defining assessment endpoints (EPA 1998; Suter et al. 2000), relevance to management goals and susceptibility require elaboration for use in radiological ERAs.

Ensuring the relevance of the assessment endpoints to management goals includes selecting ecological entities and attributes that are valued by society. Most reviews and guidance identify populations as the lowest level of organization appropriate for assessing the effects of radiation on ecological receptors (NCRP 1991; IAEA 1992; UNSCEAR 1996). Therefore, the graded approach focuses on Population-Relevant Attributes (PRAs), such as reproduction (Module 1, Section 1.2). Although the effects data for PRAs are based on studies of individual organisms, it is the viability of the population as a whole, rather than the viability of any given individual in the population, that is of interest. Management goals for alternative applications of the graded approach may include a need to protect individual organisms (e.g., protection of threatened and endangered species).

Several key issues that should be considered when determining the appropriate criteria and exposure-response assumptions for protecting individual organisms in a population are presented in Module 2, Section 8. However, final selection criteria and exposure-response assumptions should be made in consultation with the appropriate decision makers if the graded approach is to be used for this alternative purpose.

Susceptibility to the stressor is a function of exposure and sensitivity. Exposure is typically defined as co-occurrence or contact of the receptor with the stressor, i.e., ionizing radiation. Sensitivity refers to how readily the endpoint entity responds to the stressor. Sensitivity to radiation (radiosensitivity) of major taxonomic groups and life stages is discussed below. One should also consider life history and habitat when selecting susceptible receptors, with highly exposed and sensitive life-stages taking precedence. In general, recommended endpoint entities include aquatic and terrestrial vertebrates and higher plants (e.g., Pinus species and other woody plants). In contrast, invertebrates and primitive plants (e.g., mosses and lichens) are generally not appropriate assessment endpoint entities, because they are comparatively insensitive to the direct effects of irradiation.

Three generic assessment endpoints were selected for use in the graded approach, based on the issues mentioned above and the availability of relevant exposure and effects data. The selected endpoints are:

- Observable reductions of survival or reproductive capability in natural aquatic animal populations.

- Observable reductions of survival or reproductive capability in natural terrestrial animal populations.

- Observable reductions of survival or productivity of terrestrial plant populations.

1.4.1.4 Conceptual Model

Developing a conceptual model of the site entails describing and visually depicting the relationships between the stressors and the endpoint entities (ASTM 1995, EPA 1998, and Suter 1996). The conceptual model includes the known and expected relationships among the stressors, pathways, and assessment endpoints which are considered in the assessment and a rationale for their inclusion. Relationships that cannot or will not be addressed should be identified and a rationale for their exclusion should be provided. The conceptual models for the graded approach are illustrated in Figures 2.1 through 2.4 in Module 2, Section 2.1.2. These generic conceptual models depict the typical radiation exposure pathways for biota that may be evaluated using the graded approach. Some or all components of these models may be used for a specific application of the graded approach. Additional conceptual models could be developed for alternative applications of the graded approach, possibly as part of the DQO process.

1.4.1.5 Analysis Plan

The final stage of problem formulation is development of an analysis plan. This includes delineation of the assessment design, data needs, measures, and methods for conducting the analysis step of the assessment (EPA 1998). This encompasses most of the information contained in the graded approach. For example, Module 1 of this technical standard provides a description of the assessment design, general guidance on data needs, and detailed directions for conducting an evaluation using the graded approach. Modules 2 and 3 provide additional details and guidance on these issues. **That is, the graded approach is a detailed analysis plan for determining whether or not a DOE site is in compliance with DOE requirements and recommendations for the protection of aquatic and terrestrial biota from ionizing radiation.**

1.4.1.6 Measures

Of the components of the analysis plan, measures warrant further elaboration with respect to radiological assessments. Measures, formerly referred to simply as measurement endpoints, consist of measures of effects, measures of exposure, and measures of ecosystem and receptor characteristics (EPA 1998). Measures of effects include: survival of plants and animals and changes in reproduction (i.e., the processes from gametogenesis to embryonic development) of plants and animals. Measures of exposure include: (1) radiation dose rates to aquatic animals and terrestrial plants and animals, and (2) radionuclide concentrations in ambient media or biota at levels commensurate with selected radiation dose rates to aquatic animals and terrestrial plants and animals. Measures of ecosystem characteristics include the abundance and distribution of suitable habitat. Measures of receptor characteristics include feeding and migratory behaviors and natural reproduction, growth, and mortality rates.

Measures of exposure and effects were selected for the purpose of demonstrating protection through compliance with DOE requirements and the recommendations contained in the graded approach (Module 1, Section 1.1). Key selected measures of effects are the dose rates at which measurable reductions in reproduction of plants and animals are not expected (i.e., the expected safe levels of exposure). Key selected measures of exposure are the concentrations of radionuclides in ambient media that are expected to result in those dose rates. More specifically, the critical measures of exposure/effects selected for use in the graded approach are 1 rad/d for aquatic animals and terrestrial plants and 0.1 rad/d for riparian and terrestrial animals (Module 1, Section 1.1). The default assumptions related to the measures used in the graded approach can be modified for alternative applications.

1.5 Analysis

The second step in the risk assessment process is analysis, which consists of analyses of exposure and effects (EPA 1998). These analyses are typically done concurrently and itteratively (Figure 1.2)

1.5.1 Exposure Analysis

Exposure is the contact or co-occurrence of a contaminant with a receptor. The exposure analysis estimates the magnitude of exposure in terms of intensity, space, and time in units that can be combined with the effects analysis (EPA 1998). It entails describing the sources and distribution of the stressors through space and time, evaluating transport and exposure pathways, and describing the contact or co-occurrence with the receptor. The degree of detail and conservatism in the analysis of exposure depends on the tier of the assessment. The bulk of the guidance provided in the graded approach addresses exposure analysis.

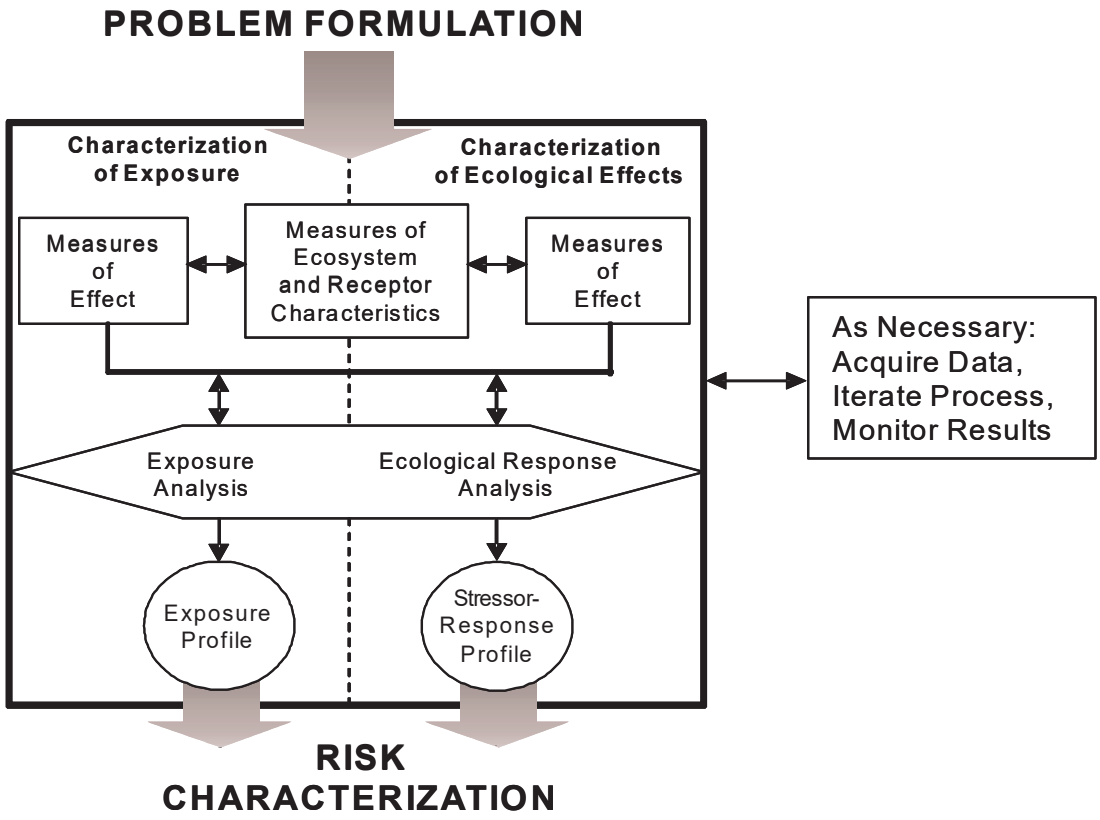

Figure 1.2 Analysis, Phase 2 of Dose Assessment (from EPA 1998)

For example, describing the sources and distribution of the stressors through space and time is addressed in the guidance on spatial and temporal averaging (Module 2, Section 3); evaluating transport and exposure pathways is addressed in the guidance on soil sampling relative to plant rooting depths (Module 2, Section 5); and describing the contact or co-occurrence with the receptor is addressed in the guidance on sources, receptors, and routes of exposure (Module 2, Section 2).

The radiation-specific characteristics mentioned above are addressed in the graded approach as follows:

- Variation in penetrating power refers to the fact that electrons and photons can penetrate tissues and at least some amount of ambient media, whereas alphas particles cannot. A corollary to penetrating ability is the potential of each type of radiation to cause biological damage. Alpha particles are non-penetrating because they are relatively large, which also means they have a high linear energy transfer. Electrons and photons have a low linear energy transfer. This is the basis for the greater

biological effectiveness of alpha particles relative to that of electrons and photons (Module 2, Section 7).

- Additivity of exposure refers to the fact that the absorbed dose (or dose rate) of ionizing radiation from all media, radionuclides, and radiations can and should be added together, provided one accounts for relative biological effectiveness (i.e., appropriate radiation weighting factors are used). This stems from the fact that the expected safe levels of exposure are based on the total absorbed dose of ionizing radiation from low linear energy transfer radiations (Module 2, Section 7).

- External exposure refers to the ability of a radionuclide to affect an ecological receptor without the radionuclide being taken into the receptor. This highlights the fact that the stressor of concern is ionizing radiation, rather than the individual radionuclides that give off that radiation. External exposure pathways are conceptualized in Module 2, Section 2.1.2 and quantified in Module 3, Section 2.

- Exposure from radioactive decay products refers to the fact that radioactive decay of one isotope may result in one or more new isotopes which are also radioactive. These decay products (progeny) may be short-lived, existing for only seconds or hours before decaying again to produce isotope-specific radiations and additional decay products (radioactive or stable). Relatively long-lived isotopes may be detected in the environment, whereas short-lived progeny might not be detected. Consequently, the absorbed dose from short-lived radioactive progeny is included in the exposure calculations for the long-lived parent isotope (Module 3, Section 2).

1.5.2 Effects Analysis

The effects analysis estimates the nature and magnitude of effects with respect to the magnitude and duration of exposure (i.e., dose or dose rate) (EPA 1998; Suter et al. 2000). It entails evaluating and summarizing the effects data in a way that facilitates relating effects to the exposure estimates. Unlike the analysis of exposure, the analysis of effects is not discussed extensively in the graded approach. This is because achieving the primary objective of the graded approach, i.e., compliance with the DOE requirements and recommendations, obviates the need for the user to select and justify the effects data and assumptions. That is, key decisions about the effects evaluated in the graded approach were made at the programmatic level, rather than at the site-specific level.

Three aspects of the analysis of radiation effects on biota are explicitly discussed in the graded approach: expected safe levels of exposure, radiation weighting factors (RWFs), and radiosensitivity of various receptors and attributes. The expected safe levels of exposure are the bases for the DOE requirements and recommendations for protection of aquatic and terrestrial biota from ionizing radiation (Module 1, Section 1.2.2). They are based on reviews of the available data for acute and chronic effects of radiation on population relevant attributes of

aquatic and terrestrial biota. The radiosensitivity of various receptors and attributes were used to select the default assessment endpoints used in the graded approach. Radiosensitivity generally increases with increasing organism complexity. However, radiosensitivity can vary by one or more orders of magnitude among phylogenetically similar species (UNSCEAR 1996). Life stage also affects radiosensitivity, with reproductive processes and the early stages of development generally being the most radiosensitive due to the ongoing activities of cell division and differentiation.

Radiation weighting factors (RWFs) account for the fact that all types of ionizing radiation are not the same with respect to their biological effectiveness (Module 2, Section 7). They are based on observed relative biological effectiveness (RBE) factors (i.e., the inverse ratio of doses causing the same level of effect) and are used to normalize the different types of ionizing radiation (i.e. alpha, electrons, and photons). The use of RWFs allows one to sum the absorbed dose rates calculated in the exposure analysis for each type of ionizing radiation to obtain a biologically significant total dose rate.

The default effects thresholds and radiation weighting factors used in the graded approach (and the associated RAD-BCG Calculator) can be changed to support alternative uses of the graded approach. For example, the expected safe level of exposure for populations of terrestrial animals might be divided by a safety factor (e.g., 10) when evaluating the potential for adverse effects on individuals of a threatened or endangered species (Module 2, Section 8). Conversely, the default radiation weighting factor of 20 for alpha particles might be reduced to 5 to be more consistent with the relative biological effects data for deterministic effects (Module 2, Section 7).

1.6 Risk Characterization

Risk characterization is the final assessment step (Figure 1.3). It entails combining the results of the exposure and the effects analysis to provide an estimate of the probability and magnitude of adverse effects (risks) at the site in question. The risks should be described in the context of the significance of the effects and available data; the uncertainties, assumptions, and qualifiers should be identified and summarized (EPA 1998). Risk characterization is often classified as either part of a screening assessment or a definitive assessment. Screening assessments are typically based on relatively simplistic exposure and effects assumptions (e.g., maximum exposure and a single threshold for effects). Definitive assessments typically include detailed exposure models and, to the extent possible, site-specific biological effects data (e.g., toxicity tests with ambient media and demographic surveys of the receptors).

Risk characterization in the graded approach is mostly of the screening-type, where exposure estimates of varying conservatism and complexity are compared with a threshold for effects for each type of receptor (see Module 1, Sections 6, 7.1, and 7.2). Definitive risk characterization in the graded approach is generally limited to the site-specific biota dose assessments, for which general guidance is provided (Module 1, Section 7.3).

The particular screening risk characterization method used in the graded approach is the sum of fractions rule. This is conceptually analogous to the standard risk characterization technique for calculating hazard quotients (HQs) and a hazard index (i.e., sum of multiple HQs). It entails dividing the concentration of a radionuclide measured in the ambient media by the Biota Concentration Guide (BCG) for that radionuclide and the selected assessment endpoint (i.e.,

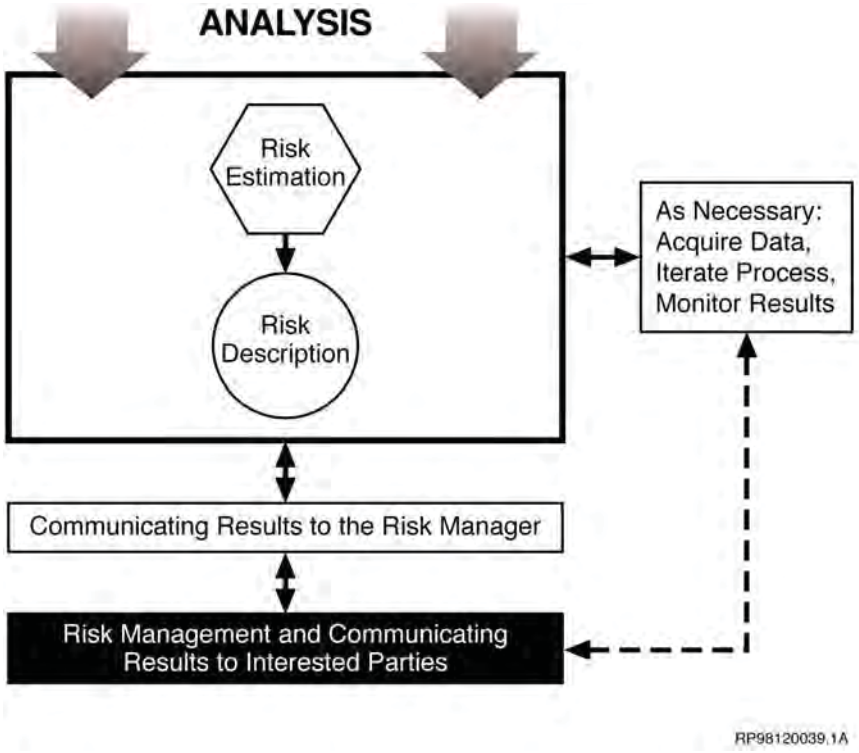

Figure 1.3 Risk Characterization, Phase 3 of Dose Assessment (from EPA 1998)

calculating the BCG fraction). The Biota Concentration Guides (BCGs) are screening values that incorporate default exposure assumptions and the effects threshold for the receptor to be evaluated. The BCG fractions are summed for each assessment endpoint (receptor), because the DOE requirements and recommendations are based on the total weighted absorbed radiation dose rate from all radionuclides and pathways.

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2 Guidance on Sources, Receptors, and Routes of Exposure

This section provides guidance and factors to consider when defining sources, receptors, and routes of exposure for application in the DOE graded approach.

2.1 General Considerations for Identifying Sources, Receptors, and Exposure Pathways

Exposure pathways are functions of the characteristics of the media in which the sources occur, and how both the released radionuclides and the receptors interact with those media. Many potential pathways exist at any given site that supports plants and animals and at which released radionuclides are found. The information presented below in Table 2.1 should generally be considered during the data assembly phase of the graded approach, and should specifically be considered in more detail during the analysis phase of the graded approach.

Table 2.1 General Considerations for Defining Sources, Receptors, and Routes of Exposure

Biogeochemical Properties of Radionuclides	• The biogeochemical properties of the released radionuclides are important because they determine the forms of the material in environmental media (i.e., solid, liquid, gaseous, dissolved), hence, its mobility and bioavailability. For example, radionuclides that are easily dissolved in water are more likely to migrate and disperse throughout the environment. These properties are also important because they determine whether a material bioaccumulates and the degree to which bioaccumulation occurs.
Nature of the Sources of Contamination	• The sources of contamination may exist in place (e.g., in soil or sediment) with or without further inputs of released radionuclides. These sources may be on the surface, buried, or moving through the medium by one or more processes. Alternatively, the sources of contamination may be point or non-point discharges of radioactive materials into the air, water, or soil. • Where the sources of contamination are located in the environment, if and how they are discharged into the environment and their subsequent mobility through environmental media are important determinants of their distribution throughout the environment in space and time.

Table 2.1 (Continued) General Considerations for Defining Sources, Receptors, and Routes of Exposure

Environmental Media	• The environmental media in which the released radionuclides are found (i.e., water, soil, or sediment) set the boundaries for the mobility of the released radionuclides through and among media. For example, released radionuclides in water may be dissolved or suspended as particulates, and their concentrations may be diluted through natural processes (e.g., currents, waves). • Suspended particulates may be deposited in the sediments, re-suspended, or even eroded by the wind if the water evaporates. • Materials in the air may be dispersed over large distances, subsequently deposited in the water or on the soil. • Released radionuclides in the soil may exist as immobile particulates or mobile dissolved forms, and may move from one form to another in space and through time, depending on the pH and redox potential of the soil. Other factors such as carbonates, organic matter, and clay content and type can also be important.
Ecology of the Receptors	• The interactions of each receptor within its environment define the routes of its exposure. A species that burrows in the soil and preys on soil organisms will have a different exposure profile than herbivores that live on the surface. • The ecology determines how the receptor is exposed in time and space. Rates of exposure and total doses will vary among similar types of organisms, based on whether an organism is immobile, mobile and local, or mobile and migratory. • Depending upon the phase of the graded approach you are working in (e.g., if you are moving from general screening to a site-specific analysis) it may be useful to develop a site conceptual model of the type used in ecological risk assessments. Helpful references include ASTM (1995), EPA (1998), and Suter (1996). An ecological scoping checklist for assembling a conceptual model is provided in Ryti et al. (1999). An automated conceptual model builder is also available (DOE 1997).

2.1.1 Sources

Ionizing radiation should be present in the environment at concentrations that are measurable using routine survey methods. Nuclide-specific information is preferred. Measurements of gross alpha radiation and/or gross beta radiation may be useful in defining the areas of contamination and the identification of localized areas of high concentration.

The sources of ionizing radiation should also be persistent. If long-lived radionuclides are present in measurable concentrations and receptors are exposed to them, an evaluation will be needed. Short-lived radionuclides (e.g., with a half-life less than 3 months), if continuously or regularly released into the environment, could be present on a regular basis. As a guide,

radionuclides with half-lives less than 6 months that are discharged into the environment in measurable quantities at least twice in a given 12-month period may warrant an evaluation.

2.1.2 Receptors and Routes of Exposure Considered in the Graded Approach

Four organism types and their corresponding dose limits were used in deriving the screening and analysis methods contained in this technical standard. The principal exposure pathways considered for aquatic animal (1 rad/d), riparian animal (0.1 rad/d), terrestrial plant (1 rad/d), and terrestrial animal (0.1 rad/d) organism types are shown in Figures 2.1, 2.2, 2.3, and 2.4, respectively. Dose evaluations for site-specific receptors (as defined by the user in the analysis phase of the graded approach) should reflect consideration of all relevant exposure pathways depicted in these figures, and as described in Module 3.

2.1.3 Examples of Receptors That Could Serve as Good Indicators of Radiological Impact

Selected examples of organisms that could be used in the analysis phase of the graded approach as indicators of radiological impact are provided in Table 2.2. Examples were provided by BDAC members from several DOE sites. The examples are based on the BDAC members' expertise in radioecology and experience in conducting radiological ERAs at their sites. The rationale used by BDAC members in identifying example representative organisms includes, but is not limited to, the following:

- The home range of the organism should be considered, with preference given to organisms with small home ranges.

- The organism should be susceptible (i.e., exposed *and* sensitive) to ionizing radiation. Organisms that are good accumulators of radionucides but are not very radiosensitive are generally not the most appropriate organisms. For example, mammals and other vertebrates are generally more radiosensitive then are invertebrates. Higher plants are more radiosensitive than mosses and lichens.

- The organism should represent the major exposure pathways for aquatic and terrestrial biota.

- The organism should be indigenous to the evaluation area and utilize the principal habitat present in the evaluation area.

- The organism is one that the general public is familiar with and can relate to.

- The organism has a reasonable amount of data available about it in the published literature or from site-specific studies (e.g., in terms of characterizing its radiosensitivity; environmental transfer factor parameters needed for application in the biota dose evaluation).

- The organism should be appropriate to the ecosystem type being evaluated (e.g., regional differences in ecosystems).

- The organism is one of the keystone or focal species for the ecosystem type being evaluated. It should be important to the function and structure of the ecosystem.

These examples are provided for illustrative purposes and are not all-inclusive. It is the user's responsibility to select site-specific organisms appropriate for the area being evaluated and to document the rationale for their selection. See also Section 6.2.2 through 6.2.4 for guidance on selection and sampling of receptors.

Table 2.2 Examples of Representative Organisms That Could Serve as Indicators of Radiological Impact

AQUATIC ANIMALS	AQUATIC PLANTS	RIPARIAN ANIMALS	TERRESTRIAL ANIMALS	TERRESTRIAL PLANTS
Savannah River Site and the Southeast				
largemouth bass	pondweed	beaver	hipsid cotton rat	loblolly pine
channel catfish	cat-tail	raccoon	cotton mouse	longleaf pine
redbreast sunfish		alligator	fox	bald cypress (also a riparian plant)
				swamp tupelo (also a riparian plant)

Table 2.2 (Continued) Examples of Representative Organisms That Could Serve as Indicators of Radiological Impact

AQUATIC ANIMALS	AQUATIC PLANTS	RIPARIAN ANIMALS	TERRESTRIAL ANIMALS	TERRESTRIAL PLANTS
Oak Ridge Site				
catfish		mink	whitefooted mouse	small vascular plants such as grasses and shrubs
carp		muskrat	deer mouse	pine trees
suckers		raccoon	cottontail rabbit	
			red and gray foxes	
Idaho National Engineering and Environmental Laboratory				
			sage grouse	sage brush
		great basin spadefoot toad	coyote	
Pacific Northwest National Laboratory				
bass		raccoon	deer mouse	gray rabbit brush
carp		beaver	great basin pocket mouse	reed canary grass
sculpin			mule deer	mulberry tree
salmonids			coyote	
			great blue heron	
			bat	
			king bird	

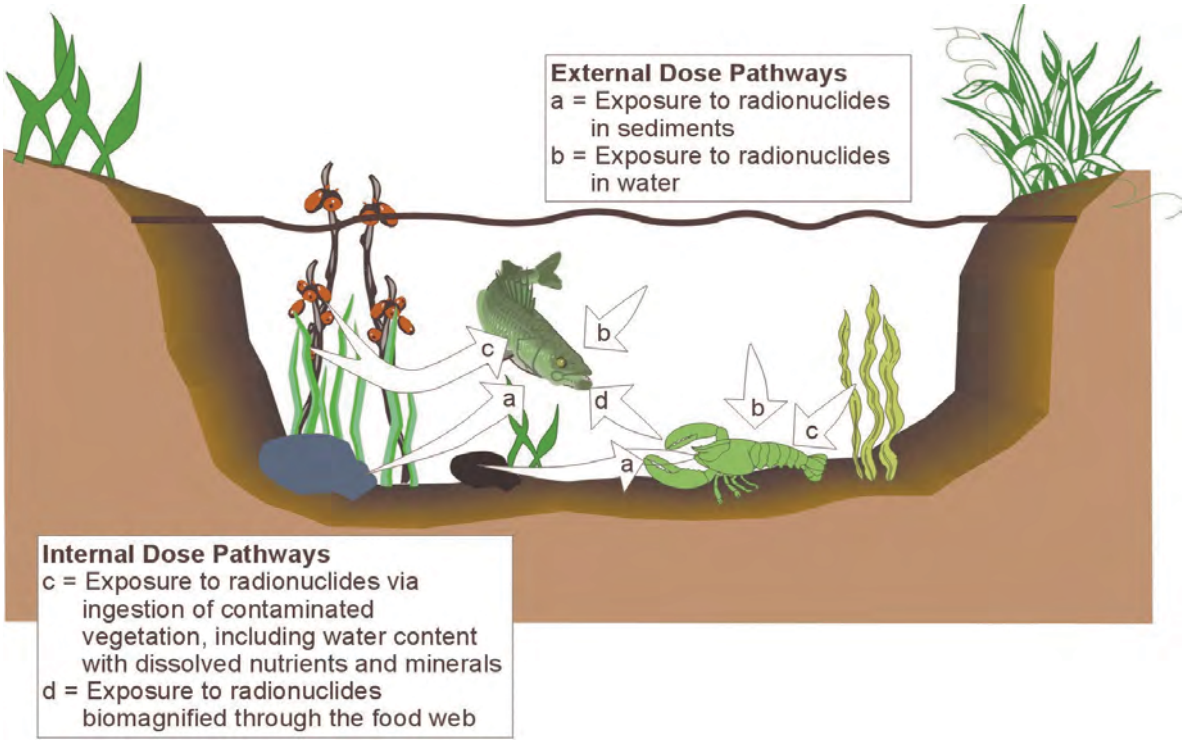

Figure 2.1 Exposure Pathways for Aquatic Animals

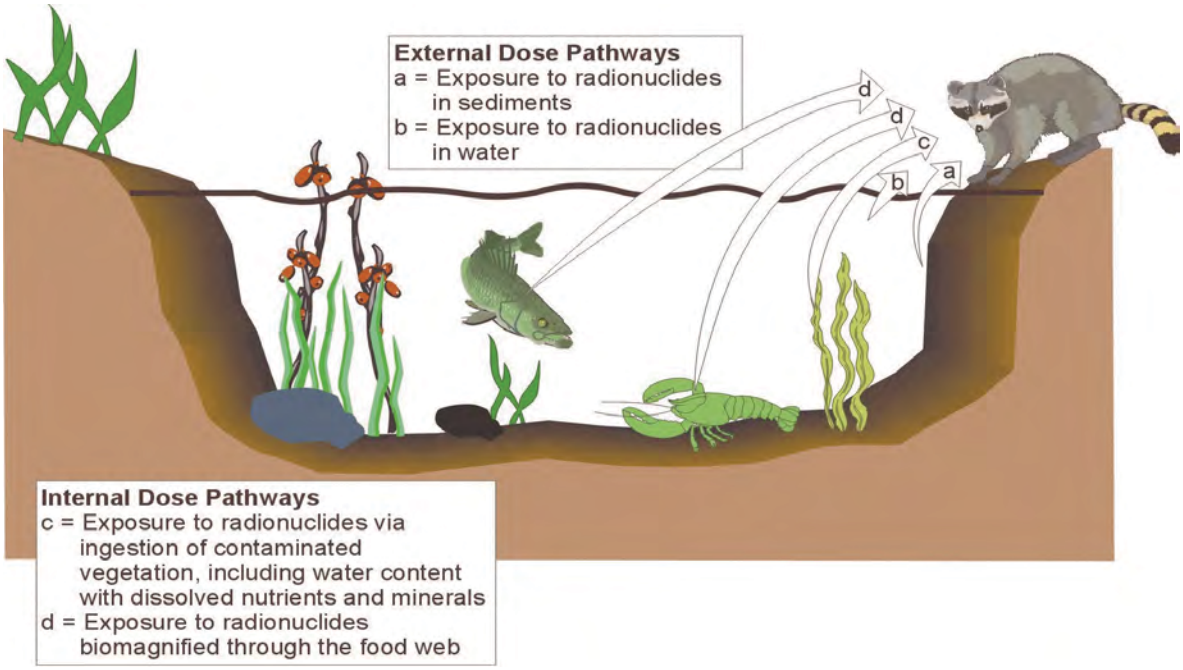

Figure 2.2 Exposure Pathways for Riparian Animals

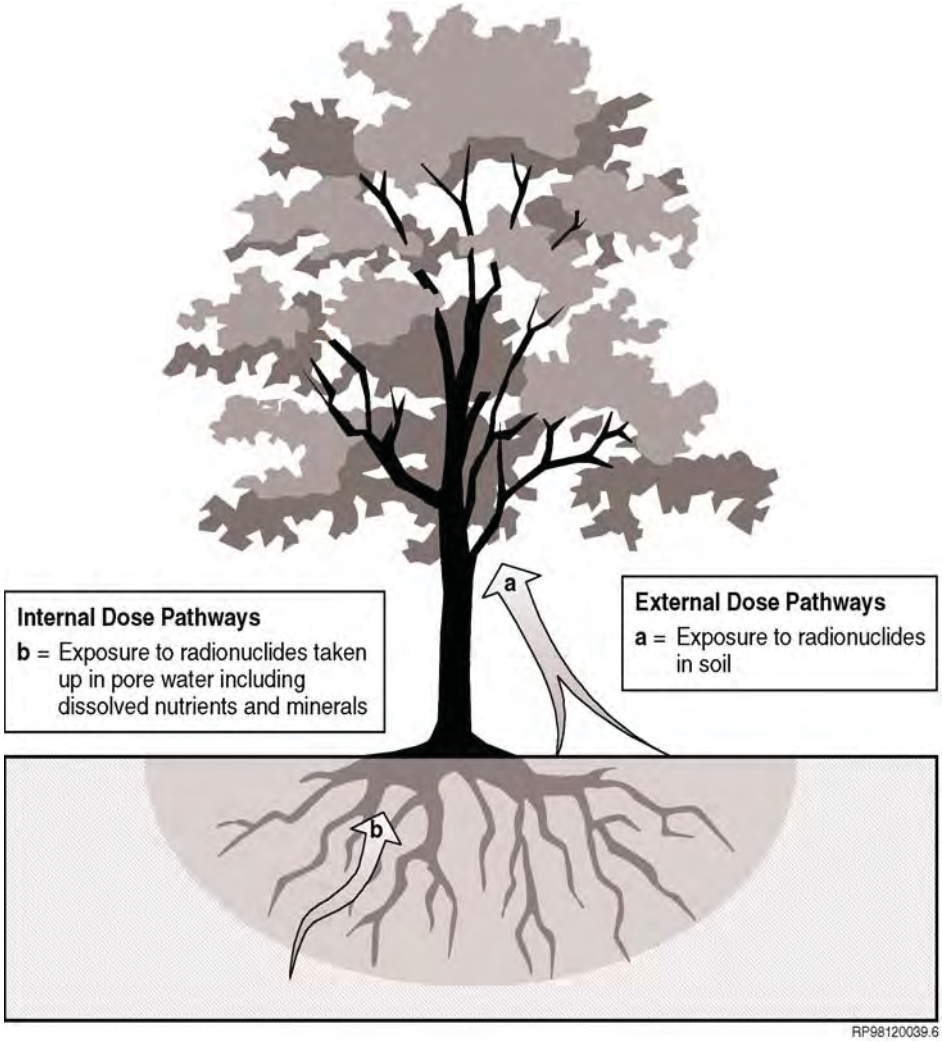

Figure 2.3 Exposure Pathways for Terrestrial Plants

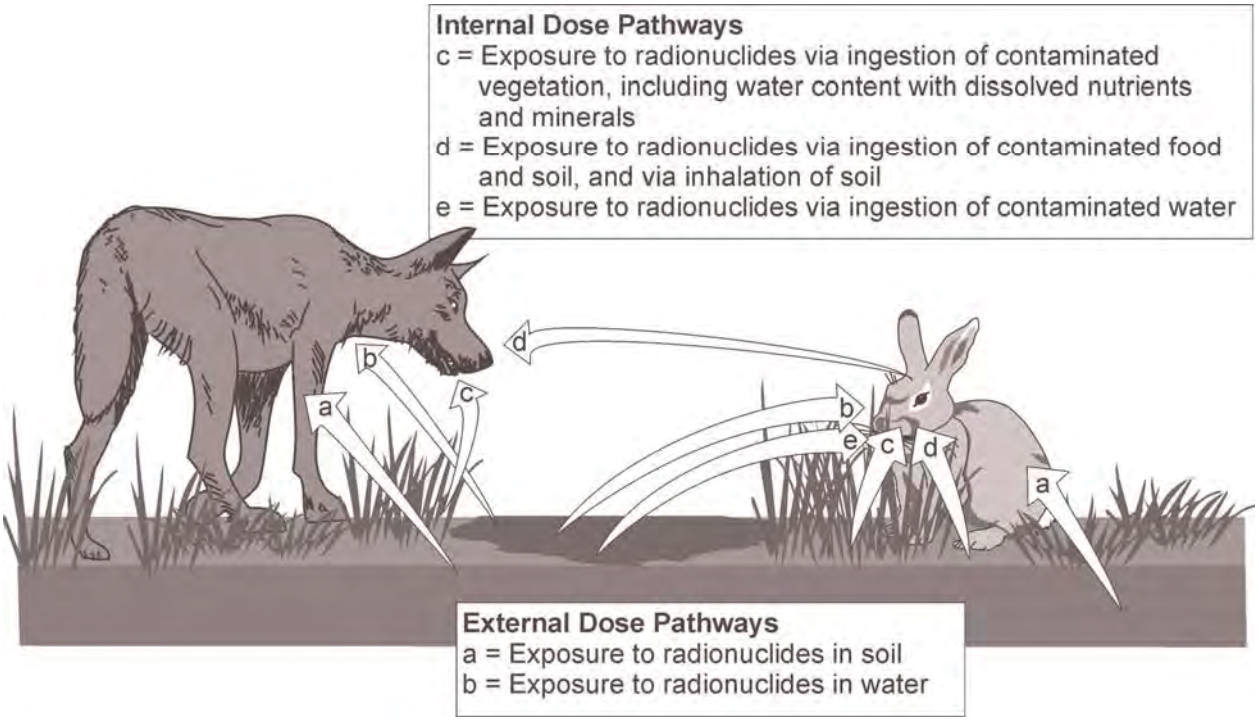

Figure 2.4 Exposure Pathways for Terrestrial Animals

2.2 Rationale for the Active Air Pathway as a Minor Source of Exposure

The active air (i.e., continuous air emission) release pathway was not included in the derivation of the BCGs because biota inhalation and immersion in air were estimated to be relatively insignificant contributors to biota dose. In response to comments received on the interim version of this technical standard regarding the statement that airborne emissions of radionuclides represent a minor source of exposure for animals and plants, the active air release pathway was further evaluated by the BDAC.

2.2.1 Behavior of Radionuclides Discharged to the Atmosphere

Unlike releases of radionuclides to water or soil, atmospheric discharges almost always rapidly disperse. For example, along the centerline of a Gaussian plume resulting from a ground-level point source, and assuming neutral stability (Pasquill-Gifford Stability Category D) to represent an average plume, the concentration at a distance of 100 m is reduced by a factor of about 500 compared with the concentration close to the source (DOE 1984). Reductions in concentrations are much greater at locations away from the plume centerline or at greater distances from a source. The rapid dispersal of airborne radionuclides is an important consideration in evaluating doses to biota.

2.2.2 Exposure Pathways Resulting from Atmospheric Releases

Within the context of the graded approach methodology, in considering radiation doses to biota resulting from atmospheric releases, there are three exposure pathways of concern. These are: (1) external exposure of terrestrial plants and animals to airborne radionuclides (cloudshine); (2) inhalation of airborne radionuclides by terrestrial animals; and (3) absorption of airborne radionuclides by terrestrial plants. All other potential exposure pathways are a consequence of deposition of airborne radionuclides onto the land surface or surface waters (including, for example, inhalation of resuspended radionuclides by terrestrial animals). *It is important to note that these other pathways are already taken into account in the graded approach methodology.*

2.2.3 Compliance with Human Radiation Dose Limits at DOE Sites Relative to Biota Dose Limits: A Perspective

First, airborne emissions of radionuclides at DOE sites are limited to very small quantities to protect human health. Current DOE (and EPA and NRC) policies restrict radioactive air emissions so that radiation exposures of the general public will be less than 10 mrem/y (0.1 mSv/y). Non-radiation workers at a DOE site are protected to 100 mrem/y (1mSv/y) from all sources (DOE 1984). These policies are significant in the original decision to not include the active air pathway in the graded approach methodology. Second, unlike exposures to radionuclides in soil, water, and sediment, the exposure pathways from active air releases are the same for biota as for humans. Terrestrial biota are exposed to approximately the same airborne concentrations and for approximately the same lengths of time. Several points are highlighted below which support these exposure-dose relationships:

- **Terrestrial animals**. Terrestrial animals typically receive external and internal (i.e., inhaled) doses of ionizing radiation from air at rates similar to those experienced by humans. No major differences have been documented either in external doses due to submersion in air, or in internal doses due to intake and biological retention rates as a result of inhalation. Thus, if a DOE facility or site is in compliance with the dose limits for humans given above, total doses to terrestrial animals should be far below the much higher recommended limit of 0.1 rad/d.

- Inhalation doses were calculated for terrestrial animals over a range of body mass and metabolic rates (e.g., a marsh wren; a heron; a large elk) at allowable air concentrations at DOE sites. It was found that the air concentrations to which populations of these terrestrial animals would need to be exposed in order to reach the dose limit for terrestrial animals at DOE sites would need to be two to three orders of magnitude greater than the allowable air concentrations for humans.

- In general, internal dose to terrestrial animals is largely a function of ingestion rather than inhalation. Doses due to inhalation of airborne activity were taken into account in the

graded approach. The BCGs derived in the graded approach use appropriately measured lumped parameters (e.g., animal:food or animal:soil values) which implicitly include both ingestion and inhalation pathways to an organism. In cases where lumped parameter values were limited or unavailable, allometric relationships, to include those for inhalation, were used to derive the BCGs for riparian and terrestrial organism types. In cases where a user believes that inhalation could be a relatively important contributor to internal dose, the inhalation parameter can be appropriately modified in the analysis phase (i.e., site-specific analysis component) of the graded approach.

- **Terrestrial plants**. Terrestrial plants also typically receive external doses of ionizing radiation from air at rates similar to those experienced by humans. Hence, the above rationale for external exposure of terrestrial animals applies equally to external exposure of terrestrial plants, especially given the higher recommended limit of 1.0 rad/d for plants.

- In regard to absorption of airborne radionuclides by plants, there is no known mechanism for significant absorption of radionuclides in particulate form. Some radionuclides in gaseous form are absorbed, especially ^{3}H as tritiated water and ^{14}C as carbon dioxide. In both cases, however, the specific activity in the water and carbon of plants would approach those in the atmosphere, so there would be no magnification of the dose compared with that in humans. Moreover, for terrestrial plants, soils serve as the ultimate integrator of radionuclides originating and transported via the air pathway. Therefore, it is highly unlikely that populations of terrestrial plants could receive a significant dose due to absorption of airborne radionuclides. The much lower maximum doses from airborne emissions that are specified for humans would provide an adequate level of protection for terrestrial plants.

2.2.4 Derivation of Biota Concentration Guides For Active Air Releases

Although active air releases are unlikely to result in significant doses to terrestrial biota, the BDAC derived BCGs for air to further evaluate the potential contribution of the active air pathway to biota dose. Active air BCGs were derived using ecologically-based modeling approaches consistent with those used for the other media types in this technical standard. Inhalation and external exposure pathways were included. Allometric equations were used to assess exposure via inhalation, and do not consider other pathways of exposure (e.g., consumption of foodstuffs contaminated by deposition of radionuclides) – as these pathways are addressed and accounted for in the derivation of the water and soil BCGs. The magnitude of the active air BCGs were then compared relative to other media BCGs, and with derived concentration guides (DCG(air)) given in DOE 5400.5 for members of the general public. The human DCG values were decreased by a factor of 10 to represent the 10 mrem/y dose limit to the public required under NESHAPS for air emissions from DOE facilities. This comparison indicated that - for exposure to radionuclides from the active air pathway - the dose limits and derived concentration guides for radiation protection of humans are more restrictive than the

BCGs derived for radiation protection of biota. This analysis is consistent with and supports the assumptions and findings presented above in section 2.2.3.

2.2.5 Summary and Conclusions

Based on the foregoing discussions: (1) it is difficult to conceive of any credible circumstances under which populations of terrestrial animals and plants could receive a dose from exposure to radionuclides released through the active air pathway at DOE sites that would be more than a small fraction of applicable biota dose limits referenced in this technical standard; and (2) compliance with the biota dose limits for populations of terrestrial plants and animals can be evaluated without the explicit need to consider external and internal exposures from the active air pathway.

2.3 Aquatic Plants

There are no DOE or internationally-recommended dose limits established for aquatic plants, primarily due to lack of data on radiation effects to these organisms. Indirect means can be used to provide a general indication of the effects on aquatic plants relative to effects on other organisms. Consider the following:

- Few investigations have been conducted on the impact of ionizing radiation on aquatic plants (Woodhead 1998). There is a paucity of data in the literature regarding the radiosensitivity of aquatic plants, even though site-specific lumped parameter values (i.e., bioaccumulation factors) for accumulation of several radionuclides are available (Whicker et al. 1990, Cummins 1994, and Whicker et al. 1999).

- In general, one would expect substantially lower radiosensitivity in higher plants in comparison to the most sensitive birds, fishes and mammals (Whicker and Schultz 1982, and Whicker 1997). For these reasons, an evaluation that demonstrates protection of aquatic and riparian animals would provide an indication that aquatic plants are also likely protected.

- Alternatively, the aquatic animal spreadsheet can be used to calculate BCGs for aquatic plants. This is done by replacing the default B_{iv} values in the aquatic animal spreadsheet within the RAD-BCG Calculator with appropriate bioaccumulation factors (B_{iv}s) for aquatic plant species. The remaining default parameters and assumptions are unchanged. Calculating BCGs for aquatic plants in this manner, if needed, should be done in consultation with EH-412 and the BDAC Core Team.

2.4 Direct Measurement of Radiation Fields

It is first important to distinguish between ionizing radiation and radioactive isotopes (radionuclides). Ionizing radiation is defined as radiated energy that is energetic enough to

eject one or more orbital electrons from the target atom or molecule (i.e., the radiation ionizes the target). Ionization can produce free radicals, which are chemically unstable atoms or molecules that have an odd number of electrons. These highly reactive products scavenge electrons by breaking chemical bonds, including those in cell membranes and DNA molecules. Thus, ionizing radiation can cause cell death (e.g., oocyte death) and mutations (e.g., cancer). However, ionizing radiation generally does not cause ambient media or biological tissues to become radioactive, which only occurs via the transfer and accumulation of radionuclides. That is, exposing an organism to a radiation field does not result in the transfer of radionuclides and does not make the organism radioactive. It follows that an organism that simply passes through a radiation field does not then become a source of radionuclides or radiation to other organisms.

2.4.1 Considerations for Evaluating Doses to Biota around Accelerators or other Sources of Direct Radiation

Accelerator facilities pose little risk regarding environmental contamination. Emissions are mainly short-lived gases which do not accumulate in the environment. Therefore, compliance with the dose rate limits referenced in this technical standard is most efficiently accomplished by direct measurement and mapping of the radiation dose rate field outside the facility. This can be accomplished during routine radiation monitoring using the techniques normally employed by the facility. If the greatest dose rate in the field does not exceed 0.1 rad/d (1 mGy/d), the facility has demonstrated protection and no further action is required.

If the greatest dose rate in the field does exceed 0.1 rad/d (1 mGy/d), it does not immediately imply non-compliance. The dose limit is based on continuous exposure and radiation from accelerators is rarely continuous. The primary radiation field exists only when the accelerator is operating. In this case, dose assessors may wish to employ dose reduction factors accounting for the fraction of the day during which the dose rate field exists. If this technique is employed, it may also be important to ensure that maximum dose rates do not exceed 10 rad/d (100 mGy/d). According to the IAEA (1992), acute dose rates below this limit are very unlikely to produce persistent and measurable deleterious changes in populations or communities of terrestrial plants or animals.

Other considerations for direct measurement of radiation fields include:

- **Measurement technique.** The technique employed to measure the dose rate field should be appropriate for the type of radiation and sufficiently sensitive to demonstrate compliance with the limits.

- **Dimensions of the field.** For most accelerators, the greatest dose rate may be observed in line with the beam. However, if the beam is potentially scattered, it may be important to obtain a 3-dimensional map of the dose rate field.

- **Activation products.** If there is a potential for the creation of activation products in soil or water outside the accelerator building, assessors should consider applying the graded approach (i.e., using the BCGs) for contaminated media.

- **Biota intrusion.** Biota intrusion may be a problem in high-dose areas such as earthen beam stops, and this possibility should be investigated.

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3 Guidance on Spatial and Temporal Averaging Regarding Application of Biota Dose Limits and Mean Radionuclide Concentrations

Spatial and temporal variability relative to the distribution of contamination in the environment can be taken into account when evaluating doses to biota. This section provides guidance on spatial and temporal averaging regarding application of biota dose limits and mean radionuclide concentrations. The rationale used to define an evaluation area is an important aspect of any spatial averaging of radionuclide concentrations that may be applied in the graded approach. Guidance on defining areas over which radionuclide concentrations can be averaged to define an evaluation area is discussed in Module 2, Section 4.

3.1 Use of Time Averaging in Applying Dose Limits for Aquatic and Terrestrial Biota

The daily dose limits for aquatic and terrestrial biota are based on recommendations of the National Council on Radiation Protection and Measurements (NCRP 1991), the International Atomic Energy Agency (IAEA 1992), and a DOE workshop (Barnthouse 1995). The guidance presented in this section on the use of time averaging in applying the daily dose limits is based on the data on radiation effects in biota found in these reports and on the intended applicability of the recommended daily dose limits. The guidance is supported by radioecological studies at highly contaminated sites in the former Soviet Union (Polikarpov 1994).

Daily Dose Limits

The daily dose limits for biota are not intended to be applied to each day of exposure. Rather, the daily dose limits should be applied as averages over substantially longer time periods.

The dose limits for radiation protection of biota at DOE sites are expressed in terms of daily limits on absorbed dose. Expressing the standards in this way suggests that the dose limits apply to each day of exposure and, therefore, that compliance with the dose limits must be demonstrated on a daily basis. However, the information in the reports identified above clearly indicates that the daily dose limits for biota are not intended to be applied to each day of exposure. Rather, the daily dose limits should be applied as averages over substantially longer time periods.

3.1.1 Guidance on Time Averaging in Applying Daily Dose Limits

The guidance on the use of time averaging in applying the daily dose limits for biota assumes that compliance with the standards will be based in part on measurements of the concentrations of radionuclides in surface water, sediments, and surface soil. The following guidelines are offered:

- The estimated daily dose rates from exposure to contaminated surface water may be averaged over a period of approximately 1 month (30 days), and up to but not to exceed 1 year (365 days).

- The estimated daily dose rates from exposure to contaminated sediments or soil may be averaged over a period substantially longer than 1 month, but not to exceed 1 year (365 days).

The above guidelines are generally consistent with the frequency of sampling of surface water, sediments, and surface soil at DOE sites.

The different time periods for averaging daily doses from exposure to surface water and exposure to sediments or soil are based on considerations of the times over which radionuclide concentrations in these environmental compartments are likely to change significantly in response to short-term fluctuations in radionuclide concentrations in effluents. Retention times of radionuclides in the water column often are relatively short, due to such processes as deposition on sediments and flushing by natural flow. Therefore, radionuclide concentrations in surface water can change relatively rapidly (e.g., with more rapid change in lotic systems, and generally less rapid change in lentic systems). However, radionuclide concentrations in sediments or surface soil usually change more slowly because of sorption of radionuclides onto these media and the immobility of sediments or soils in most environments. Site-specific conditions (e.g., intermittent storm water flows; scour and transport of contaminated sediments resulting from seasonal occurrences such as high flow conditions) that may produce wide variations of exposure to receptors should also be considered in conjunction with the guidelines provided above when determining appropriate averaging periods.

3.1.2 Rationale for Guidance on Time Averaging

The guidance on the use of time averaging in applying the daily dose limits for biota is based on reviews and evaluations of existing data and discussions of daily dose limits in NCRP (1991), IAEA (1992), and Barnthouse (1995). The rationale for the guidance is summarized as follows:

- The daily dose limits for biota are intended to provide protection of whole populations of individual species, rather than individual members of the population. Furthermore, the primary health effect of concern in protecting whole populations of individual species is impairment of reproductive capability over the normal reproductive lifetime.

- The data on radiation effects in biota that provided the basis for the daily dose limits were obtained primarily from studies involving *chronic* exposure, in which the average dose rate in the population varied substantially, often by an order of magnitude or more, over exposure times ranging from several months to several years. In the studies involving chronic exposure, the dose rate in individual organisms also varied substantially due to

spatial inhomogeneities in the dose rate and/or the movement and burrowing habits of organisms.

- Based on studies involving short-term exposures, dose rates about 2-5 times higher than the daily limits for biota appear to be tolerable for short periods of time (e.g., 30 days) if the daily dose rate averaged over the lifetime of the exposed population is limited in accordance with the standards. Single acute doses about 10-30 times higher than the daily dose limit appear to be tolerable (a) if the recovery time between such doses is sufficiently long (e.g., 30-60 days) and (b) if the daily dose rate averaged over the lifetime of the exposed population is limited in accordance with the standards.

- The *average* doses in populations of study organisms was the primary basis for reporting dose-response relationships for deterministic effects, including early mortality and impairment of reproductive capability, and for developing standards for radiation exposure of biota. Thus, time averaging, as well as spatial averaging, of dose rates was inherent in the development of daily dose limits. The dose limits were not intended as limits for each day of exposure but, rather, as limits on the average daily dose rates encountered from conception through reproductive age. Therefore, averaging times as long as 1 year may be appropriate for reproducing members of populations of the most radiosensitive organisms (vertebrate animals and some higher plants).

- Radioecological studies at highly contaminated sites in the former Soviet Union (Polikarpov 1994) suggest that radiation effects are observed at the population and community level only for annual doses greater than about 400 rad (4 Gy) or an average daily dose of about 1 rad (0.01 Gy). Thus, effects attributable to radiation exposure were observed only for average daily doses over 1 year equal to the dose limit for aquatic animals and terrestrial plants and 10 times the dose limit for terrestrial animals.

All of these factors taken together suggest that applying the daily dose limits for biota as averages over a time period between 30 days and 1 year would provide adequate protection, especially when the time-dependence of most routine releases at DOE sites is taken into account.

3.2 Guidance on Spatial Variability in Applying Dose Limits

This section discusses how spatial variability in doses could be taken into account when applying daily dose limits for biota. General considerations and rationale regarding suitable approaches to selecting measured concentrations of radionuclides in environmental media (water, sediments, and soil) to be used when demonstrating compliance with the daily dose limits based on the screening models is presented here. Guidance on selecting measured

concentrations other than maximum values is also presented. The daily dose limits for biota are intended to provide protection of whole populations of individual species rather than individual members of a population that might experience a greater dose. Thus, given that exposures of a population normally would occur over a considerable area, some type of an average value of the concentrations of radionuclides in environmental media over the area occupied by the population would be suitable for purposes of demonstrating compliance with the daily dose limits. Also, because most of the scientific data underlying the evolution of the dose limits involved averaged responses to averaged dose rates, applying rational spatial averaging schemes for environmental media concentrations used in a biota dose evaluation would be appropriate.

Significant spatial variability in the doses to aquatic and terrestrial organisms may occur in environmental systems, due to two factors:

- *The spatial variability in the concentrations of radionuclides in different environmental media, due to dispersion and dilution during transport from localized sources and the spatial variability of processes that concentrate or immobilize radionuclides.*
- *Migration of organisms from or to areas of greater or lesser contamination.*

The screening methods developed in this technical standard are intended to be conservative in their approach to estimating dose rates per unit concentration of radionuclides in water, sediments, or soil. Similarly, for judging compliance with the daily dose limits for biota, some degree of conservatism also is warranted when initially selecting the values of measured concentrations of radionuclides in the environment to be used as input to the screening methods. For example, when protecting whole populations of individual species, it would be appropriately conservative to select initial radionuclide concentrations toward the upper end of the range of measured values at a variety of locations close to any sources. Indeed, this is the rationale for first using maximum radionuclide concentrations in environmental media in the general screening phase of the graded approach. In addition, because the area of habitation for many species will be considerably greater than the area of contamination, average values of radionuclide concentrations over the contaminated area should be conservative for purposes of complying with the dose limits, albeit to a lesser extent.

It is typically labor-intensive and potentially difficult to completely characterize the distribution of radionuclide concentrations in the environment, particularly in sediments and soil. This is particularly true if such characterizations have not already been conducted. It may be resource-intensive and/or difficult to determine the ranges of concentrations of radionuclides in the exposure environment, and to provide reliable estimates of statistical measures of the distribution of concentrations with location, including, for example, the mean (average value). Also, as noted previously, many species are highly mobile. Therefore, when limited environmental data are available, an approach to applying the daily dose limits for biota that relies on some form of statistical analysis may be unlikely to be more rigorous than a more qualitative and judgment-based approach to evaluating the data.

3.3 Guidance on Estimating Mean Values

Estimating mean values

To estimate mean values, it will be necessary to know the approximate boundaries of the site, and the approximate spatial and temporal distributions of the contaminant(s) at that site. As appropriate to the characteristics of the site and the contaminants present at the site, random, stratified random, or systematic sampling may be used to collect data for estimating mean values. A more qualitative and judgement-based approach to evaluating the data may also be used. See Module 2, Section 6, for related information.

For aquatic or terrestrial biota, compliance with applicable dose limits shall always be demonstrated by first comparing the maximum measured values of radionuclide concentrations in environmental media (water, sediments, and soil), as obtained from existing networks for environmental monitoring, with the default BCGs in the general screening phase. However, if maximum measured concentrations do not comply with the biota dose limits, then estimates of average concentrations over the evaluation area, determined as described in Module 2, Section 4, can be compared with the default BCGs as the first step in the site-specific screening phase. - Depending on the spatial coverage, quantity, or quality of the existing data, either judgement or statistical methods could be used to select average concentrations for comparison with the BCGs. In all cases, the approach to selecting the average values shall be documented. If average concentrations of radionuclides over the contaminated area exceed the default BCGs in the site-specific screening phase, then efforts to demonstrate compliance probably should focus on other aspects of the graded approach, such as reducing the degree of conservatism in the BCGs (e.g., generating more accurate, realistic site-specific BCGs using site-representative parameters as described in site-specific screening and site-specific analysis elements of the graded approach).

3.3.1 Adjustments to Account for Spatial and Temporal Distributions of Radionuclides in the Environment When Estimating Mean Concentrations

Location-specific data for individual radionuclides in specific environmental media are used in the screening process. When conducting a screening evaluation, it is important to use radionuclide concentrations that are estimated to be **mean values or greater than mean values** for the contaminated area. Only data at or above the mean are adequate for screening purposes because mean concentrations are assumed in this technical standard to approximate those concentrations to which a representative individual within a population would be exposed.

Available data may not be adequate to ascertain that radionuclide concentrations are likely at or above mean values for the contaminated area. Non-representative measurements may occur and result in values that are considerably higher (or lower) than the actual mean concentration. That is, concentrations are so far above the mean value that they falsely indicate that biota are receiving doses above the recommended limits, or so far below the mean value that they falsely indicate that biota are receiving doses below the recommended limits. In these cases, it is

acceptable to account for both spatial and temporal distributions of radionuclides in the environment when estimating mean values of radionuclides for use in site-specific screening.

Radionuclide concentrations can be adjusted to account for site-specific spatial and temporal factors that will bring them closer to mean values. Consider the following examples:

- If the source of radionuclides is an intermittent discharge to the environment, concentrations of radionuclides discharged to the receiving environment may be adjusted over time based on discharge records.

- A correction factor for exposure area or organism residence time may be applied in the site-specific analysis component to account for intermittent sources of exposure that would affect all receptors in the evaluation area, or to account for the movements of organisms in and out of the contaminated area over time, for example, because of seasonal migration or diurnal migration in and out of the contaminated area.

- If the contamination exhibits a decreasing gradient of concentration away from the source, then mean concentrations of contaminants within the contaminated area may be used, taking into account the intersections with distinct habitats as described in Module 2, Section 4. Where available contaminant data are comprehensive, it would be possible to accurately estimate the size of the contaminated area and the distribution of contamination within that area. Statistical methods given and/or referenced in Module 2, Section 6, may be used to calculate mean values. The statistical methods selected should be widely-used methods referenced in standard statistical texts and/or recommended by a qualified statistician. However, where contaminant data are not sufficiently comprehensive to conduct rigorous statistical analyses but provide a semi-quantitative basis for estimating mean values, subjective judgement may be used with justification.

- If the area being considered has been documented to have high background levels of naturally occurring radionuclides, these background levels may be taken into account when determining compliance of DOE activities with the recommended biota dose limits. For example, this may be an important consideration for the two isotopes of radium (see BCGs for Ra-226 and Ra-228, Tables 6.1 - 6.4 of Module 1). Background levels for water, soil and sediment media should be estimated based on data for the same or similar water, soil or sediment types in areas unaffected by facility effluents.

- If available data cannot be justified to be at or above mean values, or if the initial screening analysis suggests a false positive result, additional data on contaminants may need to be collected to obtain more realistic estimates of mean values. Either or both of the following types of data may be needed: (a) data on the spatial distribution of concentrations of radionuclides within the contaminated area; and (b) data on the size of the contaminated area.

Both of these types of data are needed for estimating the mean concentrations of contaminants that are assumed to approximate the concentrations that a representative individual would encounter. Although Module 2, Section 6, discusses methods for sampling biota, much of the general information on sampling design is relevant to collecting data on the concentrations of radionuclides in the environment and should be consulted. Additional information is found in the “Environmental Regulatory Guide for Radiological Effluent Monitoring and Environmental Surveillance" (DOE 1991) and the “Multi-Agency Radiation Survey and Site Investigation Manual (MARSSIM)" (DOD-DOE-EPA-NRC 2000). In cases where very little data are available on the distributions of radionuclide concentrations, a preliminary survey may be needed.

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4 Guidance for Defining the Evaluation Area

As stated in Module 1, Section 5.3, the approach in the general screening phase shall be to use maximum radionuclide concentration data applicable to the largest area of interest (e.g., the entire site). If the default BCGs in the general screening phase are exceeded, then mean radionuclide concentrations may be applied in the site-specific screening phase of the graded approach. The definition of the evaluation area is an important aspect of any spatial averaging of radionuclide concentrations that may be applied in the graded approach. This section provides an approach for defining the evaluation area which uses the intersections of contaminated areas and habitats to define the areas over which concentrations can be averaged. Refer to Module 2, Section 3 for guidance on spatial and temporal averaging of radionuclide concentrations.

4.1 General Considerations

The selection of an appropriate spatial area is governed by the principles of susceptibility and ecological relevance (EPA 1999). For large DOE sites, the entire site would, in most cases, be too large an evaluation area, because most of the biota on the reservation would not be exposed to the contamination. Biota which do not come into contact with contaminants, do not receive dose, and the inclusion of non-contaminated areas in the calculation of mean concentrations would result in low doses not representative of the actual impacts to the affected biota. On the other hand, the individual operable unit, waste trench, or contamination source would, in most cases, be too small to be ecologically meaningful. Although biota living in a 100 m^2 waste trench may be greatly affected by trench contaminants, their loss will likely have little impact on the population of small mammals in the region or on a broader scale ecosystem function. Beyond these limits, the scale of application depends greatly on site-specific conditions.

4.2 Step-by-Step Guidance

It is possible, however, to provide general guidance for selecting an appropriately scaled application area. This guidance is not meant to be prescriptive. Each step of the process involves a significant element of professional judgement and requires appropriate justification and documentation. In particular, the environmental monitoring organization at the site will be required to determine, justify, and document appropriate boundaries for areas with similar environmental concentrations of the same radionuclides (referred to hereafter as contaminated areas). Similarly, the site ecologists will be required to determine, justify, and document appropriate boundaries of similar habitat types.

The intersection of contaminated areas and habitats define the areas over which concentrations can be averaged if use of the maximum concentrations at any locations does not show compliance with the dose limits. This kind of analysis is most easily done using area maps, and Geographic Information Systems (GIS) will prove an invaluable tool. The following steps can be applied to determine this intersection.

1. **Determine whether this method is necessary.** First, use the default BCGs in the general screening phase with the input contaminant concentrations set at the highest concentrations found in your area of interest (e.g., the entire site). If you pass the general screening phase, no further consideration is necessary. If use of the maximum concentrations at any locations does not pass the general screening phase, then proceed below.

   The following steps of the process center around determining the boundaries of the contaminated areas and their relationship to ecological habitat types. This will likely involve consideration of: (a) boundaries presented by the quality, quantity, and distribution of available environmental radionuclide data, and resulting from the design of the site environmental monitoring and surveillance program; (b) boundaries presented by the susceptibility, ecological relevance, and habitat of receptors relative to the radionuclide contamination; and (c) boundaries resulting from the management and administration of facilities and operations areas on the site (e.g., location and extent of waste management facilities, production facilities, operable units, and operations areas).

2. **Determine and map the boundaries of the contaminated areas.** One possible set of boundaries might be the background isopleths of a contamination plume, but there are other possibilities, particularly if the radionuclides present, their historical deposition, or their present environmental concentrations differ from location to location. The site environmental monitoring organization should determine the most meaningful and justifiable boundaries for their site.

3. **Determine and map the boundaries of discrete habitat types.** Within a habitat type, one assumes that ecological structure and function are sufficiently homogeneous to be represented by a single parameter and that the species of concern are distributed throughout the habitat type. Between habitat types one assumes that structure and function are dissimilar. The site ecologists should use best professional judgement and all available data to justify these habitat boundaries.

4. **Overlay the maps and identify the intersections.** Each area of discrete habitat that lies within a discrete contaminated area can be appropriately defined as an assessment area. This may occur in several ways:

- A single contaminated area may be completely covered by a single habitat patch (Figure 4.1 (a)). In this case, the contaminated area bounds the assessment area. An example of this kind of intersection might be a small pond with uniformly contaminated sediment.

- A single contaminated area might also intersect multiple habitat patches (Figure 4.1 (b)). This might be the case at any site which releases airborne contaminants from a stack. In this case, there will be multiple assessment areas bounded by habitat type.

- Multiple contaminated areas of the same type may intersect a single discrete habitat patch (Figure 4.1 (c)), in which case it is acceptable to integrate or average over multiple contaminated areas within a single habitat type.

- Finally, there may be multiple habitat patches of the same type which intersect one or more areas with radionuclides in the same environmental concentrations (Figure 4.2). In this case, arguing that patches of the same type have similar species assemblages and similar structure and function, these intersections could be assumed to be one assessment area, even though they are separated in space.

In all these examples, it is important that contamination levels or parameters only be averaged over the intersection of the contaminated area and the habitat type of interest and not the areas between the intersection. If the areas outside the intersection were included, the averages would not likely be representative of the habitat type and/or contaminant levels of interest. The contaminated areas outside this intersection will be included in a different intersection of habitat type and contaminated area.

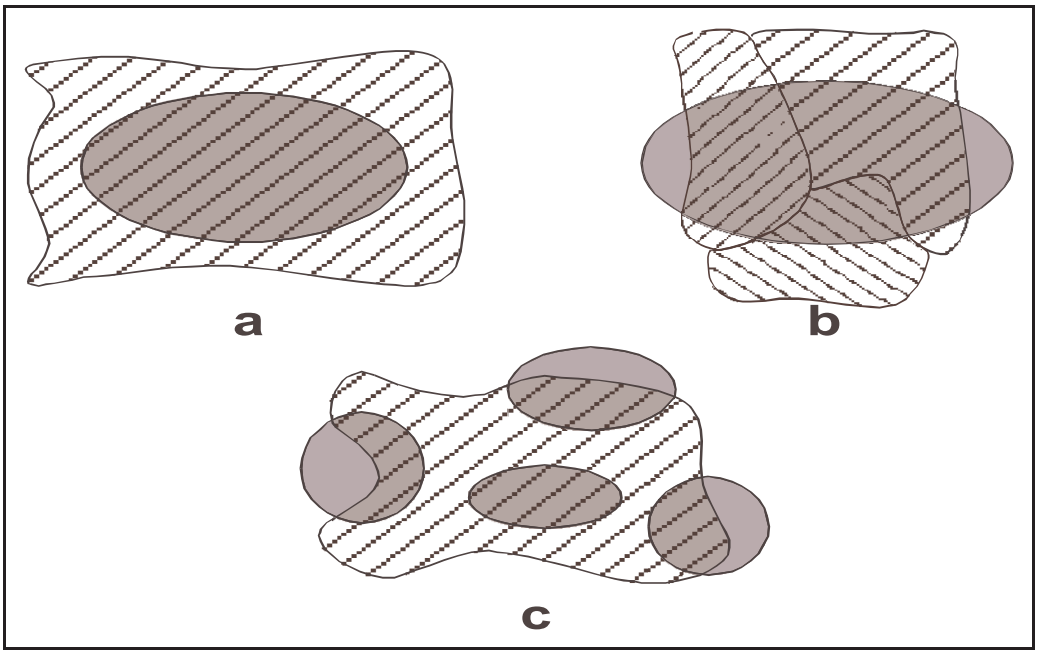

Figure 4.1 Hypothetical maps of contaminated areas and discrete habitat used to determine appropriately scaled assessment areas. Shading indicates contaminated areas. The cross-hatching indicates habitat types. Three cases are considered: **(a)** a single contaminated area intersects a single habitat patch; **(b)** a single contaminated area intersects multiple habitat patches; **(c)** multiple small contaminated areas intersect a single large habitat patch.

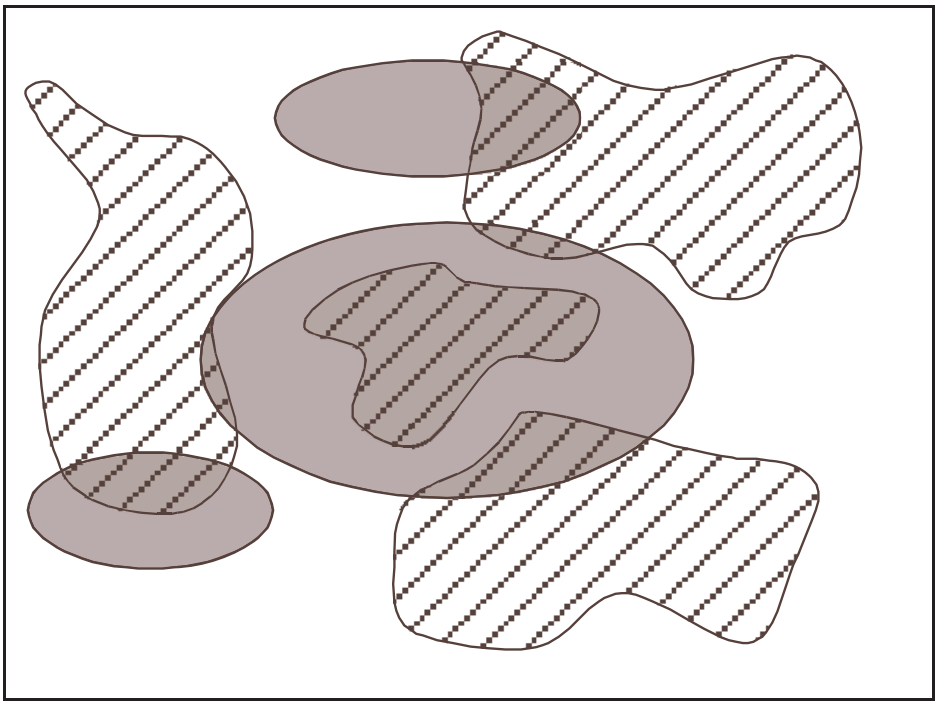

Figure 4.2 A hypothetical map of multiple areas with the same contamination intersecting multiple patches of the same discrete habitat type used to determine appropriately scaled assessment areas.

5 Guidance on Soil Sampling Relative to Plant Rooting Depths

In terrestrial environments, particular attention should be directed toward assessing whether plant roots are penetrating through relatively clean surface soils into subsurface zones that are contaminated with radionuclides. When this condition exists, plants will transport radionuclides from the subsurface into the vegetation canopy (for example, see Rickard and Kirby 1987). Potential for exposure via this route is considerable, as many plants have rooting depths in excess of 10 m (see Foxx et al. 1984, Canadell et al. 1996, and Jackson et al. 1996). Data from surface soil samples will not indicate that the plants and the biota dependent on those plants are receiving significant doses of ionizing radiation. The condition will be detectable, however, because the concentrations of radionuclides in plant tissues will exceed the concentrations that are predicted by concentration ratios for surface soils to plants. Therefore, it may be necessary to sample deep-rooted plant tissues directly in any areas where subsurface contamination is known or suspected to exist, for example above waste sites and plumes of contaminated ground water. Guidance on rooting depths and designing a survey to assess potential vertical transport of radionuclides by plants is provided in this section.

5.1 Overview of the Problem

DOE sites typically have numerous areas of subsurface contamination, for example cribs, trenches, solid waste sites, contaminated soil columns, and contaminated ground water plumes (see DOE 1995; 1996). Most of these areas of subsurface contamination have been mapped, although surprises do occur on occasion. In many cases, contaminants including radionuclides are moving through the subsurface environment. With the exception of ground water, however, the subsurface environment is generally not sampled. In particular, soil samples are typically collected only at the surface in response to a need for information about atmospheric deposition of radioactive fallout from operations and past nuclear tests. These samples do not necessarily indicate types and levels of contamination below the surface.

Incomplete and imperfect data on contamination of the subsurface environment can be problematic for assessing radiation doses to plants because plants extend their roots into the subsurface environment and can transport radionuclides from the soil column and ground water up into their canopies (see Rickard and Kirby 1987). This route of transport and exposure may not be apparent from surface soil samples. However, it can be detected by comparing co-located concentrations of radionuclides in surface soil with concentrations in plant tissues. Concentrations in plant tissues that are higher than expected based on surface soil concentrations and the application of the appropriate soil to plant concentration ratio strongly suggest that vertical transport by plants is occurring. When vertical transport occurs, the plants themselves are receiving an internal dose, as are organisms at higher trophic levels that are dependent on those plants (e.g., herbivores and predators of those herbivores).

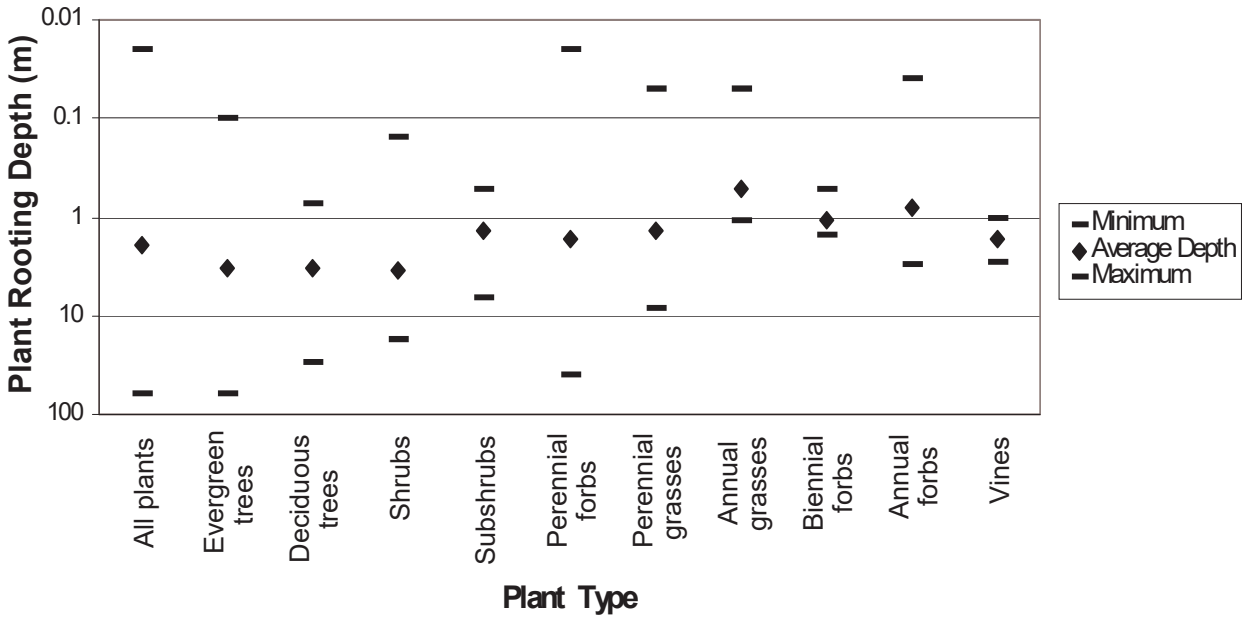

Figure 5.1 Average Rooting Depth by Plant Type
(source data from Foxx et al. 1984)

Because of the potential for transport and exposure via this mechanism, if deep-rooted plant receptors are present in areas of known or suspect sub-surface contamination, plant tissues may need to be sampled even if surface soil samples do not indicate the presence of radioactive materials. It is not necessary to collect additional subsurface soil or ground water samples for analysis because the plants themselves are the best indicators of uptake and transport from the subsurface to the surface. A statistically sound sampling and analysis plan will yield good estimates of the area over which transport is occurring, and tissue burdens of radioactive materials within the plants.

5.2 Plant Rooting Depths

Plant roots can extend considerable depths into the subsurface, as indicated in Figure 5.1 and Table 5.1. Ranges of rooting depths vary considerably among plant types (Figure 5.1) and individual species. For these reasons, the data in Figure 5.1 and Table 5.1 are only a general representation of rooting depth. Regional or local data on rooting depths of individual plant species should be consulted whenever available. Foxx et al. (1984) is presently the best review and compilation of data on rooting depths for the contiguous 48 states. More recent references that may be consulted include Klepper et al. (1985), Tierney and Foxx (1987), Gilman (1989), Breda et al. (1995), Parker and Van Lear (1996), Jackson et al. (1996), Canadell et al. (1996), and Gerzabek et al. (1998).

Table 5.1 Average and Ranges of Rooting Depths by Plant Type (m)

Life Form	Range		Average Depth	Sigma
	high	low		
All plants	60.96	0.02	1.9	3.3
Evergreen trees	60.96	0.1	3.36	9.54
Deciduous trees	30	0.73	3.32	4.51
Shrubs	17.37	0.15	3.50	3.5
Subshrubs	6.4	0.51	1.40	1
Perennial forbs	39.32	0.02	1.70	2.5
Perennial grasses	8.23	0.05	1.40	0.9
Annual grasses	1.10	0.05	0.52	0.41
Biennial forbs	1.52	0.53	1.07	0.38
Annual forbs	3	0.04	0.8	0.8
Vines	2.8	1.02	1.68	0.78
All trees	**60.96**	**0.1**	**3.34**	**6.11**
All perennials	**39.32**	**0.02**	**1.6**	**2**

5.3 Consider the Need for Site-Specific Plant Uptake Factors

In some cases, it may be desirable to calculate site-specific uptake factors. For example, published uptake factors may be of questionable utility, resulting in a need to derive site-specific uptake factors. Examples include uptake factors that were derived exclusively in dissimilar climatic regions or soil types, factors for which reported values are highly variable, and factors based on very different plant taxa. In other cases, it may be possible to derive site-specific plant uptake factors easily because co-located plant and soil and/or groundwater samples have been taken (e.g., from routine monitoring programs), and the data are readily available. It is essential that soil and subsurface conditions including all major contaminant sources be reasonably well understood and that data from relevant locations and media (i.e., soil and/or groundwater) be available or be collected.

5.4 Survey Design Considerations

It is not the intent of this section to provide detailed guidance on sampling plant tissues for radionuclide analysis. However, the following general considerations are offered as a starting point for designing and conducting a plant tissue-sampling program that will generate data on tissue burdens of radionuclides.

- **Plant species.** When sampling to determine whether a transport problem exists at a given location, the sampling program should be designed to sample multiple species with rooting depths that range from the near surface to the greatest depths possible. Multiple species will minimize the possibility that the contaminated zone is above or below the root zone of any single species. Plants in riparian areas should not be overlooked, as deep-rooted riparian species will have the potential to intercept contamination at considerable depth, while shallow-rooted riparian species will intercept contamination where ground water is discharged into surface water.

- **Target radionuclides.** When selecting target radionuclides for analysis, information on the history of the site will be important for determining *a priori* what radionuclides may be present and should be considered in the survey design. For example, information on the radionuclides in a subsurface ground water plume that is suspected to be under the vegetated area will be important. Hence, all information on radionuclides known or suspected to exist in a given area should be reviewed before the survey is designed.

- **Data quality objectives.** Sampling should be designed and samples collected to meet or exceed specified data quality objectives for the survey. Specification of data quality objectives will help ensure that plant tissue data are of sufficiently high quality to ensure that reasonable accurate estimates of doses can be derived from them using the methods in this handbook. Refer to Gilbert (1987), EPA (1994), and Bilyard et al. (1997) for information on the data quality objectives process. In most cases where vertical transport is suspected, data quality objectives will need to specify that mean concentrations of specific radionuclides in plant tissues can be estimated with an acceptable, specified degree of precision.

- **What to sample.** The physical and chemical properties of the target radionuclides will be important to the survey design. For example, radionuclides in volatile (e.g., H-3 as gas or tritiated water, C-14, I-129), semi-volatile (e.g., Cs-137, at higher temperatures), and solid states (e.g., all U and Pu) may require different handling and/or analysis procedures. In addition, they will differentially partition among the parts of the receptor plants. Solids and semi-volatiles will concentrate in roots>stems>leaves>seeds. Volatile radionuclides will partition differently. For example, H-3 as tritiated water will exhibit highest concentrations in leaves, while C-14 will be highest in woody tissues such as stems and roots, and I-129 will be higher in leaves than in stems.

  Characteristics of the sampled vegetation are also important to survey design. For example, more mature plants will have better developed root systems with greater surface areas available for absorption of radioactive substances, and may exhibit higher concentrations of radionuclides in their tissues. For radionuclides that exhibit highest concentrations in the leaves, sampling will necessarily be restricted to the growing season.

- **Sample numbers and sizes.** Plants exhibit considerable inter-individual variability. Hence, several plants should be sampled at each location. Samples may be pooled within locations to obtain the mass needed for analysis consistent with data quality objectives. Analytical laboratories may need to be consulted prior to sampling to determine the minimum masses needed for analyses to meet specified detection limits. Sample masses are generally on the order of 10 – 50 g dry weight for analytes other than tritiated water. Samples for tritiated water are generally on the order of 20 – 100 g dry weight.

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6 Guidance on Biota Sampling to Support Implementation of the Graded Approach

This section provides guidance and summarizes important issues associated with collecting biological samples for dosimetric assessments of biota. Guidance is provided on sampling biota to estimate mean radionuclide body burdens in representative individuals of a population. This section does not address sampling to estimate effects (e.g., reduced species richness or abundance). The sampling methods discussed here are to estimate the body burdens of radionuclides in biota. These data may be used to estimate the internal dose to the sampled organisms and the ingestion dose to receptors that consume the sampled organisms.

This guidance is intended to supplement and complement the guidance presented in the *Environmental Regulatory Guide for Radiological Effluent Monitoring and Environmental Surveillance* (DOE 1991), hereafter referred to as the Environmental Surveillance guidance. The biological samples collected in accordance with the Environmental Surveillance guidance are intended for assessing the dose to humans from ingestion of contaminated foodstuffs. These samples can also be used for preliminary dose assessments for biota. However, the data collected for human dosimetric assessments may not be representative of the internal or ingestion doses to ecological receptors. The types of organisms collected and the potential exposure pathways for the collected organisms should be evaluated to determine the appropriateness of these data for use in assessments for ecological receptors.

The recommended approach to biota sampling consists of six major steps, which are shown in Figure 6.1 and described in this section. The process begins with a clear definition of the scope and objectives of the sampling effort. This includes selecting appropriate receptors, defining the spatial and temporal context of the project, and identifying the data required for the dosimetric assessment for the non-human receptors. Based on these decisions, sampling methods and a sampling design are selected. The biota samples are collected and analyzed, possibly in a multi-phased effort to allow for optimization of the sampling plan. The resulting data are statistically summarized and the site data are compared to the background data, as appropriate. Ultimately, the biota concentration data are incorporated into the dosimetric assessments performed in accordance with the recommendations presented elsewhere in this technical standard.

6.1 An Important Note about Biota Sampling and Temporal Variation

Biota are considered a valuable sampling tool because they integrate exposures over time and, for mobile organisms, space. This is particularly helpful when the distribution of abiotic media samples may be inadequate to characterize the variation in exposure. For example, high concentrations of soil contamination (hot spots) may be missed by a soil sampling program but included in exposures contributing to the measured body burden of a terrestrial organism. In this way, measured body burdens help account for spatial variations in contaminant concentrations.

However, biota sampling is not a cure-all for contaminant monitoring. In particular, the kinetics of accumulation and depuration must be considered when evaluating the usefulness of body burden data for situations in which temporal variations in contaminant concentrations occur. For example, concentrations in flowing water may be highly variable through time, making it difficult to estimate exposures for aquatic biota. Fish samples will typically provide a good estimate of the actual exposures. The time over which this exposure is integrated depends on the clearance rate of the elements measured. Therefore, if fish samples are collected once annually but the element is rapidly eliminated from the fish, then the measured concentration is highly dependent on when the exposures occurred. For the aqueous exposure example, summer low flow conditions may result in elevated exposure concentrations with concomitant increases in tissue concentrations. But if tissue samples are only collected in the spring, these elevated body burdens will be missed if the biological half-life is on the order of days or weeks. Therefore, the assessor should take into account the expected variation in exposure through time and the accumulation and depuration rates for the radionuclides of concern.

The first issue to be evaluated is the temporal variation in exposure concentrations. Are the concentrations cyclical or relatively stable? If they are relatively stable, as for existing surface soil contamination, then the kinetics of accumulation are unlikely to influence the measured body burdens and can, therefore, be disregarded for purposes of screening. If the concentrations of contaminants are periodic, as for streams receiving contaminated discharges, then the frequency and duration of elevated exposure concentrations must be considered. At this point, the assessor should acquire relevant estimates of the accumulation and depuration rates of the radionuclides of concern from the literature. To the extent practicable, biological samples should be collected after the organism has reached equilibrium with the elevated exposure concentrations and before significant depuration has occurred. If equilibrium is not expected to occur, then biota sampling should occur at the end of the period of elevated concentrations. In the absence of relevant accumulation and depuration information, biological samples should be collected at the end of the period of elevated concentrations, to the extent practicable.

6.2 General Planning Considerations

General planning considerations include use of the Data Quality Objectives (DQO) process, selection of receptor species, variability of exposure, definition of representative population exposures, and use of dosimetry models.

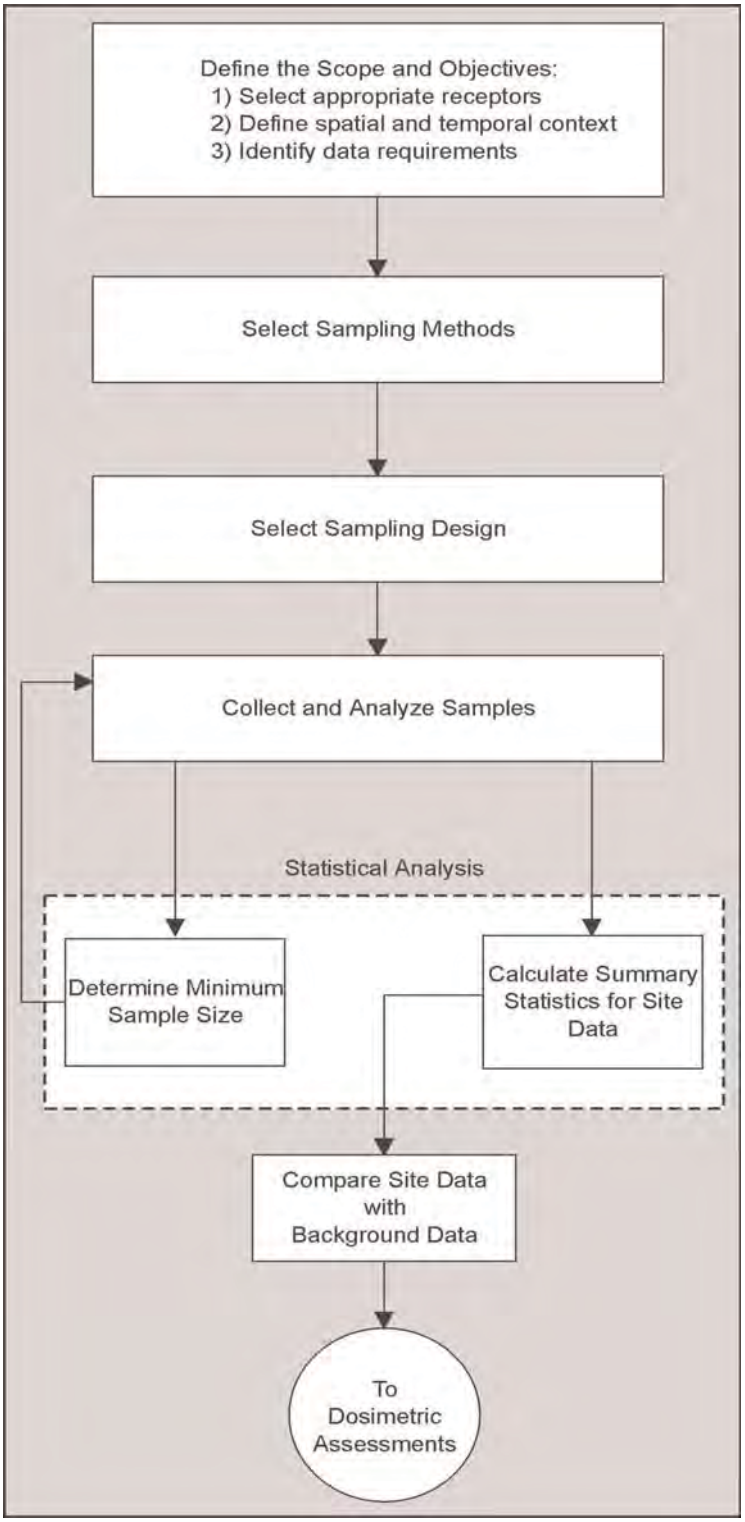

Figure 6.1 Flow Diagram for Collecting Biological Samples to Produce Data for Dosimetric Assessments of Biota

6.2.1 Use of Data Quality Objectives

The biota sampling plan to support biota dose assessments must begin with a clear definition of the study objectives and decisions to be made. Defining these objectives is best accomplished through the use of the DQO process, as set forth in related DOE guidance (Bilyard et al. 1997). This process compels investigators to fully consider the intended uses of the data they will collect, ensures that the data users (e.g., including radioecologists, risk assessors, site managers, and regulators) have considered the methods they will use to evaluate the data and requires that the decision makers understand and agree with the objectives and limitations of the sampling effort. At a minimum, the plan should define the populations to be evaluated, select the receptors to be sampled, and determine the acceptable level of uncertainty associated with the estimates of body burdens.

6.2.2 Selection of Receptor Species Sampled

The most appropriate receptors to collect are those that meet the criteria for appropriate assessment endpoints. These criteria include ecological relevance and relevance to management goals (EPA 1988), susceptibility to irradiation, and a relatively high tendency for bioaccumulation. Selection based on ecological relevance is not unique to the evaluation of radionuclides and is not discussed further in this technical standard (see EPA 1998 and Suter 1993). Endpoints selected to meet management goals typically include species that are protected (e.g., threatened and endangered species), economically important (e.g., salmon), and culturally valued (e.g., medicinal plants used by Native Americans). The more general management goal of protecting all other populations of biota should be met if care is taken to select susceptible and ecologically relevant endpoints.

Susceptibility to irradiation is critical to the selection of species to be sampled. An organism is considered susceptible if it is sensitive and exposed (EPA 1998). How readily an organism is affected by radiation (i.e., its radiosensitivity) can vary by one or more orders of magnitude among phylogenetically similar species (UNSCEAR 1996; see also Module 1, Section 1). However, vertebrates and higher plants are generally more radiosensitive than invertebrates and lower plants (UNSCEAR 1996). It is protection of these more evolved organisms that is the basis of the acceptable dose limits and the focus of this technical standard (see Module 1, Section 1 in this technical standard and NCRP 1991, IAEA 1992, and Barnthouse 1995).

Radiosensitivity within these more general classifications has been reviewed elsewhere (UNSCEAR 1996). Unfortunately, the available data are too sparse to aid reliably in discriminating among similar species at a site for the purposes of biota sampling. Two exceptions are worth noting. First, salmonids are the most sensitive fishes that have been tested to date. Second, pine trees (*Pinus spp*.) are among the most sensitive plants, with sensitivity being correlated with the relatively large chromosomes of these species (IAEA 1992). Moss-lichen communities are the most resistant, with woody and herbaceous vascular plants ranging between pines and lichens (IAEA 1992).

Exposure is an endpoint selection criterion that is frequently used synonymously with sensitivity. While highly exposed biota may also be sensitive, this is not necessarily so. Because radiosensitivity is poorly known for many potential endpoints, those species expected to experience high exposure are frequently selected. Determination of exposure is based on two types of information: (a) the expected isotopes, sources, fates, and transport processes at the site; and (b) the behavior and habitat requirements of the biota at the site. This information is then used to develop the conceptual site model for exposure. Although exposures will vary by site, two general considerations are worth noting. First, receptors with small home ranges relative to the defined sampling area are preferred because they will be more exposed to the radionuclides at the site than will wide-ranging and migratory receptors. That is, the quality of the site-specific bioaccumulation factors (B_{iv}) is largely determined by the representativeness of the exposure concentrations. Second, contaminants are often localized in particular environmental media (e.g., cesium in soil and sediments, and tritium in water). Receptors with behaviors that increase their contact with those media should be preferred. For example, bottom-feeding fish may accumulate more cesium than fish feeding primarily in the water column (IAEA 1994).

6.2.3 Variability of Exposure

Exposure of the selected receptor may vary temporally and spatially. Exposure may vary through time for several reasons. The radionuclide concentration in the receptor may not have reached equilibrium with the ambient media if either the sources or the uptake by the receptor are variable relative to the physical and biological half-lives of the radionuclide. For example, contaminant discharges may vary seasonally while uptake by plants (especially annual plants) will be controlled by the growing season. Also, the foods that are available may have different tissue concentrations. Cesium levels in roe deer (*Capreolus capreolus*), for example, were found to be highest in August and September when fungi are most prevalent because fungi accumulate more cesium than the herbs and grasses that the deer otherwise consumed (IAEA 1994). At a minimum, and to the extent practical, sampling should be timed to coincide with the expected maximum tissue concentrations. It must be recognized that this is a biased sampling design, resulting in the maximum annual internal exposure to the representative individual of the population. The representative annual internal exposure to the representative individual of the population would require repeated sampling throughout the year. This is desirable, but may be impractical to implement and unnecessary to achieve the DQOs. Approaches to address this source of variation should also incorporate the recommendations on time averaging presented in Module 2, Section 3.

Exposure also may vary through space at ecologically relevant scales. There may be a contamination gradient away from a source (e.g., discharges to water or air) or a highly heterogeneous distribution resulting from complex fate and transport processes (e.g., fluvial and alluvial deposition of contaminated sediment). Exposure may also vary due to the discontinuity of the spatial distribution of contamination and habitat suitable for specific receptors. For example, the magnitude of exposure experienced by an ecological receptor is a

function of the overlap of contamination and habitat (Module 2, Section 4). If contamination and suitable habitat do not overlap spatially, exposure is unlikely. Sampling designs that account for these issues are presented in Section 6.3 of this Module.

6.2.4 Representative Population Exposures

It has been suggested that the acceptable Dose Rate Guidelines are "applicable to representative rather than maximally exposed individuals" (Barnthouse 1995). For the purposes of this section, it is assumed that representativeness refers to exposure within a population, not exposure among all populations at a site. It also is important to realize that representativeness does not refer to radiosensitivity within or among populations. Rather, it is likely that a limited number of populations would be sampled with an emphasis on those that are expected to be most exposed and sensitive, to the extent practical. The alternative is to demonstrate that the representative individuals of the representative populations were sampled, which would require much more extensive sampling. Hence, the expected reductions in uncertainty must be weighed against the costs of additional sampling.

It may be appropriate to define the receptor "population" to be sampled to include multiple species that are expected to be similarly sensitive and exposed (e.g., ground-feeding herbivores). The most common approach is to group organisms by trophic group or feeding guild. Combining species is typically done to increase the number of sampling units or to obtain the sample mass required for analysis. The disadvantage is that this may increase the variability of the results. For example, shrews are known to ingest considerably more soil than herbivorous small mammals (Talmage and Walton 1993). Hence, it is important to carefully consider any expected differences in exposure.

6.2.5 Dosimetry Models

An important planning question is, "How will the internal concentrations be used to estimate dose rates?" The dosimetric models available for biota are limited and relatively simplistic in design. Isotopic whole-body concentrations for fish and wildlife and vegetative- or reproductive-tissue concentrations for plants are generally recommended and sufficient for these models.

6.2.5.1 Aquatic and Terrestrial Vertebrates

The simplest approach is to modify the general screening model in this technical standard to better reflect the actual exposures at the site. The screening method makes no assumptions about the shape of the organism (e.g., an ellipsoid with specific dimensions) or the distribution of isotopes within the organism. It may be possible to improve the estimated internal dose rate by developing site-specific B_{iv}s that can be substituted into the general screening method. Indeed, this is what occurs in the site-specific screening component of the analysis phase of the graded approach. Given the non-dimensional nature of the screening method, a B_{iv} based on whole-body concentrations would be sufficient for this approach.

Whole-body concentrations also are sufficient for point-source dose distribution models that assume a uniform distribution within the organism and a specific geometry (NCRP 1991 and Baker and Soldat 1992). Mechanically homogenizing the whole organism dilutes any high-concentration tissues with lower-concentration tissues. This approach yields the average whole-body dose. The resulting whole body concentration would underestimate the actual dose to highly contaminated tissue, assuming that the emitted radiations would be absorbed primarily within that tissue (e.g., alpha particles and weak beta emissions). This uncertainty could only be reduced by using an exposure model that explicitly accounts for the non-uniform contaminant distribution.

Detailed dosimetric models are not available for most kinds of biota (Barnthouse 1995). Such models would account for intra-organism distribution of radionuclides, the penetration of various radioactive particles in a variety of tissues, and the geometry of the organism. In the absence of a comprehensive research and development program, dosimetry for biota will continue to be limited to the more simplistic and conservative dosimetric models that assume uniform distribution within the organism. These models are assumed to be conservative because, in part, the assumption of uniform contamination is unlikely to underestimate the actual dose to the tissues of concern (i.e., reproductive organs), given two conditions. One condition is that the radionuclide of concern must not be preferentially localized 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 animal data (Garten 1981, Garten et al. 1987, and Kaye and Dunaway 1962). The second condition is that the acceptable doses to the reproductive tissues should be comparable to the acceptable whole-body doses. 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., Cs-137 or Co-60), which is generally the case for biota (see NCRP 1991 and IAEA 1992). 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.

Concentrations in muscle tissue are commonly used to calculate dietary exposures for humans (DOE 1991). If biological samples are intended to be used to estimate both human and non-human exposures, then both muscle and carcass should be analyzed for at least some of the samples, as is practicable. The use of muscle tissue alone may underestimate the B_{iv} for non-uniformly distributed elements. This is of particular concern when estimating food-chain transfers for biota; wildlife generally consume the entire organism, not just the muscle tissue. Hence, whole-body concentrations are generally the appropriate measurements for estimating food chain transfers to biota.

6.2.5.2 Terrestrial Plants

Plant concentrations are commonly based on individual tissues rather than the whole organism (e.g., including roots and woody stems). Reproductive and growing vegetative tissues are recommended because they are sensitive and the effects data are based primarily on exposure to high-energy photons (IAEA 1992). That is, the site-specific dose to these tissues should be consistent with the doses used to estimate acceptable radiation limits. A comprehensive sampling effort would include both vegetative and reproductive tissues. If schedule and resources do not allow for this, then selection of the tissues to be sampled should consider the life history and physiology of the chosen plant species. For example, metals in general are found at higher concentrations in foliage than in fruits and seeds (Greenleaf-Jenkins and Zasoski 1986, Sadana and Singh 1987, Bysshe 1988, and Jiang and Singh 1994). However, the available data are far too limited to generalize among all radionuclides and plant species.

In addition to the dose to plants, radionuclide concentrations in plants can be used to improve the dose estimate for receptors at higher trophic levels (e.g., herbivores and omnivores). Selection of the plant species and tissues to be sampled must consider the life history, physiology, and feeding preferences of the representative consumers.

6.2.5.3 Analytical Requirements

A general sample preparation issue that should be considered is whether or not external contamination is removed prior to analysis. On one hand, it would be prudent not to wash external contamination from biota tissues prior to assay, as this would provide a more conservative estimate of biota dose. On the other hand, including deposited contamination in biota samples may be counter to the purpose of collecting site biota in order to improve the reliability of the B_{iv} and dietary exposure estimates. Although wildlife generally do not wash their food, dietary exposure models often include contaminated soil as a separate variable (Sample et al. 1997). Failure to remove external contamination would overestimate dietary exposures if such models are used in their original form. Thus, the user should carefully consider the exposure pathways to be included in calculating the B_{iv}, and the types of models that tissue concentration data may be applied in, when deciding on the inclusion or exclusion of external contamination. See also Section 6.4.3.2, Sample Handling, of this Module.

6.2.5.4 Other Data Needs

Collecting biota is only one component of any sampling plan intended to refine the dose estimates produced by the graded approach; biota concentrations can only be used to improve the estimated internal dose. External exposures must also be considered and may be an important pathway for gamma-emitters (e.g., dose to aquatic biota from cesium in sediment). At a minimum, the external dose rates from the screening method could be used in conjunction with the site-specific internal exposures. It is important to consider past or planned environmental sampling with respect to the planned biota sampling to ensure compatibility of

sampling designs. That is, site-specific bioaccumulation factors are best derived from co-located soil, sediment, and water radionuclide concentration data and biota samples. This approach reduces the uncertainty of the bioaccumulation factors by ensuring that the ambient media concentrations used to derive them are representative of the concentrations to which the sampled organisms were exposed. This is straightforward for relatively immobile receptors (e.g., plants and soil invertebrates) exposed to relatively immobile media (e.g., soil). Reducing this uncertainty in bioaccumulation factors derived from mobile media (e.g., water) or for mobile receptors (e.g., fish and small mammals) requires more extensive sampling protocols, which should be evaluated as part of the DQO process.

6.3 Sampling Design and Statistical Methods

Many excellent texts have been written concerning sampling design and statistics, and it is beyond the scope of this guidance to reiterate these texts. As you proceed in planning and performing field sampling, you may refer to these texts for additional information concerning the topics outlined below. Recommended references for general statistics include Snedecor and Cochran (1980), Sokal and Rolf (1981), Dowdy and Wearden (1983), Zar (1984), and Newman (1995). Discussions of the application of statistical methods to contamination studies are provided by Provost (1984), Gilbert (1987), and the Washington State Department of Ecology (WADOE 1992). Green (1979) is the pre-eminent text for sampling design for ecological field studies. Krebs (1989) provides additional discussion of methods for the collection and analysis of ecological data. For application to the DOE graded approach, the following discussion is presented in three parts: sampling considerations, statistical considerations, and suggestions for dealing with uncontrollable factors that influence sampling and analysis.

6.3.1 Sampling Considerations

For sampling, population definitions, sampling units, and sampling design must be considered.

6.3.1.1 Definition of Population

The population represents the group from which samples are to be taken and about which conclusions will be made. The most critical component in sampling is to define the population of interest. In the context of this guidance, the population of interest is the aggregation of animals or plants that are resident at the radionuclide contaminated site (EPA 1998). This population must be defined in terms of space (both of the site and in biological terms), time, and receptor species. Only by defining the population of interest can the appropriate samples be collected to determine the body burden of a representative individual.

6.3.1.2 Sampling Units

Sampling units represent the unit of material that is collected in an effort to draw inferences about the population. Sampling units may be naturally occurring (e.g., whole animals or parts of animals) or artificially derived (e.g., composite samples from quadrats). Sampling units must cover the whole population and must be independent, i.e., they cannot overlap (Krebs 1989). For this guidance, sampling units are likely to consist of whole organisms (e.g., vertebrates or plants) or composites of biotic material (e.g., plants or invertebrates) collected from within quadrats of other sampling devices. It is important to point out that sampling units are not samples. A sample is a collection of sampling units. For example, if individual small mammals represent the sampling unit, 20 small mammals collected from a given area represent the sample (Ludwig and Reynolds 1988). If these 20 small mammals were collected using an appropriate and valid design, the resulting data distribution (as characterized by statistics such as the range, median, mean, variance, etc.) can be assumed to be representative of the distribution of the population from which they were taken.

6.3.1.3 Types of Sampling Designs

Before field data can be collected, spatial and temporal arrangement of samples (i.e., a sampling design) must be identified. The sampling design should be chosen so that the distribution of data that is collected best represents the actual, underlying population distribution. Excellent detailed discussions of sampling designs are presented by Green (1979), Krebs (1989), and Gilbert (1987). Additional sampling designs, specific for sampling of small mammals, are discussed in Call (1986), Jones et al. (1996), and EPA (1997). Sampling designs for plants are discussed in Hays et al. (1981) and EPA (1994a, 1994b, and 1997). In practice, sampling methods appropriate for the endpoint biota of interest (see Section 6.4) are first selected; then, the times and locations when and where samples are collected are determined by the sampling design. Three common and recommended sampling designs are random, stratified random, and systematic sampling.

- **Random Sampling.** The validity of most statistical methods requires that samples be collected randomly from within the population of interest. Random sampling uses the concept of uniform probabilities to choose representative sample locations. The objective of this sampling approach is to give each sampling unit in the population an equal probability of being included in the sample. Random sampling generally is employed when little information exists concerning the contamination or site. It is most effective when the number of available sampling locations is large enough to lend statistical validity to the random selection process.

- **Stratified Random Sampling.** Stratified random sampling involves the division of the sample population into strata based on knowledge of certain characteristics within the strata. Random samples are then taken from within these strata. This approach is used to increase the precision of the estimates made by sampling; it is most applicable when the

contaminant distribution is heterogeneous and clumped or associated with distinct habitats. Stratified random sampling is advantageous when contaminant concentration distributions within the strata are more homogeneous than they are between divisions.

- **Systematic Sampling.** Systematic sampling involves the collection of samples at predetermined, regular spatial or temporal intervals. It is the most often employed sampling scheme. However, care must be used to avoid bias. If, for example, there are periodic variations in the material to be sampled, the systematic plan may become phased with these variations (Krebs 1989). A systematic plan often results from approaches that are intended to be random. This is because investigators tend to subdivide a large sample area into increments prior to randomization (Green 1979). Studies performed comparing results from systematic and random sampling in ecological systems found no significant difference (Krebs 1989). Consequently, Krebs (1989) suggests that systematic sampling be employed for ecological applications, with the resulting data treated as if they were the results of random samples.

6.3.1.4 Sampling Bias

Sampling bias refers to the lack of representativeness of the sample with respect to the population of interest. This may result from the over-representation of sampling units that share a particular characteristic due to nonrandomness in the sampling design or execution. In this technical standard, the population of interest is the resident biota at the radionuclide contaminated site, not just those residing in the most contaminated portions of the site. Sampling only in areas of known contamination or hot spots, while potentially useful in determining maximum risks, will result in biased samples that overestimate the exposure to the representative individuals in the entire population at the site. Use of a good sampling design will reduce the likelihood of generating biased results.

Sampling schemes that will result in biased samples should be avoided. These include accessibility sampling (e.g., samples are collected at the most accessible locations), haphazard sampling (e.g., where and when samples are collected is determined by the whims of the investigator), or judgmental sampling (e.g., samples are collected based on the judgment of the investigator, such as in hot-spot sampling) (Krebs 1989, Gilbert 1987).

6.3.1.5 Background/Reference Areas

In addition to originating from anthropogenic sources, radionuclides are naturally occurring and ubiquitous in the environment. Quantities of naturally occurring radionuclides in the environment can vary dramatically, depending on the geology of an area (Eisler 1994). The BCGs and the biota dose limits for the protection of biota applied in this technical standard do not differentiate between radionuclides originating from anthropogenic and natural sources. It is important to recognize that it is the total weighted dose rate (i.e., taking into account all sources and types of radiation) to biota at the site that is to be evaluated. Therefore, background dose

rates should be included in the total weighted dose rate and should not be subtracted from the dose rates at the site (Jones 2001). However, radiation dose rates at local background areas can be used to ensure that the site-related dose rates represent an actual increase in exposure. This is particularly important if remedial activities are being considered, so that limited resources are not applied to an effort to remediate background levels of radionuclides.

The solution is to compare the data from the contaminated site to that collected from one to several uncontaminated background or reference sites. These sites should be selected such that they are as comparable as possible to the contaminated site. Background sites should possess similar geological, physical, chemical, and biological attributes, while being uninfluenced by the activities or releases from the contaminated site. The level above which contaminated media are determined to be greater than background should be determined through the DQO process (see Bilyard et al. 1997). Maximum site concentrations that are twice the mean background concentration have been commonly employed at hazardous waste sites to establish differences from background (Suter et al. 2000). Other comparison approaches are outlined in WADOE (1994), California EPA (1997), and Suter (1995). If the total weighted dose rate at the site is comparable to or less than that at the local background area, then it is unlikely that endemic biota populations are adversely affected from ionizing radiation at the site.

6.3.2 Statistical Considerations

Statistical concerns include underlying data distributions, summary statistics and confidence limits, and minimum sampling size.

6.3.2.1 Determination of Underlying Data Distribution

Many statistical procedures require knowledge of, or at least an assumption about, the type of distribution to which the data belong. Determining the distribution underlying the data is generally performed using various goodness-of-fit tests. Methods to perform these tests, which include the chi-square test, the Kolmogorov-Smirnov test, and others, are presented in many statistical texts (e.g., Gilbert 1987, Zar 1984, and Sokal and Rohlf 1981). Computer programs that fit distributions to sample data are also available. It should be noted that in most goodness-of-fit tests, a particular distribution is assumed (as the null hypothesis of the test) and the data are tested for the probability that they may have come from that distribution. Therefore, acceptance of a “good fit” means that the assumed distribution could not be rejected as a possible underlying distribution of the data and that statistical procedures based on that distribution can probably be used with minimal chance of increased error rates. Acceptance of a “good fit” does not mean that the data came only from the assumed distribution excluding all other possibilities.

Two of the more common types of distributions encountered for environmental data are the normal and lognormal distributions. A wide variety of tests are available to evaluate if the data

are normally distributed. Three highly recommended tests include the Shapiro Wilk W test, Filiben's test, and the Studentized Range test (Breckenridge and Crockett 1998).

While some environmental samples may be normally distributed, most are likely to be best fit to a lognormal distribution. An extensive discussion of the properties and applications of lognormal distributions is provided by Burmaster and Hull (1997). In a lognormal distribution, the log-transformed values display a normal distribution. Lognormal distributions may be readily identified by performing the Shapiro Wilk W test on log-transformed data. If the W statistic for the transformed data is not significant, then the data are lognormally distributed. Burmaster and Hull (1997) present a simplified approach for fitting a lognormal distribution to sample data based on probability plots.

An important component of determining distributions is the identification of outliers. Outliers are data values that are extreme upper or lower tails of the observed data. These values may or may not be representative of the overall data distribution of interest. Statistical methods for the identification of outlier values are presented in Gilbert (1987), Newman (1995), and WADOE (1992).

6.3.2.2 Calculation of Summary Statistics and Confidence Limits

Summary statistics describe the shape, spread, and location of the data (on the real number line). These values can then be used to determine the minimum number of samples required for statistical comparisons between samples from different populations. Because the estimation of summary statistics such as the mean and variance from sample data can be biased due to the shape of the underlying distribution, methods for estimating these statistics that control for bias have been developed for some specific distribution types. Selected formulas for calculation of summary statistics are briefly outlined below. Users should refer to the cited texts when applying these methods. Additional detail and formulas may be found in many standard statistical texts, including Zar (1983), Gilbert (1987), Green (1979), Krebs (1989), and WADOE (1992).

The mean and standard deviation for a normal distribution may be calculated using the following formulas (Zar 1983):

$$\bar{x} = \frac{\sum_{i=1}^{n} x_i}{n}$$

$$s = \sqrt{\frac{\sum x_i^2 - \frac{(\sum x_i)}{n}}{n-1}}$$

where $\bar{x}$ = arithmetic mean;

x_i = value for the i^{th} sample measurement;

n = sample size; and

s = standard deviation of the arithmetic mean.

Confidence intervals are limits representing a range within which there is a quantified degree of surety that the true population mean lies. Confidence intervals are calculated using the sample mean, standard deviation, and values from the students-t distribution that are selected based on the sample size (n) and the α level (the likelihood that the true mean falls outside of the confidence interval) that is acceptable. A standard formula for calculating the confidence interval of the sample mean (Dowdy and Wearden 1983) is:

$$\bar{x} \pm t_{(1-\alpha/2),\, n-1} \frac{s}{\sqrt{n}}$$

where s is the standard deviation of the arithmetic mean.

Values for the students-t distribution ($t_{(1-\alpha/2),\, -1}$) are readily obtained from tables presented in most statistical texts.

If the underlying distribution of the data is determined to be lognormal, four methods to estimate the mean ($\hat{\mu}$) and standard deviation ($\hat{\delta}^2$) are available (Gilbert 1987). One of the simplest of these methods is:

$$\hat{\mu} = \exp(\bar{y} + s_y^2/2)$$

$$\hat{\delta}^2 = \hat{\mu}^2 \left[\exp(s_y^2) - 1\right]$$

where $\bar{y}$ and s_y^2 are the arithmetic mean and variance for the transformed values $y_i = \ln x_i$. Confidence limits on the mean of the lognormal distribution may also be calculated:

$$UCL_{1-\alpha} = \exp\left(\bar{y} + 0.5 s_y^2 + \frac{s_y H_{1-\alpha}}{\sqrt{n-1}}\right)$$

$$LCL_{\alpha} = \exp(\bar{y} - 0.5s_y^2 - \frac{s_y H_{\alpha}}{\sqrt{n-1}})$$

with $UCL_{1-\alpha}$ and LCL_{α} representing the upper and lower confidence limits, respectively, and s_y being the square root of the variance of the transformed values (s_y^2). Values for H are obtained from a table in Land (1975); Gilbert (1987) presents a subset of these values.

6.3.2.3 Determination of Minimum Sampling Size

A key question in any sampling effort is how many samples need to be taken. The answer depends on the degree of precision in the estimate of the population mean that is desired and the acceptable probability of error. Determining the minimum sample size is a two-step process in which a preliminary sample is taken and the mean and variance from this sample are used to estimate the appropriate sample size. Methods for determining minimum sample sizes for data from a normal distribution are presented in Krebs (1989), Green (1979), and Gilbert (1987).

If the desired variance (V) of the mean (-) is specified, the number of samples required is calculated as follows (Gilbert 1987):

$$n = (s^2/V)(1-2/n_1)$$

where n = estimated number of samples required;

n_1 = number of preliminary samples taken; and

s^2 = variance from preliminary samples.

If the desired margin of error is specified, the number of samples is

$$n = (Z_{1-\alpha/2}\ \delta/d)^2$$

where $Z_{1-\alpha/2}$ is the standard normal deviate (readily obtained from Z-tables in most statistical texts), δ is the standard deviation of the population being sampled, and d is the relative error (expressed in the same units as the samples, x_i).

Gilbert (1987) also reports a method for determining sample size to estimate the median for a lognormal distribution:

$$n = \left((Z_{1-\alpha/2})^2\ s_y^2 \right) / \left([\ln(d-1)]^2 - (Z_{1-\alpha/2})^2\ s_y^2 \right) / N$$

where N is the number of potential sampling units in the population (generally assumed to be very large), and d is the prespecified tolerable relative error in the median.

If the estimated sample size that results cannot be supported within the budget constraints of the study or sufficient biota are not available, Gilbert (1987) suggests considering either a larger percent error or lower confidence (greater α). For example, if we are determining the minimum sample size for a median from a lognormal distribution, and if we assume d = 0.1 (10% relative error), α = 0.05, s^2_y = 2, and N being very large, the estimated sample size will be >800 (Gilbert 1987). However, if the acceptable relative error is increased to 50% (d = 0.5) with α = 0.05, the minimum sample size declines to a more manageable 47.

6.3.3 Uncontrollable Events

Uncontrollable events are an inherent component of any field sampling. Equipment breaks or fails to operate as expected, weather conditions impair sampling efficacy, or target species of interest either are not present at the site or do not respond to the selected sampling method. Consequences of these sorts of events are sample sizes smaller than the calculated minimum and data that may not be representative of the population at the site. The occurrence of uncontrollable events generally results in an increase in the uncertainty associated with the data and a weakening of the strength of conclusions that can be made from these data. Such events are not, however, insurmountable.

A simple approach to dealing with uncontrollable events is to expect their occurrence and develop contingency plans. These plans could include alternate endpoint species, sampling methods, or sampling designs if the first choice is not available or does not work. In some cases, however, no contingency plan will solve the problem. In these instances, it is likely that the investigator will have to accept less than ideal data and, therefore, greater uncertainty. In these situations, it is imperative that the investigator report detailed statistical summaries of the data along with explanations of the uncontrolled events and how they may have influenced the final results. These descriptions will allow risk managers to determine the quality and utility of the data.

6.4 Biota Sampling Methods

A wide variety of methods are available for collecting biota samples for contaminant analyses, with sampling methods generally being medium- or taxon-specific. Common collection methods for aquatic (e.g., fish, benthic invertebrates, reptiles, and amphibians) and terrestrial biota (e.g., plants, mammals, birds, and earthworms) are outlined below. Application of these methods within an appropriate sampling design will generate samples that can be used to define the radionuclide body burden experienced by representative individuals at the site.

6.4.1 Aquatic Biota

Aquatic biota include fish, benthic invertebrates, and amphibians and reptiles.

6.4.1.1 Fish

Sampling techniques for fish include electrofishing, nets, or traps. Selection of the appropriate method will depend on the species of interest and the type of aquatic system being sampled.

Most of these techniques may require a scientific collection license or similar permission. In electrofishing, an electric current is employed to stun fish, which are then captured with a net. Electrofishing is effective for both juveniles and adults of most species and for sampling structurally complex habitats. It also efficiently samples large areas in a relatively limited time while capturing a large percentage of individuals within an area. Numerous studies indicate that under proper conditions, electrofishing can be the most effective sampling technique (Jacobs and Swink 1982, Wiley and Tsai 1983, and Layher and Maughan 1984). Disadvantages include potential mortality (not a significant issue for sampling for contaminant analyses); low efficacy for benthic or deep water species, for very low- or high-conductivity water, and for turbid water; and potential hazards to users. Additional information on electrofishing can be found in Hartley (1980) and Reynolds (1983).

A wide variety of nets and traps are used to sample fish populations. Two basic types exist: nets that snag or entangle fish, and traps or net arrangements that provide a holding area into which fish are enticed. The most common entanglement nets are gill nets and trammel nets that use an open mesh through which fish attempt to swim. As the fish attempts to pass through, gill covers or fins become snagged on the fine filament netting. Gill nets are generally more effective in turbid water and areas without snags (Hubert 1983) and are effective for sampling deep areas not accessible by other techniques. Gill nets are also highly effective for a variety of larger fish sizes (depending on mesh size used) and for fast swimming or schooling species. Disadvantages include potential injury or mortality of snagged fish, the ability of any one gill net mesh size to sample only a limited size of fish, the capture of nontarget species at high rates (with the resulting increase in sampling time and total mortality), low success for fish species with low mobility (e.g., sunfish), and highly variable results. Further details are given in Hartley (1980), Hamley (1980), and Hubert (1983).

Stationary fish traps include fyke nets, hoop nets, trap nets, and pot gear (e.g., slat baskets and minnow traps). All of these devices work by allowing the movement of the fish to take them through a small opening into a larger holding area. Stationary traps are available in small (minnow traps) to large (fyke nets) sizes, allowing multiple species and life stages to be sampled. Because fish remain alive while in the trap, they do not need to be checked as frequently as entanglement nets. Stationary traps are effective for cover-seeking species (e.g., sunfish) or benthic species (e.g., catfish). Disadvantages of these traps are that they are not equally effective for all species and that catch rates are susceptible to changes in temperature

and turbidity. The larger fyke, trap, and hoop nets are most effective in reservoirs, ponds, lakes, and river backwaters. Pot gear and smaller hoop nets can be more effective in smaller streams or faster water. In both cases, traps can be combined with weirs or directional structures that channel fish into areas where the traps are deployed. Additional discussions can be found in Craig (1980) and Hubert (1983).

6.4.1.2 Benthic Invertebrates

Many techniques are suitable for collecting benthic macroinvertebrates for exposure evaluation, including grab and core samplers for standing waters, and kick sampling or Surber samplers for running water (Murkin et al. 1994).

Grab samplers such as the Ekman, Petersen, Ponar, and Smith-McIntyre samplers may also be used to collect organisms from deep-water habitats. These devices engulf a portion of substrate (and its associated organisms), which is then hauled to the surface for processing. Organisms are separated from the sample material by washing the substrate in a box screen. Grab samplers are generally easy to use and are suitable for a variety of water depths. Depth of sediment penetration may vary with sediment type and rocks or other obstructions may prevent complete closure, resulting in partial sample loss. Because grab samplers tend to produce large samples, the processing effort may be considerable (Murkin et al. 1994). Isom (1978) reviews several types of grab samplers, their specifications, the type of substrate each was designed for, and advantages and disadvantages associated with each type. Standard methods for the collection of benthic invertebrates using various types of grab samplers are also presented in ASTM (1997).

Core samplers may be employed in both shallow and deep water. They consist of a metal or plastic tube which is inserted into the substrate. When the tube is removed, samples of both the substrate and organisms are obtained (Murkin et al. 1994). The samples are then washed in a sieve and the organisms are removed from the remaining sample debris. Core samplers are inappropriate for loose or unconsolidated sediment, sand, or gravel (Murkin et al. 1994). Additional information on core sampling can be found in Smock et al. (1992) and Williams and Hynes (1973).

Kick sampling is a sample method used in running waters. A net is placed against the streambed, and the substrate upstream of the mouth of the net is agitated for a defined time period to suspend the organisms, which are then washed into the net by the current (Murkin et al. 1994). While this method is easy, the exact area sampled is undefined; therefore, it is unsuitable when quantitative samples are needed.

When quantitative samples from running water are needed, Surber samplers should be used. Surber samplers consist of a frame with an attached net. The frame is placed on the streambed, the substrate within the frame is disturbed and rocks and other debris are rubbed to dislodge invertebrates. Water current carries invertebrates into the sampling net (Murkin et al.

1994). Standard methods for the collection of benthic invertebrates using Surber and related types of samplers are presented in ASTM (1997).

6.4.1.3 Amphibians and Reptiles

Methods selected to sample reptiles and amphibians will vary depending on the type of habitat, time of year, weather conditions, and age of target species. Representative techniques for sampling reptiles and amphibians in aquatic and terrestrial habitats include opportunistic collection by hand, nets and traps, electrofishing, and seines. Additional discussion of methods may be found in Jones (1986) and Heyer et al. (1994).

Opportunistic collection consists of searching suitable habitats for species of interest. Once found, individuals are collected by hand, net, or other devices that may facilitate immobilizing individuals.

Numerous types of nets and traps are available for sampling herpetofauna. Traps are generally effective for alligators, turtles, snakes, and aquatic salamanders. Stebbins (1966), Conant (1975), and Shine (1986) discuss various aquatic trapping methods. Some traps may be set by one person. To prevent inadvertent mortality from trapping, traps should be checked at least daily (trap mortality is generally low if checked often). Aquatic traps should be set partially above water line to permit the captured organisms to breathe.

Although developed for sampling fish, electrofishing may also be very effective for aquatic salamanders and aquatic snakes (Jones 1986). This method occasionally yields turtles, sirens, and hellbenders. Electrofishing requires two or more people (a shocker and a netter) and is most effective in shallow water (streams, ponds, and shallow rivers). Deep-water habitats (lakes, reservoirs, and embayments) may be shocked from boats, but this approach is probably less effective for most herpetofauna than for fish. One disadvantage is that electroshocking may cause some mortality, especially in hot weather.

The use of small-mesh seines (7 mm or less) is moderately effective for sampling of aquatic salamanders, frogs, snakes, and turtles (Jones 1986). This method requires at least two people to operate the seine. Other personnel are beneficial for disturbing the substrate, blocking potential escape routes, and handling the catch.

6.4.2 Terrestrial Biota

Terrestrial biota taken for sampling include plants, mammals, birds, earthworms, and terrestrial arthropods.

6.4.2.1 Plants

Collecting plant material for residue analyses is a simple procedure. After plants of the appropriate species are identified in accordance with a suitable sampling design, they may be sampled either as whole organisms (roots plus aboveground parts) or as discrete parts (roots, foliage, seeds, fruit, etc.). Samples may be collected by stripping or breaking parts from the plant, by cutting plant parts with shears, or by digging up plants with a spade. Additional information on vegetation sampling for contaminant analysis, including sampling designs, may be found in EPA (1997), EPA (1996), DOE (1987), EPA (1994a), EPA (1994b), Hays et al. (1981), and Temple and Wills (1979).

6.4.2.2 Mammals

Numerous methods are available for collecting mammals. Suitable methods vary by species and habitat, with multiple methods often being suitable for the same species (Jones et al. 1996). For risk assessment purposes, small mammals, primarily within the orders Rodentia, and Insectivora, are the taxa most commonly collected. This is because they are often assessment endpoints themselves, important food items for predatory endpoints, and more likely to be present in sufficient numbers than larger mammals. Methods discussed will, therefore, focus on these taxa. Methods for collecting other mammalian taxa are discussed in Wilson et al. (1996), Schemnitz (1994), Kunz (1988a), and Nagorsen and Peterson (1980).

Small mammals are generally collected by one of three methods: snap traps, box traps, or pitfall traps. Snap traps are the familiar “mouse trap,” consisting of a spring-powered metal bale that is released when the animal contacts the baited trigger pan (Jones et al. 1996). These traps are lethal, with animals being killed by cervical dislocation. Nagorsen and Peterson (1980) report snap traps to be the most successful trapping method for small rodents and insectivores. However, because they are non-selective, snap traps may collect any animal that may be attracted to the bait. This may be a serious concern if threatened or endangered species are believed to be resident in the study area.

Box traps are the most effective method for capturing small mammals unharmed (Jones et al. 1996). The use of box traps allows the selection of species of interest and the release of non-target species. Box traps are typically metal or wooden boxes with openings at one or both ends and a baited trip pan. Animals are captured when they contact the trip pan, causing spring-loaded doors to close. Captured animals may be maintained in box traps for up to several hours if food and bedding are provided. The type and size of the trap, ambient conditions at the trapping site, and body size of animals to be trapped all influence trapping success (Jones et al. 1996). Because some animals are reluctant to enter box traps (shrews in particular), box traps are not as effective as snap traps (Nagorsen and Peterson 1980).

Pitfall traps consist of a container buried into the ground so that its rim is flush with the surface. Animals are captured when they fall into the container. Pitfall traps are among the most

effective traps for collecting shrews (Jones et al. 1996). Success rates for pitfall traps may be dramatically increased by employing drift fences. Drift fences are barriers of metal, plastic, fiberglass, or wood that direct small mammals into the pitfall trap. Pitfall traps may be employed as either live or killing traps. Killing pitfall traps are partially filled with water to drown animals. Live pitfall traps must be at least 40 cm deep to prevent small mammals from jumping out (Jones et al. 1996).

Both snap traps and box traps must be baited. Baits depend on the species sought. Generally, peanut butter and oats or other seeds are effective for most granivorous or omnivorous small mammals (Jones et al. 1996). Because small mammals simply fall into pitfall traps, these traps do not need to be baited (Nagorsen and Peterson 1980). Trapping success is generally enhanced if traps are set but locked open within the sampling area for several days prior to trapping. This allows the animals to acclimatize to the presence of the traps. Once traps are baited and set, both snap and box traps should be checked daily. Pitfall traps should be checked more frequently (twice daily) to prevent shrews from starving or consuming each other (Jones et al. 1996).

Trap placement to collect animals for contaminant analysis differs from a population survey. Sampling for contaminant analyses does not require a trapping array suitable to determine density. Sampling along transects is adequate. Jones et al. (1996) recommend that traps be placed along transects that are at least 150 m long with traps placed every 10 to 15 m. Regardless of spacing, traps should be placed at habitat features favored by or indicative of small mammals, e.g., logs, trees, runways, burrow entrances, dropping piles, etc. (Jones et al. 1996, Nagorsen and Peterson 1980). In addition, sampling must be appropriately distributed with respect to concomitant distributions and locations where media are sampled. Additional discussion of trap placement and sampling designs specific for sampling of small mammals are presented in Call (1986), Jones et al. (1996), and EPA (1997).

6.4.2.3 Birds

Methods for collecting birds include firearms, baited traps, cannon nets, mist nets, drive and drift traps, decoy and enticement lures, and nest traps (Schemnitz 1994). Methods employed depend upon the species to be sampled. Additional information concerning methods for capturing birds may be found in Schemnitz (1994), the *North American Bird Banding Manual* (USFWS and Canadian Wildlife Service 1977), *Guide to Waterfowl Banding* (Addy 1956), and *Bird Trapping and Bird Banding* (Bub 1990).

Firearms used to collect birds may include rifles, shotguns, or pellet guns. This method, while highly dependant on the skill of field personnel, may be used for all groups of birds. However, because samples may be extensively damaged during collection, projectiles or shot may interfere with contaminant analyses. Moreover, because of safety considerations, the use of firearms is not a recommended sampling method. In addition, the use of firearms precludes repeated sampling of the same individual.

Baited traps are most useful for gregarious, seed-eating birds. In their simplest form, a wire-mesh box is supported at one side by a stick over bait (generally seeds or grain). Once birds enter the box to feed on the seeds, the operator pulls a string attached to the support stick, the box falls, and the birds are entrapped. Other types of baited traps include funnel or ladder traps. These traps are designed with entrances through which birds can easily enter but not easily exit.

Cannon nets may be used for birds that are too wary to enter traps. This type of trap is frequently used for wild turkey and waterfowl and has been successfully used for sandhill cranes and bald eagles (Schemnitz 1994). Cannon nets consist of a large, light net that is carried over baited birds by mortars or rockets. In use, nets are laid out and baited for 1 to 2 weeks to allow the birds to become acclimated to the net and bait. Once birds make regular use of the bait, the trap may be deployed.

Mist netting is a method useful for some species that are not attracted to baits. A detailed review of the use and application of mist nets is provided by Keyes and Grue (1982). This method may be used for birds as large as ducks, hawks, or pheasant but is most applicable to passerines and other birds under ~200 g. Mist nets are constructed from fine black silk or nylon fibers; the nets are usually 0.9 to 2.1 m wide by 9.0 to 11.6 m long, attached to a cord frame with horizontal crossbraces called “shelfstrings” (Schemnitz 1994). The net is attached to poles at either end such that the shelfstrings are tight but the net is loose. The loose net hangs down below the shelf strings, forming pockets. When the net is properly deployed, birds (or bats) strike the net and become entangled in the net pocket. Mist nets may be employed passively or actively. In a passive deployment, nets are set across flight corridors and birds are caught as they fly by. For an active deployment, a group of nets is set and birds are driven toward the nets. Another effective approach is to use recorded calls of conspecifics or distress calls to attract birds to the net.

The following must be considered when using mist nets:

- Avoid windy conditions; wind increases the visibility of the net.
- Check nets frequently. Unintended mortality may result from stress if birds are left in the net for more than 1 hour.
- Do not use mist nets during rain. Birds may become soaked, and mortality may result from hypothermia.
- Special permits are required to use mist nets for migratory birds. These must be obtained from the U.S. Fish and Wildlife Service.

Drive and drift traps consist of nets or low wire mesh fencing erected at ground level. Birds are driven or herded into the fence, which then guides them into an enclosure. This method is most frequently used to capture waterfowl while they are molting and flightless. Drift traps have also been used successfully with upland gamebirds, rails, and shorebirds (Schemnitz 1994). Because many birds are reluctant to flush and fly when birds of prey are present, trapping success may be enhanced by playing recorded hawk calls.

Decoy and enticement lures are used most frequently for birds of prey. The most common trap of this type is the bal-chatri trap. This trap consists of a wire mesh cage, on top of which are attached numerous monofilament nooses. A small bird or rodent is placed in the trap as bait. When a hawk or owl attempts to attack the bait, it becomes entangled in the nooses.

Nest traps are useful to capture birds at the nest for reproductive studies. For ground-nesting birds, drop nets erected over the nest are sometimes effective. For cavity nesting birds, trip doors may be devised that can be closed once the adult enters the nest. Other types of nest traps are discussed by Schemnitz (1994).

6.4.2.4 Earthworms

The primary methods for collecting earthworms are hand sorting of soil, wet sieving, flotation, and the application of expellants. Hand sorting is regarded as the most accurate sampling method and is frequently used to evaluate the efficacy of other methods (Satchell 1970, Springett 1981). While accurate, hand sorting is very laborious and may underestimate the abundance of small individuals. Its efficiency depends on the density of the root mat, clay content of the soil, and weather conditions, if sorting is done in the field. Wet sieving consists of using a water jet and a sieve to separate earthworms from the soil (Satchell 1970). The efficiency of this method is not documented, and it may damage worms during washing. Flotation is another water-extraction method (Satchell 1970). Soil samples are placed in water; earthworms are collected as they float to the surface. This method may be used to extract egg capsules and adults of species too small to recover efficiently by hand sorting.

In contrast to methods that require excavation and processing of soil, expellants are applied *in situ* to collect earthworms. In practice, an expellant solution is applied to the soil surface within a sampling frame laid on the soil and allowed to percolate. Earthworms are then collected as they emerge from the soil. To enhance absorption of the expellant by the soil and to facilitate collection of earthworms as they emerge, vegetation at each sampling location should be clipped down to the soil surface. Expellants have traditionally consisted of formaldehyde or potassium permanganate solutions (Satchell 1970, Raw 1959). Drawbacks to these expellants include carcinogenicity, phytotoxicity, and toxicity to earthworms. In addition, these expellants also may introduce additional contamination and interfere with contaminant analysis. As an alternative, Gunn (1992) suggested the use of a mustard solution as an expellant. A commercially available prepared mustard emulsion was mixed with water at a rate of 15 mL/L and applied to soil within a 1-m^2 frame (to confine the expellant). Efficacy of mustard was found to

be superior to formaldehyde and equivalent to potassium permanganate (Gunn 1992). Recent work at Oak Ridge National Laboratory indicates that dry mustard (1 tsp/L) is also an effective expellant (B. Sample, pers. obs.). If worm samples are being collected for residue analysis, analyses should be performed on samples of the mustard expellant. These data will indicate if any contamination can be attributed to the extraction method.

6.4.2.5 Terrestrial Arthropods

Many methods are available to sample terrestrial arthropods. Because of the great diversity of life-history traits and habitats exploited by arthropods, no single method is efficient for capturing all taxa (Julliet 1963). Every sampling method has some associated biases and provides reliable population estimates for only a limited number of taxa (Kunz 1988b, Cooper and Whitmore 1990). Reviews of sampling methods for insects and other arthropods were given by Southwood (1978), Kunz (1988b), Cooper and Whitmore (1990), and Murkin et al. (1994). Descriptions of 12 commonly employed methods, arthropod groups for which they are appropriate, and advantages and disadvantages of each are summarized in Table 5.1.

6.4.3 Additional Sampling Considerations

Apart from methods and target species, a variety of concerns relate to sampling: quality assurance/quality control, sample handling, permitting, killing of sample animals, and human health and safety.

6.4.3.1 Quality Assurance/Quality Control

To ensure that all data collected are of the highest quality, verifiable, defensible, and suitable for regulatory decisions, a quality assurance and quality control (QA/QC) plan should be developed and all data collected and evaluated in accordance with this plan. General QA/QC requirements are outlined in DOE Order 414.1A (DOE 1999b). Specifications and guidelines for quality systems for environmental data collection and environmental technology programs are presented in ASQC (1994).

6.4.3.2 Sample Handling

The manner in which biological samples are handled and prepared will have a profound influence on the utility of the resulting data for risk assessment purposes. Sample-handling issues include how samples are pooled (i.e., compositing), sample washing, and denudation.

If the amount of sample material is too small for accurate radionuclide analysis (e.g., individual earthworms or other invertebrates or organs from vertebrates), samples from multiple individuals may be composited to produce a sample of sufficient size. Alternatively, samples may be composited over the contaminated site in an effort to reduce analytical costs. While the resulting composited sample represents the mean radionuclide concentration from all included

samples, it does not provide any information concerning the distribution of contaminant levels about the mean. Consequently, minimum and maximum values within the composite are unknown, a single high or low concentration may dominate the resulting composite value, and the composite value may over- or underestimate the concentrations present in the majority of samples. Compositing of samples must be appropriate for the intended use of the data. Compositing is generally suitable for biota samples to be used for dietary exposure modeling. This is because consumers are exposed to the average concentration in their diet. In contrast, if the samples are to represent internal body burdens for endpoint species (e.g., concentrations in target organs), compositing of samples will result in underestimates of body burdens. Because compositing samples loses information and may result in biased estimates, all compositing must be performed with caution.

In addition to containing contaminants within their tissue matrix, biota samples may have external contamination in the form of soil or dust adhering to their surfaces. Depending on the purpose of the analyses and the intended use of the analytical results, these external residues may or may not be washed off prior to analysis. If the contaminant of interest has a significant aerial deposition pathway or if soil ingestion is not being considered in the exposure model, then samples should not be washed. It should be recognized that these unwashed samples will be biased and will represent both bioaccumulation factors and external adhesion of contaminants.

Depuration refers to the voiding of the GI tract of sampled animals and is a consideration primarily for earthworms. Undepurated earthworms will generally have higher radionuclide concentrations than depurated earthworms from the same location. This is due to the large amount of soil retained in the GI tract of undepurated earthworms. Radionuclides in the soil in the GI tract will bias the body-burden estimates. If the model used to estimate exposure of animals that consume earthworms does not include a term for soil ingestion, this bias is not critical. However, if a soil ingestion term occurs in the model, the use of undepurated worms will result in some double counting of the amount of soil consumed and will overestimate exposure.

6.4.3.3 Permits

In most states, collecting biota is regulated by fish and game laws. National and international statutes may also apply, depending upon the species of interest. As a consequence, before any biota collection program is initiated, all appropriate permits must be obtained. Failure to obtain the needed permits may result in the rejection of the data or civil or criminal actions against the parties involved. For example, taking of migratory waterfowl requires a USFWS permit or a state hunting license (in season) and a Federal waterfowl stamp. Any activity involving threatened or endangered species requires a permit from the USFWS and/or the responsible state conservation agency. Permits for the collection of migratory birds must also be obtained from the USFWS. All states regulate the collection of fur-bearing species, such as muskrats, and game mammals, such as deer. In many states, collection of large numbers of

small mammals and lagomorphs requires special collection permits. Local USFWS offices and state fish and wildlife agencies should provide assistance on regulations and permits that are required.

6.4.3.4 Euthanasia

Although most capture techniques described are designed to capture animals alive, animals generally must be sacrificed prior to preparation for contaminant residue analysis. (An exception is blood, fur, or feather residue analysis, which may be performed on live animals.) It is essential that humane euthanasia methods be employed to sacrifice animals for analysis.

Gullet (1987) provides a detailed discussion of euthanasia methods for birds; these methods are also adaptable for mammals. Euthanasia may be achieved using either physical or chemical methods. Physical methods include cervical dislocation, decapitation, stunning and bleeding (exsanguination), and shooting. Chemical methods include lethal injection or inhalation of anesthetic or toxic gas. There are a number of questions to consider when choosing a technique (Gullet 1987):

- Is it appropriate for the size and type of animal?
- Does it present a risk to human health and safety?
- Is specialized equipment or training required?
- Is it time- and cost-effective?
- Will the technique offend the casual observer?
- Is it humane?

6.4.3.5 Health and Safety

Many wild animals either have or serve as vectors for parasites and pathogens that are communicable to humans. These include ticks, mites, rabies, hantavirus, and histoplasmosis. Depending on the taxa being collected, anyone involved in collection or preparation may be exposed. To ensure the health and safety of personnel, it is imperative that disease be considered as part of the sampling protocol and that all appropriate protective measures be taken. Kunz et al. (1996) present an extensive discussion of human health concerns associated with mammalian sampling.

Table 6.1 Comparison of Common Arthropod Sampling Techniques[a]

Method	Method Description	Arthropods Sampled	Advantages	Disadvantages
Sticky Trap	Adhesive material applied to a surface, usually cylindrical; arthropods adhere to surface upon contact.	Flying or otherwise active arthropods	Simple, inexpensive, versatile, and portable	Messy; temperature affects adhesive; adhesive likely to interfere with residue analysis; removal of samples from adhesive difficult; requires use of hazardous chemicals; quantification of area sampled difficult.
Malaise Trap	Fine mesh netting 'Tent' with baffles that guide arthropods into a collection jar that may or may not contain a killing agent/preservative	Primarily flying arthropods; crawling arthropods to a lesser degree	Versatile and simple to use; samples suitable for residue analysis (depends on use of preservative)	Expensive and bulky; catch strongly affected by trap placement; biased against Coleoptera; fewer catches per unit time; quantification of area sampled difficult
Shake-Cloth	Cloth or catch basin placed beneath plant; when plant is beaten or shaken, arthropods drop onto sheet and are collected	Foliage-dwelling arthropods	Simple, fast, and easy to perform; requires minimal equipment; samples suitable for residue analysis	Biased against active arthropods and individuals that adhere tightly to vegetation; quantification of area sampled difficult.
Sweep Net	Among most widely used methods; insect net is swept through vegetation in a predetermined manner	Foliage-dwelling arthropods	Simple, fast, and easy to perform; requires minimal equipment; samples suitable for residue analysis	Sample efficacy highly dependent upon vegetation structure and sampling personnel; biased against arthropods that adhere tightly to vegetation; quantification of area sampled difficult

Table 6.1 (Continued) Comparison of Common Arthropod Sampling Techniques[a]

Method	Method Description	Arthropods Sampled	Advantages	Disadvantages
Pitfall Trap	Cup or bucket (covered or uncovered) buried in ground up to rim; may or may not contain killing agent/preservative; may be employed with drift fences	Ground/litter arthropods	Simple and inexpensive; may estimate population density using mark-recapture; samples suitable for residue analysis (depends on use of preservative)	Biased against inactive arthropods; very active individuals may escape; captures affected by density and type of ground cover
Light Trap	Light source (generally ultraviolet) attached to vanes and a collecting bucket; may or may not employ killing agent/ preservative	Nocturnal, phototactic, predominantly flying arthropods	Portable; simple to use; collects many taxa, but Lepidoptera predominate; samples suitable for residue analysis (depends on use of preservative)	Catch affected by environmental conditions and trap placement; species-specific responses to light unknown; area sampled cannot be quantified
Pesticide Knockdown	Pyrethroid insecticide applied to vegetation by a fogger; arthropods killed are collected on drop sheets.	Foliage-dwelling arthropods	Simple, fast, and easy to perform; samples many arthropods with approximately equal probability	Foggers, pesticides expensive; affected by wind; may miss extremely active or sessile arthropods; pesticide may interfere with residue analysis; quantification of area sampled difficult
Emergence Trap	Conical or box shaped traps erected over water or soil to collect emerging adult arthropods	Arthropods emerging from soil or water	Inexpensive; simple to use; can estimate density of emerging arthropods; samples suitable for residue analysis	Large number may be needed to accurately estimate population

Table 6.1 (Continued) Comparison of Common Arthropod Sampling Techniques[a]

Method	Method Description	Arthropods Sampled	Advantages	Disadvantages
Pole Pruning	Foliage samples clipped; arthropods on foliage manually removed and counted	Foliage arthropods (especially Lepidoptera larvae)	Inexpensive and easy to perform; good for inactive and tightly attached arthropods; population density can be calculated; samples suitable for residue analysis	Biased against active arthropods; few arthropods per sample; sample processing is labor intensive
Portable Vacuum Samplers	Uses portable, generally backpack mounted vacuums to sample insects (Dietrick et al. 1959); widely used to sample agricultural pests	Foliage arthropods	Easy to use; population density can be calculated; samples suitable for residue analysis	Expensive (>$1000 each); best suited for low vegetation; application in forest is questionable; may not accurately sample all taxa
Stationary Suction	Consists of fan that pushes air through a metallic gauze filter to remove insects (Johnson and Taylor 1955)	Flying arthropods	Easy to use; population density can be calculated; samples suitable for residue analysis	Expensive; not very portable; use limited to areas with electrical power; difficult to sample large areas
Tree Bands	Burlap bands are attached to trees; takes advantage of tendency of some arthropods to move vertically on tree trunks	Vertically mobile arthropods	Simple and inexpensive; population density may be calculated; samples suitable for residue analysis	Installation is time-consuming; biased against most flying species

(a) Information obtained from Murkin et al. (1994), Cooper and Whitmore (1990), Kunz (1988b), and Southwood (1978), unless otherwise stated.

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7 Guidance on Radiation Weighting Factor for Alpha Particles

This section discusses how radiation doses due to alpha particles should be calculated in demonstrating compliance with the dose limits for aquatic and terrestrial biota to take into account the relative biological effectiveness of this radiation type. Guidance is presented on an assumed radiation weighting factor for alpha particles that should be used by DOE sites. In addition, information that could lead to a revision of the guidance is summarized.

7.1 Statement of Issue

The limits on radiation dose to aquatic and terrestrial biota adopted in this technical standard are expressed in terms of absorbed dose. These dose limits are based on studies of radiation effects in biota resulting from exposure to photons having a low linear energy transfer (LET); e.g., see NCRP (1991) and IAEA (1992). For exposures of biota to alpha particles, which are high-LET radiations, consideration must be given to whether a calculated absorbed dose should be increased by a factor representing the relative biological effectiveness (RBE) of this type of radiation.[1] Use of a radiation weighting factor for alpha particles would be based on the observation that, for the same absorbed dose, biological damage in tissue generally increases with increasing LET, and it would take into account that the purpose of the limits on absorbed dose is to limit the occurrence of deleterious biological effects in aquatic and terrestrial biota.

A radiation weighting factor for alpha particles is of concern only in estimating dose to biota resulting from internal exposure to alpha-emitting radionuclides. Alpha particles are assumed not to contribute to the absorbed dose from external exposure, due to their very short range in matter.

7.2 Previous Assumptions About Radiation Weighting Factor

In radiation protection of humans, an average quality factor ($\bar{Q}$) is used to represent observed RBEs for a given radiation type; RBEs generally depend on LET and the particular biological effect of concern.[2] For alpha particles of any energy, the usual assumption is $\bar{Q} = 20$ (ICRP 1991). This value is intended to represent RBEs for different stochastic biological effects of concern in humans (NCRP 1990).

Based on the assumption of $\bar{Q} = 20$ for alpha particles used in radiation protection of humans, the IAEA has included a radiation weighting factor of 20 for alpha particles in calculating a

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1 The RBE of any radiation is defined as the ratio of the absorbed dose of a reference radiation (normally gamma rays or X rays) required to produce a level of biological response to the absorbed dose of the radiation of concern required to produce the same level of biological response, all other conditions being kept constant.

2 The average quality factor now is called the radiation weighting factor (w_R) by the International Commission on Radiological Protection (ICRP 1991).

weighted absorbed dose to aquatic and terrestrial biota (IAEA 1992). This value also has been used by other investigators (Blaylock et al., 1993; Jones 2000).

Other investigators have not used a radiation weighting factor for alpha particles in calculating absorbed dose to biota. This choice has been justified in one of two ways. Some investigators argued that a radiation weighting factor of 20, based on the value $\bar{Q}$ = 20 used in radiation protection of humans, may not be appropriate for biota (Baker and Soldat 1992; Amiro 1997), because the radiation effects of concern are not the same in the two cases. The NCRP argued that the use of conservative models to estimate concentrations of alpha-emitting radionuclides in the tissues of aquatic biota compensates for the neglect of a radiation weighting factor for alpha particles (NCRP 1991).

7.3 Radiation Effects of Concern in Biota

Radiation protection of biota usually is concerned with ensuring adequate protection of whole species, rather than individual members of species. For exposures of aquatic and terrestrial biota, the critical biological endpoint appears to be impairment of reproductive capability (NCRP 1991; IAEA 1992). Other biological endpoints affecting the viability of species (e.g., substantial morbidity) occur only at doses higher than those that significantly affect reproductive capability.

Furthermore, the critical biological endpoint of concern in radiation exposures of biota appears to be deterministic in nature,[3] rather than stochastic.[4] That is, effects of radiation exposures on populations of species are not observed below doses and dose rates that are much higher than natural background, and the effects occur soon after exposure. The dose limits for biota are intended to prevent the critical deterministic biological effect in sensitive species.

7.4 Data on Deterministic RBEs for High-LET Radiations

Since deterministic effects appear to be the most important in radiation protection of biota, stochastic RBEs for alpha particles that provide the basis for the average quality factor of 20 used in radiation protection of humans may not be relevant. Data on RBEs for deterministic radiation effects have been reviewed and evaluated by the ICRP (1989). The RBEs at low doses and dose rates for different types of high-LET radiation estimated by the ICRP may be summarized as follows.

- The RBE for deterministic effects induced by 1-5 MeV neutrons varies from 4 to 12, and the average value based on the results of 19 determinations is about 7.

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3 Deterministic effects are those for which the severity is a function of dose, and for which a threshold usually exists.

4 Stochastic effects are those for which the probability of occurrence is a function of dose, without threshold, but the severity of the effect is independent of dose.

- The RBE for deterministic effects induced by 5-50 MeV neutrons varies from 1 to 10, and the average value based on the results of 31 determinations is about 5.

- The RBE for deterministic effects induced by heavy ions (C, Ne, and Ar) varies from 1 to 8, and the average value based on the results of 19 determinations is about 4.

- The data on deterministic effects induced by alpha particles are much less extensive than the data for the other high-LET radiations, but two separate determinations yielded estimated RBEs of about 7 and 10.

The average RBE for deterministic effects, based on all determinations, is about 5.

The information summarized above leads to the conclusion that, for high-LET radiations, the radiation weighting factor for deterministic effects is substantially less than the corresponding average quality factor used in radiation protection of humans. Based on this information, the radiation weighting factor for deterministic effects induced by alpha particles appears to lie in the range of about 5-10.

7.5 Recommendations on Radiation Weighting Factor for Alpha Particles

Use of a radiation weighting factor of 5 for alpha particles in calculating a weighted absorbed dose in biota has been suggested by the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR 1996). The basis for this value was not discussed, except it assumes that deterministic effects are the most important in exposures of biota. The suggested radiation weighting factor for alpha particles presumably was based on the evaluation of RBEs for deterministic effects by the ICRP (1989), as summarized in the previous section.

In radiation protection of humans, the ICRP has continued to use a radiation weighting factor of 20 for alpha particles in predicting deterministic effects, even though the ICRP also acknowledges, based on its review of RBEs for deterministic effects, that this approach likely results in overestimates of the contribution to the deterministic risk from alpha particles (ICRP 1991). The ICRP's conservative approach to assessing deterministic effects for high-LET radiations is of no consequence in radiation protection of humans, because allowable exposures of workers and members of the public generally are controlled by limits on effective dose that are intended to limit the risk of stochastic effects, rather than deterministic limits on equivalent dose in any organ or tissue (ICRP 1991). The ICRP has not considered the question of an appropriate radiation weighting factor for high-LET radiations in radiation protection of biota.

7.6 Guidance on Radiation Weighting Factor for Alpha Particles

The guidance of DOE's Office of Environmental Policy and Guidance, Air, Water, and Radiation Division (EH-412) on a radiation weighting factor for alpha particles to be used in dose assessments for biota is the following:

All DOE sites shall use a radiation weighting factor of 20 for alpha particles in calculating a weighted absorbed dose to aquatic and terrestrial biota for the purpose of demonstrating protection with the applicable dose limits applied in this technical standard.

The dose assessment methodology described in this technical standard uses this radiation weighting factor in calculating dose from internal exposure to alpha-emitting radionuclides.

The guidance on a radiation weighing factor for alpha particles is based mainly on two considerations. First, based on the review of deterministic RBEs for high-LET radiations by the ICRP (1989), a radiation weighting factor of 20 for alpha particles is likely to be conservative, and a conservative assumption is considered appropriate for use in a screening methodology for evaluating compliance with the limits on absorbed dose to aquatic and terrestrial biota.

Second, although there is considerable evidence that the radiation weighting factor for alpha particles that could be used in radiation protection of biota is less than the value of 20 used in radiation protection of humans, authoritative organizations, such as the ICRP and NCRP, and regulatory authorities, such as the U.S. Environmental Protection Agency, have not developed a recommendation on the most appropriate value based on a careful review of available information. Absent such a recommendation, it is prudent to assume the radiation weighting factor for alpha particles used in radiation protection of humans.

The guidance on a radiation weighting factor for alpha particles to be used in radiation protection of aquatic and terrestrial biota at DOE sites is subject to change as authoritative organizations and regulatory authorities develop a consensus on an appropriate value for deterministic radiation effects.

8 Guidance on the Applicability of the Graded Approach for Evaluating Dose to Individual Organisms

8.1 Considerations on the Meaning of "Individual" Organism

At the outset, the concept of an “individual” needs to be understood. A system for protection of an “individual,” such as the system for radiation protection of humans, is never intended to apply to each and every specific, identifiable individual (e.g., a named member of the public). Rather, the concept of an “individual” refers to a *reference* organism that is intended to represent typical characteristics within a particular population group. The main reason for use of the concept of a reference individual is that the characteristics of specific, identifiable individuals (e.g., individual radiosensitivities, the behavior of radionuclides in the body of an individual) can never be known. In radiation protection of humans, for example, compliance with the dose limits for individual workers or members of the public is demonstrated by calculating doses to a hypothetical construct called Reference Man. The hope is that by limiting dose (and risk) to a reference individual, no real individual will experience unacceptable doses (and risks), but it cannot be ensured that unacceptable outcomes will never happen to any real individual.

8.2 Applicability of Methods and Models Contained in the DOE Graded Approach to Evaluations of Individual Organisms

The graded approach for evaluating radiation doses to aquatic and terrestrial biota developed by DOE, taken as a whole, can be viewed as consisting of two components:

- A set of models for calculating dose to biota per unit concentration of radionuclides in environmental media (water, sediment, and soil); and

- A set of dose criteria or limits for aquatic animals, terrestrial plants, and terrestrial animals, which represent dose levels of concern based on current information on dose-response relationships in a variety of organisms.

By combining calculated doses per unit concentration of radionuclides in environmental media with the dose criteria, BCGs are obtained. The BCGs then are compared with measured concentrations to assess compliance with the dose limits. The models for calculating dose per unit concentration of radionuclides in environmental media clearly apply to individual organisms. Thus, these models are directly applicable to individual organisms (e.g., for application to individual members of threatened and endangered species). However, the question of whether the dose criteria can be applied to protection of individual members of a species, in contrast to protection of populations of species, requires further consideration.

8.3 Applicability of Biota Dose Limits to Protection of Individual Organisms

The dose criteria used by DOE are based on studies of dose-response relationships in *populations* of aquatic animals, terrestrial plants, and terrestrial animals. The particular biological endpoints for which dose-response relationships have been obtained include early mortality and impairment of reproductive capability, the latter including effects on reproductive tissues and the embryo/fetus or seeds. Since reproductive effects in a population generally occur at lower doses than early mortality, the dose-response relationships for reproductive effects were used to derive the dose criteria. Thus, at first sight, it would appear that the dose criteria should be applied only when protection of populations of organisms is of concern, but they may not be appropriate when protection of individual members of a species is of concern.

However, the following points about the dose criteria should be noted. First, even if protection of populations is the primary concern, effects on populations of organisms can be inferred only by considering effects in individual organisms comprising a given population. That is, in determining effects on populations, one would essentially need to count the number of impaired organisms in an irradiated population compared with the number of similarly impaired organisms in an unexposed population. Second, the dose criteria are based on the lowest dose at which any reproductive effects are observed in any species of aquatic animals, terrestrial plants, or terrestrial animals. Thus, if it is assumed that the species studied include those which are among the more radiosensitive, the dose criteria intended to ensure that there would be no significant effects at a population level should ensure that there would be no *observable* effects on individual members of a species, bearing in mind that there is always a background of similar effects from all causes, which limits the ability to observe radiation-induced effects.

8.4 Use of the DOE Graded Approach for Evaluating Dose to Individual Organisms: Application Considerations

In examining the models and methods contained in the graded approach, and the basis for the biota dose limits, one key difference between applying them to protection of individuals or protection of populations is in regard to the extent to which calculated doses could be averaged over the spatial extent of contamination and over time. In protecting populations, considerable averaging over space and time could be allowed and still ensure adequate protection. In protecting individuals, however, it could be more appropriate to allow little or no averaging over space and time. Thus, in protecting individuals, use of the maximum concentrations of radionuclides in the environment at any location and at any time could be more appropriate.

Use of safety factors, appropriate default parameter values, maximum radionuclide concentrations in environmental media, and 100 percent organism residence time and exposure may support the application of the graded approach for evaluating doses to individuals.

8.5 Consideration of Deterministic vs. Stochastic Effects

There is one additional caution that should be heeded in applying the dose limits to individual organisms, such as those for a threatened and endangered species. The dose criteria were derived from observed dose-response relationships for effects that generally are assumed to be deterministic in character, meaning that there should be no effects at doses below some threshold. However, there also is a possibility that stochastic radiation effects could be important in exposures of biota.

Information on stochastic effects in biota was considered in the 1996 UNSCEAR report on *Effects of Radiation on the Environment*. The effects studied were at the cellular level, and include scorable cytogenetic effects (effects on DNA). The UNSCEAR report concluded that as long as the dose was kept below the dose criteria derived from dose-response relationships for reproductive effects, stochastic effects should not be significant at a population level.

However, the discussion in the UNSCEAR report leaves open the question of whether stochastic effects could cause harm in an individual organism (e.g., induction of a tumor that would result in premature death of an individual compared with the normal life span). There are two difficulties with interpreting the available data. First, the data on scorable cytogenetic effects appear to be considerably limited compared with the data on early mortality and reproductive effects. Second, although the available data in mammals and arthropods appear to indicate that scorable cytogenetic effects can be observed at dose rates roughly 100 times lower than the lowest dose rates causing early mortality and roughly 10 times lower than the lowest dose rates causing reproductive effects, it is difficult to interpret the significance of these effects in regard to harm to an individual organism (e.g., induction of tumors). For example, effects on DNA in humans who live in areas of unusually high natural background are easily observed, but increased incidence of cancers has not been observed in these populations.

Therefore, it is difficult to know how to apply the available information on scorable cytogenetic effects in a system for protection of individuals or populations. The best that can be said is that observations of these effects provide one more piece of information that could be used in evaluating the consequences of radiation exposures of biota and in deciding how to respond to those consequences.

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A Graded Approach for Evaluating Radiation Doses to Aquatic and Terrestrial Biota

MODULE 3

METHODS DERIVATION

MODULE 3: METHODS DERIVATION

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1 Introduction and Basis for the Approach

The Department of Energy (DOE) currently has in place a radiation dose limit of 1 rad/d (10 mGy/d) for the protection of aquatic organisms (DOE Order 5400.5), and has proposed dose limits for both aquatic and terrestrial organisms. These limits are: 1 rad/d (10 mGy/d) for aquatic animals; 1 rad/d (10 mGy/d) for terrestrial plants; and 0.1 rad/d (1 mGy/d) for terrestrial animals. Because the biota protection limits are dose-based, a calculational method is needed to demonstrate compliance. In theory, derived radionuclide concentration limits for environmental media (e.g., Biota Concentration Guides, BCGs, for water, sediment, or soil) provide a relatively straightforward and simple means to do so. However, because of the inherent complexity of environmental systems, and the vast array of biota that can potentially be exposed to any radionuclide contamination level, it was decided that a graded approach to evaluating compliance would be the most appropriate.

The first step in evaluating compliance would be to compare measured environmental concentrations with very conservative (i.e., very restrictive or protective) BCGs in a general screening process. To be useful in general screening, the concentration limits (BCGs) must be set so that real biota exposed to such concentrations are not expected to ever exceed the biota Dose Rate Guidelines. Since the screening limits would be chosen to protect “all biota, everywhere” they would, by their nature be restrictive, and in many circumstances conservative with regards to specific environments. Consequently, the graded approach for evaluating compliance had to allow site users to examine and revise, if appropriate, the screening limits to more realistically reflect the conditions at their site. This approach parallels methods currently used to protect human health from residual levels of radionuclides in the environment (e.g., site-specific conditions can be considered in deriving residual radionuclide concentration levels).

This Module provides detailed descriptions of the dose models, equations, and default parameters used in the graded approach for evaluating doses to biota. Topics presented include: (1) selection of pathways, media, organism types, and target radionuclides; (2) derivation and selection of lumped parameters; (3) derivation of internal and external dose conversion factors; (4) equations and models for calculating dose to biota and deriving BCGs; and (5) default parameters and their sources.

1.1 Pathways, Media Types, and Organism Types Addressed

The Biota Dose Assessment Committee (BDAC) had to consider several factors in developing the general screening methodology. The method had to be simple, defensible, and user-friendly. It also had to have broad applicability - from aquatic animals through terrestrial species. It also had to address radiation dose in small organisms (e.g., mice) and large carnivores (e.g., cougars). The method had to provide a logical and consistent departure point should additional in-depth evaluation of dose be required. Should additional analysis be required, the method had to utilize existing data - either from the technical literature or from

site-specific monitoring - whenever possible. Lastly, the method had to be useful in evaluating the potential impacts of combined media: water, sediment, and soil.

The BDAC's choice of organisms for the methodology evolved from consideration of the existing and proposed radiation dose limits for biota. Biota dose limits had been set for aquatic animals, and were being considered for terrestrial plants and animals. Accordingly, the screening methodology had to accommodate these three general categories. A fourth, riparian animal, was added after recognizing that the riparian pathways of exposure combined aspects of both the terrestrial and aquatic systems.

The pathways of exposure evaluated for each of the four organism types were developed based on consideration of the likelihood of dose occurring through a specific route, or "pathway." Based on the potential pathways of exposure, BCGs were derived for surface water, sediment, and soil. Calculated using conservative assumptions, the BCGs are intended to preclude the relevant biota from being exposed to radiation levels in excess of the relevant existing or recommended biota dose limits.

1.2 Selection of Target Radionuclides

Biota Concentration Guides (BCGs) that are considered to be conservatively protective of non-human biota were derived for twenty-three radionuclides. These BCGs are provided for radionuclide concentrations in water, sediment, and soil. They have been calculated based on limiting the potential radiological dose rate to the most sensitive receptors: aquatic, terrestrial, and riparian animals, and terrestrial plants. These radionuclides (see Module 1, Tables 6.1-6.4) were selected because they are relatively common constituents in past radionuclide releases to the environment from DOE facilities. This list is not meant to imply particular concern for biotic impact from these twenty-three specific radionuclides. Rather, it is a starting point for application of the methodology. The list was developed in consultation with BDAC members, and health physics and radioecological staff at several Federal facilities. It represents a general consensus as to the most prevalent radionuclides in environmental releases.

1.3 Overview of the Technical Approach for Deriving the BCGs

The derivation of BCGs used to demonstrate compliance with the biota dose limits is based on the fact that biota dose is a function of the contaminant concentration in the environment, and is the sum of internal and external contributions. It is possible, given a unit concentration (i.e., 1 Bq kg^{-1}) of a contaminant in a single media (e.g., soil) to estimate the potential dose rate to a receptor from both internal and external exposures (admittedly, several assumptions must be made to do so, and these are described in the following sections). Once the dose rate has been calculated, it can be ratioed to the dose rate limit, and used to back-calculate a concentration of the contaminant in the media that could generate a dose rate at the specified biota dose limit. If multiple contaminated media are present then the dose evaluation can be performed for each, and the results individually ratioed to the standard. This "sum of fractions"

approach is commonly used in evaluating compliance for humans exposed to radionuclides discharged to air, soil and water.

Once the target radionuclides had been selected, external dose coefficients (also called dose conversion factors, DCFs) were developed which relate environmental concentrations of the contaminants in water, sediment and soil to projected organism dose rate. Internal dose coefficients (DCFs) were also developed to estimate dose rate from internally deposited radionuclides.

General Dose Equation and Approach Used to Derive BCGs

$$\text{Limiting Concentration} = \frac{\text{Dose Rate Limit}}{(\text{Internal Dose Rate}) + (\text{External Dose Rate}_{\text{soil/sed.}}) + (\text{External Dose Rate}_{\text{water}})}$$

The limiting concentration in an environmental medium was calculated by first setting a target total dose (e.g., 1 rad/d for aquatic organisms and terrestrial plants, or 0.1 rad/d for riparian and terrestrial animals) and then back-calculating to the medium concentration (i.e., the BCG) necessary to produce the applicable dose from radionuclides in the organism (internal dose), plus the external dose components from radionuclides in the environment (external dose). The denominator of the generic equation may be broken down into the base components of internal and external dose. Internal doses originate from radionuclides inside the organism's body. The internal dose is calculated as the product of the internal radionuclide concentration and the internal dose conversion factor. External doses originate from radionuclides external to the organism and are calculated as the product of the radionuclide concentration in the environmental medium in which the organism resides and an appropriate dose conversion factor.

1.4 Selection of the Most Limiting BCGs for Use in General Screening

As discussed, BCGs were derived for a matrix of radionuclides and media types for each of four organism types. That is, BCGs were derived for twenty-three radionuclides within water, sediment, and soil media for aquatic animal, riparian animal, terrestrial plant, and terrestrial animal organism types. The resulting BCGs from this matrix of radionuclides, media types, and organism types were then reviewed to determine the most limiting (i.e., most conservative or protective) values that could be summarized in two tables for the general screening phase of the graded approach: one for aquatic systems and one for terrestrial systems. The logic flow for selecting the BCG values for use in the general screening phase of the graded approach is illustrated in Figure 1.1.

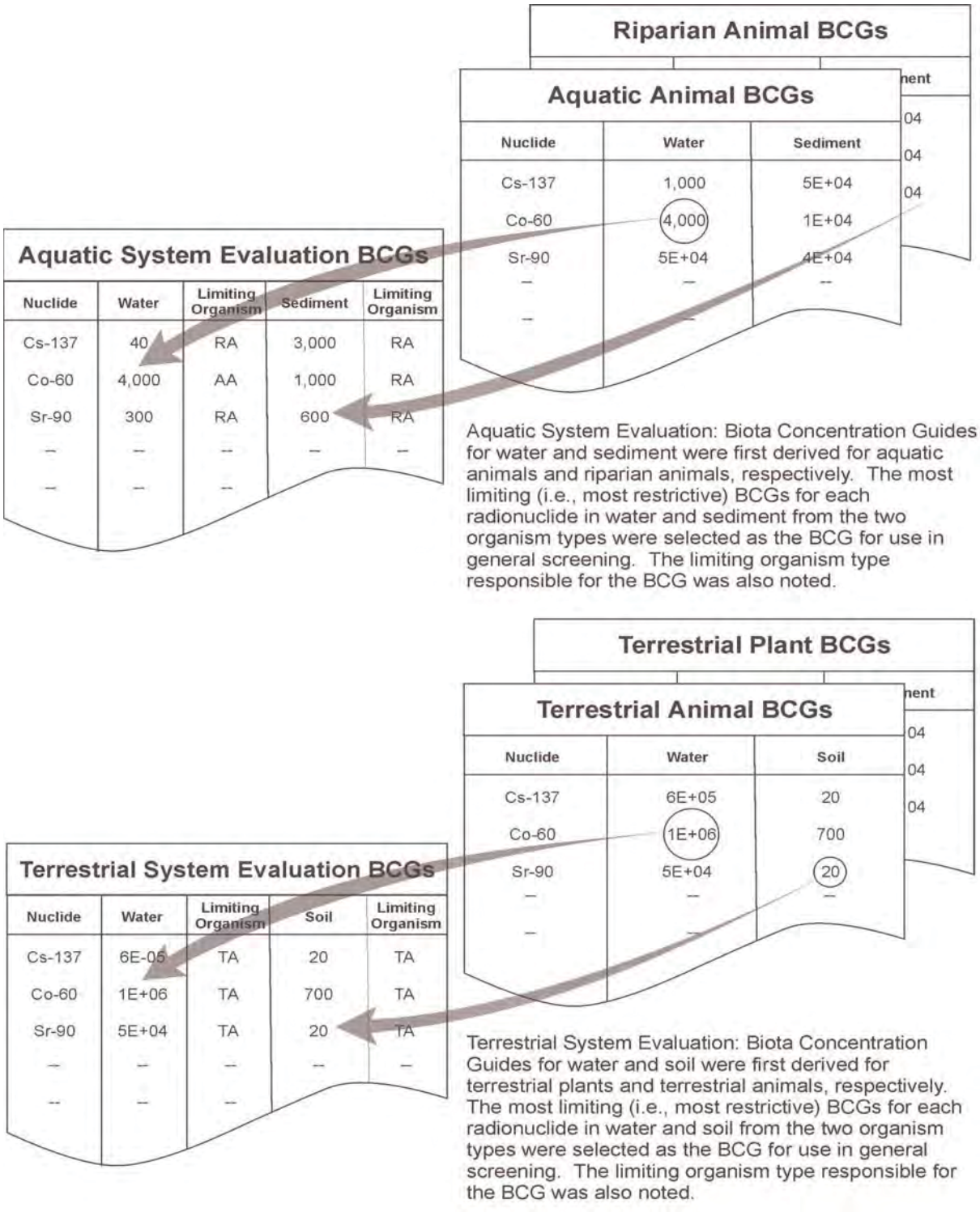

Figure 1.1 Selection of Biota Concentration Guides (BCGs) for Use in Aquatic and Terrestrial System Evaluations.

2 Dose Coefficients

2.1 External Dose Coefficients

This section describes a simple approach to calculating external dose coefficients for aquatic and terrestrial biota that can be used for purposes of screening in demonstrating compliance with specified limits on absorbed dose rates to biota, and it presents tables of screening-level external dose coefficients for exposure of aquatic and terrestrial biota to selected radionuclides in the environmental media of concern.

2.1.1 Introduction

External dose coefficients (also called external dose rate conversion factors or external dose conversion factors) give dose rates from external exposure per unit concentration of radionuclides in environmental media. For external exposure to radionuclides in the environment, only penetrating radiations (photons and electrons) are of concern, and non-penetrating radiations (e.g., alpha particles) need not be considered. The environmental (source) media of concern are contaminated water and sediments for exposure of aquatic animals and contaminated soil and water for exposure of terrestrial biota. Contaminated air (i.e., the active air pathway) is not an important source medium for terrestrial biota, because the limits on allowable concentrations of radionuclides in air based on requirements for protection of on-site workers and members of the public would result in absorbed dose rates to terrestrial biota that are far less than specified limits (see Module 2, Section 2.2).

The essence of screening-level external dose coefficients for aquatic and terrestrial biota is that they clearly must provide conservative overestimates of absorbed dose rates from external exposure to given concentrations of radionuclides in the environment. Screening-level dose coefficients thus provide a means of demonstrating compliance with specified limits on absorbed dose rate for aquatic and terrestrial biota that can be used at any DOE site, without the need for a detailed exposure pathway analysis based on site-specific considerations of the important species at risk and the important exposure pathways.

2.1.2 Approach to Calculating External Dose Coefficients

The approach to calculating external dose coefficients for aquatic and terrestrial biota for use in general screening should be simple and transparent, so that it can be easily implemented and understood. Furthermore, as indicated above, the approach must clearly result in conservative estimates of external dose rates to aquatic and terrestrial biota for given concentrations of radionuclides in the environment. The approach to calculating screening-level external dose coefficients for aquatic and terrestrial biota is based on the following assumptions:

- First, the source medium (water, sediment, or soil) is assumed to be infinite in extent and to contain uniform concentrations of radionuclides. This assumption results in reasonably realistic estimates of dose rates for radionuclides which are dispersed in the

source medium, because the range of electrons emitted in radioactive decay is no more than a few cm and the mean-free-path of emitted photons is no more than a few tens of centimeters (Shleien et al. 1998).

- Second, the exposed organism is assumed to be very small (less than the mean free path of the electron emitted in decay). This assumption results in overestimates of external dose rates for any finite-sized organism, because the attenuation of photons and electrons in transport through an organism is ignored. In addition, the assumption of a very small organism combined with the assumption of an infinitely large and uniformly contaminated source medium leads to a particularly simple approach to calculating screening-level external dose coefficients developed in the following section. Specifically, because all of the energy emitted by radionuclides in a uniformly contaminated and infinite source medium is absorbed uniformly throughout the medium, the dose rate in the organism is essentially the same as the dose rate in the medium itself, and the absorbed dose rate can be calculated directly from the energy of photons and electrons emitted per disintegration of the radionuclides in the medium.

- Third, because the organism is assumed to be very small, the energies of all photons and electrons emitted by radionuclides are taken into account in calculating the screening-level external dose coefficients. This approach is particularly conservative for electrons when the irradiated tissues of concern lie below the body surface of an organism and lower-energy electrons could not penetrate to the location of these tissues. Taking into account the energies of all photons and electrons in radioactive decay is tantamount to assuming that the radiosensitive tissues of concern (i.e., the reproductive tissues) lie on the surface of a very small organism.

Based on the foregoing discussions, the approach to calculating screening-level external dose coefficients is simple, because the dose coefficients are calculated based only on the known energies and intensities of photons and electrons emitted in the decay of radionuclides, and it is evidently conservative in providing overestimates of external dose rates to the reproductive tissues of finite-sized organisms. The calculations of screening-level external dose coefficients for aquatic and terrestrial biota based on this approach are described in the following sections.

2.1.2.1 Screening-Level External Dose Coefficients for Aquatic Animals

Screening-level external dose coefficients for exposure of aquatic animals to radionuclides in sediments and water are calculated based on the assumptions described in the previous section and the additional conservative assumption that the organism is located 100 percent of the time at the water-sediment interface. Thus, it is assumed that the organism was exposed at the boundary of two semi-infinite and uniformly contaminated media. The assumption of exposure at the boundary of a semi-infinite medium results in an absorbed dose rate in the organism that is one-half of the dose rate in an infinite source volume. The calculation of the screening-level external dose coefficients for aquatic animals then proceeds as follows.

The total energies of all photons and electrons emitted in the decay of radionuclides are assumed to be given in units of MeV per disintegration. For exposure to contaminated sediments, the desired units for the external dose coefficients are rad/d per pCi/g. The emitted energy in MeV per disintegration (i.e., per Bq-s) is expressed in terms of the desired units for the external dose coefficients by multiplication of the known factors relating energy in MeV to ergs, absorbed energy in ergs/g to rads, time in seconds to days, and activity in Bq to pCi:

$$\left(1\ \frac{\text{MeV}}{\text{Bq}\cdot\text{s}}\right)\left(1.6\text{E}\cdot 06\ \frac{\text{ergs}}{\text{MeV}}\right)\left(0.01\ \frac{\text{g}\cdot\text{rad}}{\text{erg}}\right)\left(8.64\text{E}\cdot 04\ \frac{\text{s}}{\text{d}}\right)\left(0.037\frac{\text{Bq}}{\text{pCi}}\right)\cdot\cdot 5.12\text{E}\cdot 05\ \frac{\text{rad/d}}{\text{pCi/g}}$$

If SI units are used for absorbed dose (Gy), activity (Bq), and mass (kg), and the unit of time is taken to be the year, the factor for converting emitted energy to the external dose coefficient is obtained by a similar calculation as:

$$\left(1\ \frac{\text{MeV}}{\text{Bq}\cdot\text{s}}\right)\cdot\cdot 5.04\text{E}\cdot 06\ \frac{\text{Gy/y}}{\text{Bq/kg}}$$

As noted above, the external dose coefficient at the sediment-water interface is one-half of the value for exposure in an infinite medium. Therefore, given the total energies (E) of photons and electrons in MeV per disintegration of a radionuclide, the external dose coefficient (d_{ext}) for exposure to contaminated sediments is given by:

$$(d_{ext})_{\text{sediments}}\left(\frac{\text{rad/d}}{\text{pCi/g}}\right)\cdot\cdot(2.56\text{E}\cdot 05)E_{\text{photons}\cdot\text{electrons}}\left(\frac{\text{MeV}}{\text{dis}}\right)$$

If the desired units for the external dose coefficients are Gy/y per Bq/kg, the factor by which the decay energy is multiplied is 2.52E-06.

For exposure to contaminated water, the desired units for the external dose coefficients are rad/d per pCi/L. If the density of water is assumed to be 1 g/cm^3, the external dose coefficient for exposure to contaminated water at the sediment-water interface is obtained from a calculation similar to that for contaminated sediments given above as:

$$(d_{ext})_{\text{water}}\left(\frac{\text{rad/d}}{\text{pCi/L}}\right)\cdot\cdot(2.56\text{E}\cdot 08)E_{\text{photons}\cdot\text{electrons}}\left(\frac{\text{MeV}}{\text{dis}}\right)$$

Similarly, if the desired units for the external dose coefficients are Gy/y per Bq/m^3, the factor by which the decay energy is multiplied is 2.52E-09.

The screening-level external dose coefficients for exposure of aquatic animals to selected radionuclides in contaminated sediments calculated as described above are given in Table 2.1,

and the values for exposure to contaminated water are given in Table 2.2. The energies of all photons and electrons per disintegration of the radionuclides are obtained from the compilation by Kocher (1980), which summarizes the data contained in a handbook of decay data tables (Kocher 1981). For most radionuclides, the decay data compiled by Kocher are in good agreement with the data compiled by the ICRP (1983).

2.1.2.2 Screening-Level External Dose Coefficients for Terrestrial Biota

Screening-level external dose coefficients for exposure of terrestrial biota to radionuclides in soil are calculated based on the assumption that the organism is immersed 100% of the time in an infinite and uniformly contaminated source region. This assumption takes into account that some terrestrial animals reside well below ground for a substantial fraction of the time, and it is appropriately conservative for purposes of screening.

For exposure to contaminated soil, the desired units for the external dose coefficients are rad/d per pCi/g. Therefore, based on the calculations for contaminated sediments discussed in the previous section, the external dose coefficient for exposure to contaminated soil is given by:

$$(d_{ext})_{soil}\left(\frac{\text{rad/d}}{\text{pCi/g}}\right) = (5.12E-05)E_{photons,electrons}\left(\frac{\text{MeV}}{\text{dis}}\right)$$

If the desired units for the external dose coefficients are Gy/y per Bq/kg, the factor by which the decay energy is multiplied is 5.05E-06.

The screening-level external dose coefficients for exposure of terrestrial biota to selected radionuclides in contaminated soil calculated as described above are given in Table 2.3. The values for contaminated soil are twice the values for contaminated sediments in Table 2.1.

2.1.3 Discussion of Results

Several points about the screening-level external dose coefficients in Tables 2.1-2.3 should be noted. The first point concerns the treatment of radioactive decay chains in obtaining the results.

Several radionuclides - including Sr-90, Zr-95, Sb-125, Cs-137, Ce-144, Pb-210, Ra-226, Ra-228, Ac-227, Th-228, Th-229, U-235, U-238, Np-237, and Am-243 - have radioactive decay products that are sufficiently short-lived that the decay products are assumed to be in secular equilibrium with the parent radionuclide in each environmental medium. For these radionuclides, the external dose coefficients are the sum of the values for the parent and its indicated short-lived decay products, taking into account the branching fractions in the decay of the parent.

For several radionuclides, however, the external dose coefficients do not include possible contributions from decay products that are sufficiently long-lived that they may not be in activity

equilibrium with the parent radionuclide, even though the contributions from the decay products may be significant. The radionuclides of concern (with the decay products in parentheses) include Ra-226 (Pb-210), Ra-228 (Th-228), Th-232 (Ra-228 and Th-228), Pa-231 (Ac-227), and U-232 (Th-228). If separate data on the concentrations of the shorter-lived decay products in sediments, water, or soil are not available, the decay products could be assumed to be in activity equilibrium with the parent, and the dose coefficients for the parent and the decay products should be added. This approach may or may not be conservative, depending on differences in the environmental behavior of the parent and its decay products.

The second point concerns the importance of the external dose coefficients for exposure to contaminated water in Table 2.2. For most radionuclides, the concentration in aquatic animals relative to the concentration in water should be considerably greater than unity (Kennedy and Strenge 1992). Therefore, the dose rate from internal exposure calculated for purposes of screening by assuming that all radiations emitted in the decay of radionuclides in an organism are absorbed in the organism, usually would be considerably higher than the screening-level dose rate from external exposure. In addition, for most radionuclides, the solid/solution distribution coefficient (K_d) in sediments should be considerably greater than unity (Onishi et al. 1981). Therefore, for the assumption of exposure at the sediment-water interface, the screening-level dose rate from external exposure to contaminated sediments should be higher in most cases than the corresponding dose rate from external exposure to contaminated water.

Based on these arguments, the screening-level external dose coefficients for exposure of aquatic animals to contaminated water in Table 2.2 are unlikely to be important for most radionuclides in determining screening-level concentrations in water. Rather, the screening-level concentrations of most radionuclides in aquatic environments should be based on considerations of external exposure to contaminated sediments and internal exposure.

The third point concerns a comparison of the screening-level external dose coefficients obtained in this technical standard with values given by Amiro (1997). The calculations of Amiro assumed that the organism is located 0.1 m below the surface of a semi-infinite, uniformly contaminated body of sediment, water, or soil. Compared with the assumptions of exposure in an infinite medium (soil) or at the boundary of a semi-infinite medium (sediments and water) used in this technical standard, Amiro's assumption is less conservative for exposure to contaminated soil but more conservative for exposure to contaminated sediments and water. In addition, the external dose coefficients of Amiro were calculated for a human phantom, rather than a point receptor, and the calculated values for photons apply at the body surface and the calculated values for electrons apply at a depth of 70 μm in tissue (an aerial thickness of 0.7 mg/cm^2). For high-energy photon emitters, the photon dose rate at the body surface of a human phantom is slightly higher than the dose rate in the source medium itself, but the difference is not significant. However, the depth in tissue for calculating the electron dose rate assumed by Amiro is considerably less conservative than the assumption in this technical standard of exposure at the surface of a very small organism, because the minimum electron energy that results in a non-zero dose at a depth of 70 μm is about 70 keV (Kocher and Eckerman 1981) but all such lower-energy electrons are taken into account in obtaining the

present results. Finally, in the approach to screening developed by Amiro, the external dose coefficients cannot be calculated simply on the basis of the energies of photons and electrons in radioactive decay, and results for radionuclides not considered by Amiro are not readily obtainable.

Table 2.1 Screening-Level External Dose Coefficients for Exposure of Aquatic Animals to Contaminated Sediments (These values were also used for exposure of riparian animals to contaminated sediments.)

Radionuclide[a]	Decay Energy (MeV)[b]	External Dose Coefficient	
		rad/d per pCi/g	Gy/y per Bq/kg
^{3}H	0.0057	1.5E-07	1.4E-08
^{14}C	0.0495	1.3E-06	1.2E-07
^{32}P	0.6949	1.8E-05	1.8E-06
^{60}Co	2.6016	6.7E-05	6.6E-06
^{59}Ni	0.0067	1.7E-07	1.7E-08
^{63}Ni	0.0171	4.4E-07	4.3E-08
^{65}Zn	0.5904	1.5E-05	1.5E-06
$^{90}Sr + {}^{90}Y$	1.1305	2.9E-05	2.8E-06
$^{95}Zr + {}^{95}Nb$	1.6614	4.3E-05	4.2E-06
^{94}Nb	1.7027	4.4E-05	4.3E-06
^{99}Tc	0.0846	2.2E-06	2.1E-07
$^{125}Sb + {}^{125m}Te$	0.5670	1.5E-05	1.4E-06
^{129}I	0.0789	2.0E-06	2.0E-07
^{131}I	0.5715	1.5E-05	1.4E-06
^{134}Cs	1.7171	4.4E-05	4.3E-06
^{135}Cs	0.0563	1.4E-06	1.4E-07
$^{137}Cs + {}^{137m}Ba$	0.7966	2.0E-05	2.0E-06
$^{144}Ce + {}^{144}Pr$	1.3517	3.5E-05	3.4E-06
^{154}Eu	1.5269	3.9E-05	3.8E-06
^{155}Eu	0.1224	3.1E-06	3.1E-07
$^{210}Pb + {}^{210}Bi$	0.4279	1.1E-05	1.1E-06
$^{226}Ra + D^{C}$	2.7023	6.9E-05	6.8E-06
$^{228}Ra + {}^{228}Ac^{d}$	1.3677	3.5E-05	3.4E-06
$^{227}Ac + D^{e}$	1.4916	3.8E-05	3.8E-06
$^{228}Th + D^{f}$	2.4310	6.2E-05	6.1E-06
$^{229}Th + D^{g}$	1.2282	3.1E-05	3.1E-06
^{230}Th	0.0143	3.7E-07	3.6E-08
$^{232}Th^{h}$	0.0121	3.1E-07	3.0E-08
$^{231}Pa^{i}$	0.0727	1.9E-06	1.8E-07
$^{232}U^{j}$	0.0162	4.1E-07	4.1E-08
^{233}U	0.0037	9.5E-08	9.3E-09
^{234}U	0.0128	3.3E-07	3.2E-08
$^{235}U + {}^{231}Th$	0.3729	9.5E-06	9.4E-07
$^{238}U + D^{k}$	0.9154	2.3E-05	2.3E-06
$^{237}Np + {}^{233}Pa$	0.5049	1.3E-05	1.3E-06

Table 2.1 (Continued) Screening-Level External Dose Coefficients for Exposure of Aquatic Animals to Contaminated Sediments (These values were also used for exposure of riparian animals to contaminated sediments.)

Radionuclide[a]	Decay Energy (MeV)[b]	External Dose Coefficient	
		rad/d per pCi/g	Gy/y per Bq/kg
^{238}Pu	0.0099	2.5E-07	2.5E-08
^{239}Pu	0.0056	1.4E-07	1.4E-08
^{240}Pu	0.0098	2.5E-07	2.5E-08
^{241}Pu	0.0052	1.3E-07	1.3E-08
^{241}Am	0.0575	1.5E-06	1.4E-07
^{243}Am + ^{239}Np	0.4990	1.3E-05	1.3E-06
^{242}Cm	0.0092	2.4E-07	2.3E-08
^{243}Cm	0.2547	6.5E-06	6.4E-07
^{244}Cm	0.0079	2.0E-07	2.0E-08

(a) Short-lived decay products assumed to be in activity equilibrium are listed with parent radionuclide, and "D" denotes multiple decay products listed in separate footnote. Contributions to dose coefficient from decay products take into account branching fractions in decay of parent radionuclide (Kocher 1981).
(b) Total energy of all photons and electrons emitted per decay of radionuclide from Kocher (1980).
(c) Short-lived decay products include ^{222}Rn, ^{214}Pb, ^{214}Bi, and ^{214}Po. Possible contributions to dose coefficient from ^{210}Pb decay product are not included, but dose coefficient for decay product is listed separately.
(d) Possible contributions to dose coefficient from ^{228}Th decay product are not included, but dose coefficient for decay product is listed separately.
(e) Short-lived decay products include ^{227}Th, ^{223}Fr, ^{223}Ra, ^{219}Rn, ^{215}Po, ^{211}Pb, ^{211}Bi, and ^{207}Tl.
(f) Short-lived decay products include ^{224}Ra, ^{220}Rn, ^{212}Pb, ^{212}Bi, and ^{208}Tl.
(g) Short-lived decay products include ^{225}Ra, ^{225}Ac, ^{221}Fr, ^{217}At, ^{213}Bi, ^{209}Tl, and ^{209}Pb.
(h) Possible contributions to dose coefficient from ^{228}Ra and ^{228}Th decay products are not included, but dose coefficients for decay products are listed separately.
(i) Possible contributions to dose coefficient from ^{227}Ac decay product are not included, but dose coefficient for decay product is listed separately.
(j) Possible contributions to dose coefficient from ^{228}Th decay product are not included, but dose coefficient for decay product is listed separately.
(k) Short-lived decay products include ^{234}Th, ^{234}Pa, and ^{234}Pa.

Table 2.2 Screening-Level External Dose Coefficients for Exposure of Aquatic Animals to Contaminated Water (These values were also used for exposure of riparian animals to contaminated water.)

Radionuclide[a]	Decay Energy (MeV)[b]	External Dose Coefficient	
		rad/d per pCi/L	Gy/y per Bq/m^3
^{3}H	0.0057	1.5E-10	1.4E-11
^{14}C	0.0495	1.3E-09	1.2E-10
^{32}P	0.6949	1.8E-08	1.8E-09
^{60}Co	2.6016	6.7E-08	6.6E-09
^{59}Ni	0.0067	1.7E-10	1.7E-11
^{63}Ni	0.0171	4.4E-10	4.3E-11
^{65}Zn	0.5904	1.5E-08	1.5E-09
^{90}Sr + ^{90}Y	1.1305	2.9E-08	2.8E-09
^{95}Zr + ^{95}Nb	1.6614	4.3E-08	4.2E-09
^{94}Nb	1.7027	4.4E-08	4.3E-09
^{99}Tc	0.0846	2.2E-09	2.1E-10
^{125}Sb + ^{125m}Te	0.5670	1.5E-08	1.4E-09
^{129}I	0.0789	2.0E-09	2.0E-10
^{131}I	0.5715	1.5E-08	1.4E-09
^{134}Cs	1.7171	4.4E-08	4.3E-09
^{135}Cs	0.0563	1.4E-09	1.4E-10
^{137}Cs + ^{137m}Ba	0.7966	2.0E-08	2.0E-09
^{144}Ce + ^{144}Pr	1.3517	3.5E-08	3.4E-09
^{154}Eu	1.5269	3.9E-08	3.8E-09
^{155}Eu	0.1224	3.1E-09	3.1E-10
^{210}Pb + ^{210}Bi	0.4279	1.1E-08	1.1E-09
^{226}Ra + D[c]	2.7023	6.9E-08	6.8E-09
^{228}Ra + ^{228}Ac[d]	1.3677	3.5E-08	3.4E-09
^{227}Ac + D[e]	1.4916	3.8E-08	3.8E-09
^{228}Th + D[f]	2.4310	6.2E-08	6.1E-09
^{229}Th + D[g]	1.2282	3.1E-08	3.1E-09
^{230}Th	0.0143	3.7E-10	3.6E-11
^{232}Th[h]	0.0121	3.1E-10	3.0E-11
^{231}Pa[i]	0.0727	1.9E-09	1.8E-10
^{232}U[j]	0.0162	4.1E-10	4.1E-11
^{233}U	0.0037	9.5E-11	9.3E-12
^{234}U	0.0128	3.3E-10	3.2E-11
^{235}U + ^{231}Th	0.3729	9.5E-09	9.4E-10
^{238}U + D[k]	0.9154	2.3E-08	2.3E-09
^{237}Np + ^{233}Pa	0.5049	1.3E-08	1.3E-09

Table 2.2 (Continued) Screening-Level External Dose Coefficients for Exposure to Aquatic Animals to Contaminated Water (These values were also used for exposure of riparian animals to contaminated water.)

Radionuclide[a]	Decay Energy (MeV)[b]	External Dose Coefficient	
		rad/d per pCi/L	Gy/y per Bq/m^3
^{238}Pu	0.0099	2.5E-10	2.5E-11
^{239}Pu	0.0056	1.4E-10	1.4E-11
^{240}Pu	0.0098	2.5E-10	2.5E-11
^{241}Pu	0.0052	1.3E-10	1.3E-11
^{241}Am	0.0575	1.5E-09	1.4E-10
^{243}Am + ^{239}Np	0.4990	1.3E-08	1.3E-09
^{242}Cm	0.0092	2.4E-10	2.3E-11
^{243}Cm	0.2547	6.5E-09	6.4E-10
^{244}Cm	0.0079	2.0E-10	2.0E-11

(a) Short-lived decay products assumed to be in activity equilibrium are listed with parent radionuclide, and "D" denotes multiple decay products listed in separate footnote. Contributions to dose coefficient from decay products take into account branching fractions in decay of parent radionuclide (Kocher 1981).
(b) Total energy of all photons and electrons emitted per decay of radionuclide from Kocher (1980).
(c) Short-lived decay products include ^{222}Rn, ^{214}Pb, ^{214}Bi, and ^{214}Po. Possible contributions to dose coefficient from ^{210}Pb decay product are not included, but dose coefficient for decay product is listed separately.
(d) Possible contributions to dose coefficient from ^{228}Th decay product are not included, but dose coefficient for decay product is listed separately.
(e) Short-lived decay products include ^{227}Th, ^{223}Fr, ^{223}Ra, ^{219}Rn, ^{215}Po, ^{211}Pb, ^{211}Bi, and ^{207}Tl.
(f) Short-lived decay products include ^{224}Ra, ^{220}Rn, ^{212}Pb, ^{212}Bi, and ^{208}Tl.
(g) Short-lived decay products include ^{225}Ra, ^{225}Ac, ^{221}Fr, ^{217}At, ^{213}Bi, ^{209}Tl, and ^{209}Pb.
(h) Possible contributions to dose coefficient from ^{228}Ra and ^{228}Th decay products are not included, but dose coefficients for decay products are listed separately.
(i) Possible contributions to dose coefficient from ^{227}Ac decay product are not included, but dose coefficient for decay product is listed separately.
(j) Possible contributions to dose coefficient from ^{228}Th decay product are not included, but dose coefficient for decay product is listed separately.
(k) Short-lived decay products include ^{234}Th, ^{234}Pa, and ^{234}Pa.

Table 2.3 Screening-Level External Dose Coefficients for Exposure of Terrestrial Biota to Contaminated Soil

Radionuclide[a]	Decay Energy (MeV)[b]	External Dose Coefficient	
		rad/d per pCi/g	Gy/y per Bq/kg
^{3}H	0.0057	2.9E-07	2.9E-08
^{14}C	0.0495	2.5E-06	2.5E-07
^{32}P	0.6949	3.6E-05	3.5E-06
^{60}Co	2.6016	1.3E-04	1.3E-05
^{59}Ni	0.0067	3.4E-07	3.4E-08
^{63}Ni	0.0171	8.8E-07	8.6E-08
^{65}Zn	0.5904	3.0E-05	3.0E-06
$^{90}Sr + {}^{90}Y$	1.1305	5.8E-05	5.7E-06
$^{95}Zr + {}^{95}Nb$	1.6614	8.5E-05	8.4E-06
^{94}Nb	1.7027	8.7E-05	8.6E-06
^{99}Tc	0.0846	4.3E-06	4.3E-07
$^{125}Sb + {}^{125m}Te$	0.5670	2.9E-05	2.9E-06
^{129}I	0.0789	4.0E-06	4.0E-07
^{131}I	0.5715	2.9E-05	2.9E-06
^{134}Cs	1.7171	8.8E-05	8.7E-06
^{135}Cs	0.0563	2.9E-06	2.8E-07
$^{137}Cs + {}^{137m}Ba$	0.7966	4.1E-05	4.0E-06
$^{144}Ce + {}^{144}Pr$	1.3517	6.9E-05	6.8E-06
^{154}Eu	1.5269	7.8E-05	7.7E-06
^{155}Eu	0.1224	6.3E-06	6.2E-07
$^{210}Pb + {}^{210}Bi$	0.4279	2.2E-05	2.2E-06
^{226}Ra + Dc	2.7023	1.4E-04	1.4E-05
$^{228}Ra + {}^{228}Ac$[d]	1.3677	7.0E-05	6.9E-06
^{227}Ac + D[e]	1.4916	7.6E-05	7.5E-06
^{228}Th + D[f]	2.4310	1.2E-04	1.2E-05
^{229}Th + D[g]	1.2282	6.3E-05	6.2E-06
^{230}Th	0.0143	7.3E-07	7.2E-08
^{232}Th[h]	0.0121	6.2E-07	6.1E-08
^{231}Pa[i]	0.0727	3.7E-06	3.7E-07
^{232}U[j]	0.0162	8.3E-07	8.2E-08
^{233}U	0.0037	1.9E-07	1.9E-08
^{234}U	0.0128	6.6E-07	6.5E-08
$^{235}U + {}^{231}Th$	0.3729	1.9E-05	1.8E-06
^{238}U + D[k]	0.9154	4.7E-05	4.6E-06
$^{237}Np + {}^{233}Pa$	0.5049	2.6E-05	2.5E-06

Table 2.3 (Continued) Screening-Level External Dose Coefficients for Exposure of Terrestrial Biota to Contaminated Soil

Radionuclide[a]	Decay Energy (MeV)[b]	External Dose Coefficient	
		rad/d per pCi/g	Gy/y per Bq/kg
^{238}Pu	0.0099	5.1E-07	5.0E-08
^{239}Pu	0.0056	2.9E-07	2.8E-08
^{240}Pu	0.0098	5.0E-07	4.9E-08
^{241}Pu	0.0052	2.7E-07	2.6E-08
^{241}Am	0.0575	2.9E-06	2.9E-07
^{243}Am + ^{239}Np	0.4990	2.6E-05	2.5E-06
^{242}Cm	0.0092	4.7E-07	4.6E-08
^{243}Cm	0.2547	1.3E-05	1.3E-06
^{244}Cm	0.0079	4.0E-07	4.0E-08

(a) Short-lived decay products assumed to be in activity equilibrium are listed with parent radionuclide, and "D" denotes multiple decay products listed in separate footnote. Contributions to dose coefficient from decay products take into account branching fractions in decay of parent radionuclide (Kocher 1981).
(b) Total energy of all photons and electrons emitted per decay of radionuclide from Kocher (1980).
(c) Short-lived decay products include ^{222}Rn, ^{214}Pb, ^{214}Bi, and ^{214}Po. Possible contributions to dose coefficient from ^{210}Pb decay product are not included, but dose coefficient for decay product is listed separately.
(d) Possible contributions to dose coefficient from ^{228}Th decay product are not included, but dose coefficient for decay product is listed separately.
(e) Short-lived decay products include ^{227}Th, ^{223}Fr, ^{223}Ra, ^{219}Rn, ^{215}Po, ^{211}Pb, ^{211}Bi, and ^{207}Tl.
(f) Short-lived decay products include ^{224}Ra, ^{220}Rn, ^{212}Pb, ^{212}Bi, and ^{208}Tl.
(g) Short-lived decay products include ^{225}Ra, ^{225}Ac, ^{221}Fr, ^{217}At, ^{213}Bi, ^{209}Tl, and ^{209}Pb.
(h) Possible contributions to dose coefficient from ^{228}Ra and ^{228}Th decay products are not included, but dose coefficients for decay products are listed separately.
(i) Possible contributions to dose coefficient from ^{227}Ac decay product are not included, but dose coefficient for decay product is listed separately.
(j) Possible contributions to dose coefficient from ^{228}Th decay product are not included, but dose coefficient for decay product is listed separately.
(k) Short-lived decay products include ^{234}Th, ^{234}Pa, and ^{234}Pa.

2.2 Internal Dose Coefficients

This section presents the approach used to calculate internal dose coefficients that can be used in general screening for internal exposure of aquatic and terrestrial biota to selected radionuclides. A table of screening-level internal DCFs is provided.

2.2.1 Approach to Calculating Internal Dose Coefficients

Internal dose conversion factors (Gy y^{-1} per Bq kg^{-1}) were derived for unit concentrations of each of the target radionuclides in tissue. Reference decay energies and abundances were taken from ICRP 38 (1983) for each of the target radionuclides and its progeny The default dose factor includes buildup of progeny with half-lives less than 100 y. The calculations assume all of the energies of radioactive decay were retained in the tissue of the organism (i.e., the organism was presumed to be very large in size). The radionuclides were presumed to be homogeneously distributed in the tissue. The default internal dose factors include a dose-

modifying factor of 20 (i.e., Q or w_R = 20) for alpha particles and the alpha-emitting progeny of chain-decaying nuclides. However, the RAD-BCG Calculator is constructed such that the dose-modifying factor can be modified. See Module 2, Section 7 for a detailed discussion on the rationale for the radiation weighting factor selected.

The RAD-BCG Calculator Provides the Capability to Modify the Internal DCFs

Internal DCFs. The default internal DCFs used in the graded approach include the contribution from build-up of progeny with half-lives less than 100y. A user can select whether or not the energy of the progeny will be included in the calculations. This is done in the Dose Factors and Common Parameters Spreadsheet.

Radiation Weighting Factor for Alpha Emitters. The default value of the radiation weighting factor (default = 20) for alpha particles and the alpha-emitting progeny of chain-decaying radionuclides can be modified in the Dose Factors and Common Parameters Spreadsheet.

The dose factors were calculated as the sum of all decay energies and multiplied by appropriate unit conversion factors. The equation used to calculate an internal dose factor for a specific radionuclide is shown below. The resultant dose factors are presented in Table 2.4. For internal exposure to contaminants, the units for the dose coefficients were calculated as Gy/y per Bq/kg of wet tissue.

$$DCF_{internal,i} = \left(\frac{1 dis \cdot s^{-1}}{Bq}\right)\left(\sum_i \sum_j Y_j E_j Q_j\right)\left(1.6022E{-}13\ J\ MeV^{-1}\right)\left(3.1536E07 s \cdot y^{-1}\right)\frac{1 Gy}{1 J \cdot kg^{-1}}$$

where the following terms apply:

$DCF_{internal,i}$ = Gy/y per Bq/kg of wet tissue for radionuclide i;

Y_j = yield (abundance) of radiation j per disintegration of nuclide i;

E_j = energy (MeV) of radiation j for nuclide i; and

Q_j is the radiation weighting factor (quality factor, also called W_R) for radiation j of nuclide i.

The dose factors can also be expressed in rad/d per pCi/g, where all other factors have been defined:

$$DCF_{internal,i} = \left(\frac{1 dis \cdot s^{-1}}{Bq}\right)\frac{0.037 Bq}{pCi}\left(\sum_i \sum_j Y_j E_j Q_j\right)(1.6022E{-}06\ erg \cdot MeV^{-1})(8.64E04 s \cdot d^{-1})\frac{0.01 g \cdot rad}{erg}$$

Table 2.4 Screening Level Internal Dose Factors

Radionuclide	Internal dose with progeny[a]		Internal dose without progeny	
	Gy/y per Bq/kg (wet)	Rad/d per pCi/g (wet)	Gy/y per Bq/kg (wet)	Rad/d per pCi/g (wet)
^{241}Am	5.6E-04	5.7E-03	5.6E-04	5.7E-03
^{144}Ce	6.8E-06	6.9E-05	5.6E-07	5.7E-06
^{135}Cs	3.4E-07	3.4E-06	3.4E-07	3.4E-06
^{137}Cs	4.3E-06	4.3E-05	9.4E-07	9.6E-06
^{60}Co	1.3E-05	1.3E-04	1.3E-05	1.3E-04
^{154}Eu	7.6E-06	7.7E-05	7.6E-05	7.7E-05
^{155}Eu	6.2E-07	6.3E-06	6.2E-07	6.3E-06
^{3}H	2.9E-08	2.9E-07	2.9E-08	2.9E-07
^{129}I	4.5E-07	4.5E-06	4.5E-07	4.5E-06
^{131}I	2.9E-06	2.9E-05	2.9E-06	2.9E-05
^{239}Pu	5.3E-04	5.4E-03	5.3E-04	5.4E-03
^{226}Ra	3.0E-03	3.1E-02	4.9E-04	5.0E-03
^{228}Ra	3.6E-03	3.7E-02	8.5E-08	8.6E-07
^{125}Sb	2.7E-06	2.7E-05	2.7E-06	2.7E-05
^{90}Sr	5.7E-06	5.8E-05	9.9E-07	1.0E-05
^{99}Tc	5.1E-07	5.2E-06	5.1E-07	5.2E-06
^{232}Th	4.1E-03	4.1E-02	4.1E-04	4.2E-03
^{233}U	4.9E-04	5.0E-03	4.9E-04	5.0E-03
^{234}U	4.9E-04	5.0E-03	4.9E-04	5.0E-03
^{235}U	4.5E-04	4.6E-03	4.5E-04	4.6E-03
^{238}U	4.4E-04	4.5E-03	4.3E-04	4.4E-03
^{65}Zn	3.0E-06	3.0E-05	3.0E-06	3.0E-05
^{95}Zr	8.4E-06	8.5E-05	4.3E-06	4.4E-05

(a) Includes listed radiations (α β γ, X) and an RBE of 20 for alpha particles. Progeny with half-lives less than 100 y are included at 100% abundance.

3 Equations and Models for Calculating Dose to Biota and Deriving BCGs

Based on the potential pathways of exposure, BCGs were derived for surface water, sediment, and soil. Calculated using conservative assumptions, the BCGs are intended to preclude the relevant biota from being exposed to radiation levels in excess of established or recommended biota dose limits. Determination of compliance with the dose limits requires that all organism-relevant environmental media be evaluated at the same time. This is done by using the "sum of fractions" approach commonly used in evaluating radionuclide discharges to the environment.

3.1 An Important Note on Estimating Internal Tissue Concentrations for Use in Dose Equations: The Lumped Parameter

For most radionuclides, the single most important predictor of biota dose is the method used to estimate internal tissue concentrations. For the general screening phase of the graded approach, lumped parameters were used to provide estimates of organism tissue concentration, and ultimately derive the BCG corresponding to each radionuclide, media, and organism type. The technical literature contains reference to empirically-based parameters which measure concentrations of contaminants in an organism relative to the surrounding media. These ratios are called "concentration ratios," "concentration factors," or "wet-weight concentration ratios" (B_{iv}s). These lumped parameter (e.g., B_{iv}) values are available for many radionuclides for plant:soil and for aquatic species:water. In a few instances they are also available for animal:soil or sediment. The advantage of using one of these factors is that it allows the prediction of tissue concentration based on simple measurements of contamination in environmental media such as water, sediment and soil.

The selection of a value for this lumped parameter becomes problematic, however, when considering the range of organism types meant to be covered by the graded approach. For example, there is very limited data available for riparian and terrestrial animals (i.e., very limited animal:water, animal:soil, and animal:sediment concentration ratios). As the graded approach methodology evolved it became apparent that these data gaps (e.g., for selecting appropriate lumped parameters) needed to be addressed. Two alternative approaches for deriving and selecting lumped parameters were evaluated:

- **Calculating the lumped parameter values by multiplying related concentration ratios (product approach).** For example, the product of plant:soil and animal:plant concentration ratios yields an animal:soil ratio which may be used as the lumped parameter for a terrestrial animal. This approach must be used with caution, as the data used in the process are most likely from different sources. This approach is also hampered by the general lack of environmental data.

- **Calculating the lumped parameter values by using uncertainty analysis on the kinetic/allometric method.** The kinetic/allometric method, as used in the analysis phase of the graded approach, is based on mathematically modeling the exposure of an

organism using simplistic, first-order kinetic reactions. There are several allometric equations which relate body size to many of the parameters contributing to internal dose (e.g., to include ingestion rates, life span, and inhalation rate). Uncertainty analysis (e.g., using Monte Carlo techniques) on each of the allometric equations, and on their corresponding parameters varied over their known ranges of values, can provide an upper bound estimate (i.e., at the 95th percentile) of lumped parameter values for those organism types (riparian and terrestrial animals) for which there is limited empirical data.

These alternative approaches, and the rationale for their use, are discussed further in Section 3.4. Figure 3.1 shows the logic flow for the derivation and selection of default lumped parameter values employed in the general screening phase for each of the four organism types addressed in the graded approach.

Figure 3.1 Process for Selecting Default B_{iv}/Lumped Parameter Values for Use in the General Screening Phase of the Graded Approach

	Aquatic Animal	Riparian Animal	Terrestrial Plant	Terrestrial Animal
(1) B_{iv}s/ lumped parameters compiled for each organism type (literature searches; models; empirical data)	Very good empirical data	Fair to limited	Very good empirical data	Fair to limited
(2) B_{iv}s/ lumped parameter data sets reviewed for quality, quantity, and range of values	Very good	Limited: RA: water RA: sediment Some: RA(fs) : sediment RA: RA(fs)	Very good	Limited: TA: water TA: soil Some: TA: soil TA: TP
(3) For Fair/ Limited Data:				
(3a) B_{iv}/ lumped parameters estimated using product approach (e.g. multiplying concentration ratios, CRs)	-	(RA(fs) : sediment) • (RA • RA(fs)) yields (RA: sediment)	-	(TP: soil) • (TA: TP) yields (TA: soil)
(3b) Lumped parameters estimated by using uncertainty analysis on the kinetic/allometric method (95th percentile of resulting distributions)	-	Uncertainty analysis on each allometric equation and their corresponding parameters varied over their known ranges of values.	-	Uncertainty analysis on each allometric equation and their corresponding parameters varied over their known ranges of values.
(3c) B_{iv}/ lumped parameter value comparison: product approach; uncertainty analysis (K/A method); available empirical data	-	B_{iv}/ lumped parameter comparison: product approach; uncertainty analysis (K/A method); empirical data	-	B_{iv}/ lumped parameter comparison: product approach; uncertainty analysis (K/A method); empirical data
(4) B_{iv}/ lumped parameter values selected as default values for general screening.	empirical values used	Preference for empirical values where available and of sufficient quality; otherwise uncertainty analysis (K/A method) values	empirical values used	Preference for empirical values where available and of sufficient quality; otherwise uncertainty analysis (K/A method) values

KEY	
AA	= Aquatic Animal
RA	= Riparian Animal
TP	= Terrestrial Plant
TA	= Terrestrial Animal
RA(fs)	= Food source to a Riparian Animal
Uncertainty Analysis (K/A Method)	= Uncertainty analysis on kinetic/allometric method

3.2 Equations and Models for Aquatic Systems

3.2.1 Aquatic Animals

Sediment BCGs for Aquatic Animals. The conceptual model for aquatic animals places the organism at the sediment-water interface. In this screening model, sediment presents an external dose hazard to the aquatic animal, with the BCG therefore based on a semi-infinite exposure model. Uptake of contaminants from the sediment to the organism is implicitly addressed via the empirical organism to water lumped parameter discussed in following sections. The method used to derive the aquatic animal BCGs for exposure to a single nuclide in contaminated sediment is:

$$BCG(sediment)_{i,aquatic\ animal} = \frac{365.25 \cdot DL_{aa}}{CF_{aa} \cdot DCF_{ext,sediment,i}}$$

Equation 1

where $BCG(sediment)_{i,aquatic\ animal}$ (Bq kg^{-1}) is the concentration of nuclide *i* in sediment which, based on the screening level assumptions, numerically equates to a dose rate of DL_{aa} (0.01 Gy d^{-1}) to the aquatic animal;

365.25 (days per year) is a conversion factor;

DL_{aa} (0.01 Gy d^{-1}) is the dose limit for aquatic animals. This limit can be adjusted by the user if so directed by an appropriate agency;

$DCF_{ext,sediment,i}$ (Gy y^{-1} per Bq kg^{-1}) is the external dose conversion factor used to estimate the dose rate to the tissues of the aquatic animal from nuclide *i* in the sediment; and

CF_{aa} (dimensionless) is the correction factor for area or organism residence time. This correction factor is set at a default of 1.

It should be noted that Equation 1 can also be used to evaluate compliance for aquatic plants. Both the dose factor and dose limit are the same.

Water BCGs for Aquatic Animals. The conceptual model for aquatic animals places the organism at the sediment-water interface. In this screening model, water presents both an internal and external dose hazard to the aquatic animal. Lumped parameters (e.g., bioaccumulation factors) are used to estimate the extent of internal contamination (and by extension, the dose), and external exposure is assessed with a semi-infinite source term. The method used to derive the screening-level aquatic animal BCGs for exposure to a single nuclide in contaminated water is:

$$BCG(water)_{i,aquatic\ animal} = \frac{365.25 \cdot DL_{aa}}{CF_{aa} \cdot \left[\left(0.001 \cdot B_{iv,aa} \cdot DCF_{internal,i} \right) + \left(DCF_{external,water,i} \right) \right]}$$

Equation 2

where $BCG(water)_{i,aquatic\ animal}$ (Bq m^{-3}) is the concentration of nuclide *i* in water which, based on the screening level assumptions, numerically equates to a dose rate of DL_{aa} (0.01 Gy d^{-1}) to the aquatic animal;

DL_{aa} (0.01 Gy d^{-1}) is the dose limit for aquatic animals. This limit can be adjusted by the user if so directed by an appropriate agency;

0.001 is the conversion factor for L to m^3;

$B_{iv,aa,i}$ (Lkg^{-1}) is the fresh mass aquatic animal to water concentration factor for nuclide *i*;

$DCF_{internal,i}$ (Gy y^{-1} per Bq kg^{-1}) is the dose conversion factor used to estimate the dose rate to the tissues from nuclide *i* in tissues;

$DCF_{external,\ water,i}$ (Gy y^{-1} per Bq m^{-3}) is the dose conversion factor used to estimate the dose rate to the aquatic animal from submersion in contaminated water; and

all other terms have been defined.

It should be noted that Equation 2 can also be used to evaluate compliance for aquatic plants. Both the dose factor and the dose limit are the same. In lieu of an aquatic animal B_{iv}, simply substitute an aquatic plant concentration factor.

3.2.2 Riparian Animals

Sediment BCGs for Riparian Animals. The conceptual model for riparian animals also places the organism at the sediment-water interface (as does the aquatic animal model). However, in this screening model, sediment presents both an internal and external dose hazard to the riparian animal. Lumped parameters are used to estimate the extent of internal contamination (and by extension, the dose), and external exposure is assessed with a semi-infinite source term. The method used to derive the riparian animal BCGs for exposure to a single nuclide in contaminated sediment is:

$$BCG(sediment)_{i,riparian\ animal} = \frac{365.25 \cdot DL_{ra}}{CF_{ra} \cdot [(LP_{ra,sed,i} \cdot DCF_{internal,i}) + (DCF_{ext,sediment,i})]}$$

Equation 3

where $BCG(sediment)_{i,riparian\ animal}$ (Bq kg^{-1}) is the concentration of nuclide *i* in sediment, based on the screening level assumptions, numerically equates to a dose rate of DL_{ra} (0.001 Gy d^{-1}) to the riparian animal;

DL_{aa} (0.001 Gy d^{-1}) is the recommended dose limit for riparian animals. This limit can be adjusted by the user if so directed by an appropriate agency;

$LP_{ra,sed,i}$ (dimensionless) is the fresh mass riparian animal to sediment concentration factor of nuclide *i*;

CF_{ra} (dimensionless) is the correction factor for area or organism residence time for the riparian organism. This correction factor is set at a default of 1; and

all other terms have been defined.

Water BCGs for Riparian Animals. As noted previously, the conceptual model for riparian animals has the animal situated at the sediment-water interface. In assessing potential contributors to dose, water presents both an internal and external dose hazard. As before, lumped parameters are used to estimate the extent of internal contamination. External exposure is assessed with a semi-infinite source term. The method used to derive the screening-level riparian animal BCGs for exposure to a single nuclide in contaminated water is as follows:

$$BCG(water)_{i,riparian\ animal} = \frac{365.25 \cdot DL_{ra}}{CF_{ra} \cdot [(0.001 \cdot LP_{ra,water,i} \cdot DCF_{internal,i}) + (DCF_{ext,water,i})]}$$

Equation 4

where $BCG(water)_{i,\ riparian\ animal}$ (Bq m^{-3}) is the concentration of nuclide *i* in water, which based on the screening level assumptions, numerically equates to a dose rate of DL_{aa} (0.001 Gy d^{-1}) to the riparian animal;

$LP_{ra,water,\ i}$ (L/kg) is the fresh mass riparian animal to water concentration factor of nuclide *i*; and

all other terms have been defined.

3.2.3 Important Considerations When Implementing Equations and Models in an Aquatic System Evaluation

For the aquatic environment, compliance with the dose limit is determined by comparison of the projected dose from both water and sediment. This is achieved by using a sum of fractions approach. The measured concentrations of radionuclides for the water and sediment pathways are each ratioed to their respective BCGs and the resultant values summed. If the total is less than one, then compliance (for that nuclide) is achieved. For multiple nuclides the process is repeated, with the sum of all fractions (the grand total) required to be less than one for compliance.

Co-located water and sediment samples. The preferred method of determining compliance is to use co-located water and sediment data. If such data are available, then compliance is determined in the manner described in the preceding paragraph.

Water and sediment samples not co-located. In situations where co-located water and sediment data are not available, the user estimates the missing data through use of the radionuclide-specific "most probable" distribution coefficient. If water data are present, but sediment data are unavailable, the missing sediment data are estimated through use of the following calculation:

$$C_{sediment} = 0.001 \cdot C_{water} \cdot K_{d,most\ probable}$$

Equation 5

where $C_{sediment}$ (Bq kg^{-1}) is the concentration of nuclide *i* in sediment;

0.001 ($m^3\ L^{-1}$) is the conversion factor for L to m^3;

C_{water} (Bq m^{-3}) is the concentration of nuclide *i* in water; and

$K_{d,most\ probable}$ (expressed as L kg^{-1} but also equates to mL g^{-1}) is the distribution coefficient used to relate the water concentration to the sediment concentration. In doing this calculation, median values of distribution coefficients were selected, rather than extreme values. For many nuclides, distribution coefficients range over several orders of magnitude. Selection of extreme values would result in unrealistic projections of water (or sediment) concentrations of radionuclides.

Conversely, if water data are unavailable, the RAD-BCG Calculator estimates the missing water data through use of the following calculation:

$$C_{water} = \frac{C_{sediment}}{0.001 \cdot K_{d,most\ probable}}$$

Equation 6

where all terms have been previously defined.

If the user has water data from one location, and sediment data from another (for the same radionuclide), they should use both approaches outlined above, and select the method which results in the highest (e.g., most conservative) partial fraction.

3.3 Equations and Models for Terrestrial Systems

3.3.1 Terrestrial Plants

Soil BCGs for Terrestrial Plants. In this screening model, soil provides both an internal and external dose hazard to plants. The conceptual model for terrestrial plants is based on the entire plant being surrounded by soil. While many plants may have a substantial portion of their mass above ground, the BCG thus derived, will be conservative. Lumped parameters (e.g., bioaccumulation factors) are used to estimate the extent of internal contamination (and by extension, the dose), and external exposure is assessed using an infinite source term. The lumped parameters used in the model account for aerial deposition onto plant surfaces with subsequent uptake. The method used to derive the BCGs for terrestrial plant exposure to a single nuclide in contaminated soil is:

$$BCG(soil)_{i,terrestrial\ plant} = \frac{365.25 \cdot DL_{tp}}{CF_{tp} \cdot [(B_{iv,tp,i} \cdot DC_{internal,i}) + (DCF_{ext,soil,i})]}$$

Equation 7

where $BCG(soil)_{i,terrestrial\ plant}$ (Bq kg^{-1}) is the concentration of nuclide *i* in soil which, based on the screening level assumptions, numerically equates to a dose rate of DL_{tp} (0.01 Gy d^{-1}) to the terrestrial plant;

DL_{tp} (0.01 Gy d^{-1}) is the recommended dose limit for terrestrial plants. This limit can be adjusted by the user if so directed by an appropriate agency;

$B_{iv,tp,i}$ (dimensionless) is the fresh mass terrestrial plant to soil concentration factor;

CF_{tp} (dimensionless) is the correction factor for area or time. This correction factor is set at a default of 1;

$DCF_{ext,\ soil,i}$ (Gy y^{-1} per Bq kg^{-1}) is the dose conversion factor used to estimate the dose

rate to the plant tissues from nuclide *i* in surrounding soils; and

all other terms are as previously defined.

It should be noted that the derivation of the water BCG for terrestrial plants only considers external exposure of plants from submersion in water. Although this may seem to ignore uptake of contaminants from pore water into the plant, there is very limited data available to support this type of calculation. The best estimator of internal deposition is the plant-to-soil uptake factor, utilized in Equation 7. If only water data is available, and no soil data (for example, measurements in irrigation water), you can use the relationship outlined in Equation 5 to predict the soil concentration and substitute this value into Equation 7.

Water BCGs for Terrestrial Plants. The conceptual model for terrestrial plants is based on the entire plant being surrounded by soil. However, the potential for exposure to contaminated water – from soil pore water or from irrigation exists. As a compromise to the methodology, external exposure from water was added. In this screening model, the BCG for water is based on a semi-infinite exposure model. The method used to derive the BCGs for terrestrial plant exposure to a single nuclide in contaminated water is:

$$BCG(water)_{i,terrestrial\ plant} = \frac{365.25 \cdot DL_{tp}}{CF_{tp} \cdot DCF_{ext,water,i}}$$

Equation 8

where $BCG(water)_{i,terrestrial\ plant}$ (Bq m^{-3}) is the concentration of nuclide *i* in soil which, based on the screening level assumptions, numerically equates to a dose rate of DL_{tp} (0.01 Gy d^{-1}) to the terrestrial plant; and

all other terms are as previously defined.

3.3.2 Terrestrial Animals

Soil BCGs for Terrestrial Animals. The screening conceptual model for terrestrial animals has the animal surrounded by soil. In assessing potential contributors to dose, soil presents both an internal and external dose pathway. As before, lumped parameters are used to estimate the extent of internal contamination (e.g., as might occur from ingestion or inhalation). External exposure is assessed with an infinite source term. The method used to derive the terrestrial animal BCGs for exposure to a single nuclide in contaminated soil is:

$$BCG(soil)_{i,terrestrial\ animal} = \frac{365.25 \cdot DL_{ta}}{CF_{ta} \cdot [(LP_{ta,soil,i} \cdot DCF_{internal,i}) + (DCF_{ext,soil,i})]}$$

Equation 9

where $BCG(soil)_{i,\ terrestrial\ animal}$ (Bq kg^{-1}) is the concentration of nuclide *i* in soil which, based on the screening level assumptions, numerically equates to a dose rate of DL_{ta} (0.001 Gy d^{-1}) to the terrestrial animal;

DL_{ta} (0.001 Gy d^{-1}) is the recommended dose limit for terrestrial animals. This limit can be adjusted by the user if so directed by an appropriate agency;

$LP_{ta,soil,i}$ (dimensionless) is the fresh mass terrestrial animal to soil concentration factor of nuclide *i*;

CF_{ta} (dimensionless) is the correction factor for area or organism residence time for the terrestrial organism. This correction factor is set at 1 for the general screening phase of the calculations; and

all other terms have been defined.

Water BCGs for Terrestrial Animals. The conceptual model for terrestrial animals is based on the entire animal being surrounded by soil. However, the potential for exposure to contaminated water from soil pore water or by drinking from contaminated ponds or rivers exists. Water presents both an internal and external dose hazard. As before, lumped parameters are used to estimate the extent of internal contamination (e.g., as might occur from ingestion). A semi-infinite exposure model is used for the external exposure. The method used to derive the terrestrial animal BCGs for exposure to a single nuclide in contaminated water is:

$$BCG(water)_{i,terrestrial\ animal} = \frac{365.25 \cdot DL_{ta}}{C_{ta} \cdot [(0.001 \cdot LP_{ta,water,i} \cdot DCF_{internal,i}) + (DCF_{ext,water,i})]}$$

Equation 10

where $BCG(water)_{i,\ terrestrial\ animal}$ (Bq m^{-3}) is the concentration of nuclide *i* in water which, based on the screening level assumptions, numerically equates to a dose rate of DL_{ta} (0.001 Gy d^{-1}) to the terrestrial animal;

$LP_{ta,\ water,\ i}$ (L/kg) is the fresh mass terrestrial animal to water concentration factor of nuclide *i*; and

all other factors have been defined.

How are these Dose Equations and their Parameters Used in Implementing the Graded Approach?

General Screening. The initial value of the "lumped parameter" (B_{iv}) used in the general screening phase is specifically chosen to produce conservative default BCGs. This quickly removes from further consideration contamination levels that would not cause biota to receive doses above acceptable limits. However, some sites may fail the general screen. This does not mean that they are causing biota to receive doses above the acceptable limit, but suggests that further analysis is warranted for specific radionuclides and media. It is recognized that actual B_{iv} values range over several orders of magnitude, depending upon biotic and abiotic features of the environment.

Site-Specific Screening. The next step is to examine the lumped parameter, and using data either directly from the site, or from the technical literature, select a value which is more representative for the specific-site conditions. In doing so, the screening calculation is repeated and a new site-specific BCG is provided. The process for each organism-type is as follows:

- **Aquatic Animals.** The user is allowed to modify the $B_{iv,aa,i}$ (the wet weight bioaccumulation factor) to a more site-representative value. All other aspects of the calculations remain the same.
- **Riparian Animals.** The user is allowed to modify the lumped parameter ($LP_{ra,water,I}$ and $LP_{ra,sed,I}$, the wet weight bioaccumulation factor for animal to water or animal to sediment) to a more site-representative value. All other aspects of the calculations remain the same.
- **Terrestrial Plants.** The user is allowed to modify the $B_{iv,tp,i}$ (the wet weight bioaccumulation factor) to a more site-representative value. All other aspects of the calculations remain the same.
- **Terrestrial Animals.** The user is allowed to modify the lumped parameter ($LP_{ta,water,I}$ and $LP_{ta,soil,I}$, the wet weight bioaccumulation factor for terrestrial animal to water or terrestrial animal to soil) to a more site-representative value. All other aspects of the calculations remain the same.

3.4 Alternatives to Lumped Parameters for Riparian and Terrestrial Animals: The Kinetic/Allometric Method

As discussed in Section 3.1, for most radionuclides, the single-most important predictor of biota dose is the method used to estimate internal tissue concentrations. The technical literature contains reference to these empirically based parameters that measure concentrations of contaminants in an organism relative to the surrounding media. These ratios are called "concentration ratios," "concentration factors," or "wet-weight concentration ratios" (B_{iv}s). These lumped parameters (e.g., B_{iv} values) are available for many nuclides for plant:soil and for aquatic species:water. In a few instances they are also available for animals:soil or animals:sediment. The advantage of using one of these factors is that it allows the prediction of tissue concentration based on simple measurements of contamination in environmental media

such as soil, water, or sediment. The use of lumped parameters is an integral feature of the screening approach. However, as the methodology evolved it became apparent that there were gaps in the data that needed to be addressed, particularly for riparian and terrestrial animal lumped parameters. An alternative approach, called the kinetic/allometric method, was developed. This method had two objectives: first, to fill in data gaps in the literature on lumped parameters; and second, to provide users with an alternative, more sophisticated method for evaluating dose to specific riparian and terrestrial animal receptors.

The kinetic/allometric method is applied in the site-specific analysis component of the graded approach. In site-specific analysis, the internal pathways of exposure are examined in greater detail. This evaluation relies upon mathematically modeling the exposure of the organism using simplistic, first-order kinetic reactions of the form:

$$q = \frac{R}{k}(1 - e^{-kt})$$

Equation 11

where q is the total activity (Bq) in the organism of concern at time t;

R is the intake rate of activity (Bq d^{-1}) into the organism;

k is the effective loss rate of activity (d^{-1}) from the organism; and

t is the total length of exposure to the contaminant (d).

The activity concentration in the animal is calculated as q divided by the mass; in SI units the mass would be expressed in kg. While this calculation method is simple, it still requires information on the intake rate of the organism, the total body mass, the loss rate of the radionuclide and the exposure period.

3.4.1 A Scaling Approach to Predicting Tissue Concentrations

The key to estimating body burdens in biota is an expression for intake that can account for potential change with size of the organism. There are several allometric equations which relate body size to many parameters, including ingestion rate, life span, inhalation rate, home range and more (West et al. 1997). These equations take the form of:

$$Y = \alpha X^{\beta}$$

Equation 12

where Y and X are size-related measures and α and β are constants.

While these equations were originally derived from empirical observations, there is a growing body of evidence that these relationships have their origins in the dynamics of energy transport mechanisms. An example of one use of this type of equation is illustrated in deriving soil BCGs for terrestrial animals.

3.4.1.1 Estimating Intake (Soil Pathway)

The intake of radioactivity into a terrestrial animal is presumed to come from three routes of exposure: ingestion of contaminated foodstuffs, ingestion of contaminated soil, and inhalation of re-suspended soil.

Ingestion of food. Metabolic rate is known to scale to body mass to the ¾ power (Calder 1984, Reiss 1989, and West et al. 1997). The food intake rate can also be calculated if allowances are made for several factors (Whicker and Shultz 1982):

$$r = \frac{a}{dc} 70 M^{0.75}$$

Equation 13

where r = food intake rate in g/day;

a = ratio of active or maintenance metabolic rate to the basal metabolic rate;

d = fraction of the energy ingested that is assimilated and oxidized;

c = caloric value of food in kcal /g; and

M = live body weight in kilograms.

The rate of radionuclide intake into the animal is a product of the food intake rate and the activity concentration of the foodstuff. The concentration of radionuclides in food is a product of the soil concentration (C_s, Bq/kg) and the food-to-soil uptake factor ($B_{iv,tp,i}$ dimensionless). The radionuclide intake rate via ingestion is expressed in Bq/d:

$$I_{ingestion,food,i} = C_{s,i} B_{iv,tp,i} \left[10^{-8} \frac{a}{dc} 70 M^{0.75} \right]$$

Equation 14

Where $I_{ingestion,food,i}$ is the intake rate (Bq d^{-1}) of a radionuclide into the animal via consumption of contaminated food, the concentration of radionuclides in the contaminated food is calculated as a product of the soil concentration and the food-to-soil (wet-weight) uptake

factor (B_{iv}), and the factor of 10^{-3} converts the ingestion rate of equation 13 from g d^{-1} to kg d^{-1}; and

all other terms have been defined.

Ingestion of soil. Studies on soil ingestion by wildlife indicate that it scales as a percentage of the mass of the daily diet (EPA 1993). The rate of radionuclide intake into the animal via soil ingestion (Bq d^{-1}) would therefore be the soil concentration times the daily mass of food ingested times the fraction of the daily diet that comes from soil ingestion (f).

$$I_{ingestion,soil,i} = C_{s,i} \; f \left[10^{-8} \frac{a}{dc} 70 M^{0.75} \right]$$

Equation 15

where f is the fraction of the mass of daily diet that comes from soil ingestion.

Inhalation of soil. The rate of intake of soil into the lungs of the animal can be calculated as the product of the inhalation rate (m^3 d^{-1}) and the air concentration (in Bq m^{-3}) of the nuclide.

The air concentration can be estimated using the mass loading approach. The activity in air is calculated as the product of X, the dust loading in air (in kg m^{-3}) and $C_{soil.}$ The lung ventilation rate also scales as a function of body mass (Pedley 1975 and West et al. 1997). Because of differences in solubility in body fluids, material taken into the body via inhalation may (or may not) be more readily absorbed than those taken in via ingestion. In his paper assessing the contribution of inhalation to dose, Zach (1985) derived a series of correction factors (PT/IT) which provided an adjustment for inhalation relative to ingestion. These factors are used to correct the inhalation rate to that of an equivalent amount of ingested soil:

$$I_{inhalation,normalized,i} = \frac{PT}{IT} \; X \; C_{s,i} \; 0.481 \; M^{0.76}$$

Equation 16

Calculating Total Intake. The total intake to the body can be calculated as the sum of inputs from inhalation given in equation 16, food ingestion in equation 14, and soil ingestion in equation 15. This is accomplished by direct substitution and rearrangement into the relationship $R = I_{inhalation} + I_{soil\ ingestion} + I_{food}$, as follows:

$$R = C_{soil} \left[(B_{iv} \cdot f) \left(10^{-8} \frac{a}{dc} 70M^{0.75} \right) + \frac{PT}{IT} X\ 0.481M^{0.76} \right]$$

Equation 17

Estimating the Fraction Assimilated into the Body. Because only a fraction of the material ingested actually enters into the blood, the total intake rate must be modified by a factor, f_1, to account for this difference:

$$R^{*} = f_1 R = f_1 C_{soil} \left[(B_{iv} \cdot f) \left(10^{-8} \frac{a}{dc} 70M^{0.75} \right) + \frac{PT}{IT} X\ 0.481M^{0.76} \right]$$

Equation 18

where R* is the species-independent estimate of radionuclide uptake to blood (Bq d^{-1}) from exposure to contaminated soil, and f_i is the fraction of intake assimilated to the body.

3.4.1.2 Estimating the Total Loss Rate from the Organism

The loss of radioactive material from the organism is due to radiological decay as well as biological elimination. There is substantial evidence that biological half-time of material in the body is related to metabolism, and therefore should be a function of body mass with the following relationship:

Equation 19

$$T_{1/2,biological,i} = \alpha W^{\beta}$$

where α and β are scaling constants related to the biological elimination of a particular element and W is the body mass (in g). In their book, Whicker and Schultz (1982) identified empirical relationships for five elements including Sr, Cs, I, Co, and ^{3}H. Three of these elements exhibited scaling to the ¼ power (Cs, Sr, Co). Iodine scaled at $W^{0.13}$ and ^{3}H scaled at $W^{0.55}$. The biological decay time is then used to calculate the biological decay constant (e.g, k in Equation 11). The effective decay constant, k_{eff} is calculated as the sum of the radiological and biological decay constants.

Scaling constants for other radionuclides were estimated from data provided in the literature on the biological elimination rates for various species of animals.

3.4.1.3 Calculating the Fractional Buildup to Equilibrium Tissue Concentrations

The activity in an organism continuously exposed to a constant source of contaminated material will, potentially, continue to increase until either a maximum value, or equilibrium, is attained. The degree of equilibrium that is attained is dictated by the lifespan of the organism, and the length of exposure, in conjunction with the effective loss-rate constant. For the purposes of radiological protection we need to know the maximum potential body burden in the organism. If exposure is constant throughout the life of the organism, then the time of maximum body burden will definitely occur when the exposure time equals maximum lifespan of the organism (for radionuclides with a short half-life or biological elimination rate, the time to reach maximum body burden will be substantially shorter). Using the lifespan of the organism to calculate tissue concentrations is the simplest approach.

In a manner similar to metabolic rate and inhalation rate, the maximum lifespan of an organism has been found to scale as a function of body mass. Calder (1984) analyzed the lifespan of 35 species of wild mammals to estimate their life expectancy (in the wild):

$$T_{expected,wild} = 1.02\ M^{0.30 \pm 0.026}$$

Equation 20

where $T_{expected,wild}$ is in years and M is the live weight in kg.

3.4.1.4 Calculating Species-Independent Tissue Concentrations from Soil Exposure

The activity in an organism continuously exposed to a constant source of contaminated material will, potentially, continue to increase until either a maximum value, or equilibrium, is attained. The degree of equilibrium that is attained is dictated by the lifespan of the organism, and the length of exposure, in conjunction with the effective loss-rate constant. If exposure is constant throughout the life of the organism, then the time of maximum body burden will occur when the exposure time equals the maximum lifespan of the organism (for radionuclides with a short half-life or biological elmininationrate, the time to reach maximum body burden will be substantially shorter). Equations 11, 13, 18, and 20 can be combined (with appropriate unit conversions) to provide an estimate of the maximal tissue concentration for the organism consuming contaminated plants, soil, and breathing contaminated air:

$$C_{animal\ soil} = \frac{f_1 C_{soil}\left[(B_{iv} f)\left(10^{-8}\frac{a}{dc}70M^{0.75}\right) + \frac{PT}{IT}X\ 0.481M^{0.76}\right]\left(1 - e^{-(k_{rad}+k_{bio})(365.25)(1.02M^{0.3})}\right)}{(k_{rad}+k_{bio})M}$$

Equation 21

3.4.1.5 Calculating Limiting Soil Concentrations (BCGs) Using the Kinetic/Allometric Method: An Example

Although predicting tissue concentrations of species exposed to contaminants is important, the overall purpose of this effort is to derive media concentrations that will be protective of biota at a site. The methodology can be demonstrated using the soil-terrestrial animal pathway. Equation 21 estimates the maximum potential tissue concentration in an animal from prolonged exposure to soil contaminated with radionuclide *i* at a unit concentration (e.g., 1 Bq/kg). If a particular dose limit is chosen (D_{ta} for example, in Gy/y), the limiting soil concentration to achieve that dose limit (LS_i) can be calculated as:

$$LS_i = \frac{D_{ta}}{C_{animal,i} DCF_{internal,i}}$$

Equation 22

where LS_i = limiting soil concentration in Bq/kg;

D_{ta} = chosen dose limit, in Gy/y;

C_{animal} = predicted tissue concentration of an animal from exposure to 1 Bq/kg contamination in soil; and

DCF = internal dose factor (Gy/y per Bq/kg of tissue).

The equation can be further modified to account for external exposure of the organism:

$$LS_i = \frac{D_{ta}}{C_{animal,i} DCF_{internal,i} + DCF_{ext,i}}$$

Equation 23

where $DCF_{ext,i}$ = external dose conversion factor (Gy/y per Bq/kg of soil); and all other factors have been defined.

Substitution of the tissue concentrations (Equation 21) into the equation for calculating limiting media concentrations results in the following equation:

$$LS_{terrestrial\ animal,i} = \frac{0.001 Gy \cdot d^{-1}}{\frac{f_1(\alpha + \beta)\delta DCF_{internal,i}}{K_{eff} M} + DCF_{ext,soil,i}}$$

Equation 24

where α provides an estimate of the daily intake rate of contaminated food and soil into the terrestrial animal;

$$\alpha = \frac{a}{dc} 70M^{0.75}(B_{iv,sp,i} \cdot f)$$

Equation 25

β provides the estimate of the daily intake that occurs through inhalation (and adjusts uptake relative to ingestion);

$$\beta = \frac{PT}{IT} X \ 0.481M^{0.76}$$

Equation 26

and δ provides an estimate of the exposure period, expressed as a function of the maximal life span of the target organism;

$$\delta = \left(1 - e^{-k_{eff} 1.02M^{0.30}}\right)$$

Equation 27

and all other terms have been previously defined.

3.4.2 Application of the Kinetic/Allometric Method in the Derivation of BCGs for Riparian Animals

In the analysis phase of the graded approach, a user may not have access to site-specific B_{iv}s or lumped parameters, or use of them results in exceeding site-specific screening. If that is the case, the user is allowed to conduct a more in-depth analysis of potential dose using the kinetic/allometric method. Equations have been developed for riparian animals using the methodology and equations discussed in Section 3.4.1. Two equations were developed, one for exposure to contaminated sediment, and a second for exposure to contaminated water.

Sediment. Riparian animal exposure to sediment considers external exposure as well as the inadvertent ingestion of sediment. The derivation of the sediment BCG for riparian animals is based on predicting maximal tissue concentrations after a lifetime of exposure**.** The equation used to derive the riparian BCGs for exposure to a single nuclide in contaminated sediment is:

$$BCG(sediment)_{i,riparian\ animal} = \frac{365.25 \cdot DL_{ra}}{CF_{ra}\left(\left[\frac{f_1 f\left[10^{-8}\frac{a}{dc}70M^{0.75}\right]\left(1-e^{-(k_{rad}+k_{bio})(365.25)\cdot 1.02M^{0.3}}\right)DCF_{internal,i}}{(k_{rad}+k_{bio})M}\right]+\left[DCF_{ext,sediment,i}\right]\right)}$$

Equation 28

Water. The equation used to derive the riparian BCGs for exposure to a single nuclide in contaminated water is similar but includes ingestion of contaminated foodstuff and water, as well as external exposure, and is based on predicting maximal tissue concentrations after a lifetime of exposure. Water consumption scales as a function of body mass (EPA 1993) in a manner similar to ingestion:

Equation 29

$$r_{water} = 0.099M^{0.90}$$

where r_{water} is in Ld^{-1} and all other terms have been defined.

The BCG is calculated as:

$$BCG(water)_{i,riparian\ animal} = \frac{365.25 \cdot DL_{ra}}{CF_{ra}\left[\left(0.001\cdot\frac{f_1\left[B_{iv,af}\left(10^{-8}\frac{a}{dc}70M^{0.75}\right)+0.099M^{0.90}\right]\left(1-e^{-(k_{rad}+k_{bio})(365.25)\cdot 1.02M^{0.3}}\right)(DCF_{internal,i})}{(k_{rad}+k_{bio})M}\right)+(DCF_{ext,water,i})\right]}$$

Equation 30

where $B_{iv,af}$ = aquatic foods bioconcentration factor and all other terms have been defined.

It should be noted that Equations 28 and 30 can be condensed to the simpler form of Equations 3 and 4 by substitution of a single lumped parameter constant for the organism-specific variables. Also, it is possible to use Equation 30 to assess impacts to either carnivorous or herbivorous riparian animals by substituting appropriate values of $B_{iv,aa}$ into this equation. This method is applicable to carnivores because the lumped parameters selected for the default case represent the upper-end values from the technical literature. These literature values encompass carnivores as well as herbivores. The bioconcentration factor ($B_{iv,aa}$) in Equation 30, when multiplied by the water concentration, provides a prediction of radionuclide concentration in the riparian animal's food. For herbivorous riparian animals, one can substitute B_{iv} values appropriate for aquatic plant:water in lieu of $B_{iv,aa}$ values for aquatic animals.

3.4.3 Application of the Kinetic/Allometric Method in the Derivation of BCGs for Terrestrial Animals

In a manner similar to that used for riparian animals, equations have been developed for terrestrial animals using the methodology and equations discussed in section 3.4.1.

Soil. The derivation of the soil BCG considers ingestion of contaminated foodstuff, and soil, inhalation of soil, and external exposure. It is based on predicting maximal tissue concentrations after a lifetime of exposure.

$$BCG(soil)_{i,terrestrial\ animal} = \frac{365.25 \cdot ED_{ta}}{CF_{ta}\left[\frac{f_1\left[(B_{iv} \cdot IR)\left(10^{-6}\frac{a}{dc}70M^{0.75}\right) + \left(\frac{PT}{IT}\right)0.481M^{0.76}\right]\left(1 - e^{-(k_{rad}+k_{bio})(365.25)\cdot 1.02^{0.3}}\right)(DCF_{internal,i})}{(k_{rad}+k_{bio})M} + (DCF_{ext,soil,i})\right]}$$

Equation 31

where all terms have been defined.

Water. The equation used to derive the terrestrial animal BCGs for exposure to a single nuclide in contaminated water is similar to that used for soil, but includes ingestion of contaminated water, as well as external exposure, and is based on predicting maximal tissue concentrations after a lifetime of exposure.

$$BCG(water)_{i,terrestrial\ animal} = \frac{365.35 \cdot ED_{ra}}{CF_{ta}\left[0.001\frac{f_1 0.099M^{0.90}\left(1 - e^{-(k_{rad}+k_{bio})(365.25)\cdot 1.02M^{0.3}}\right)(DCF_{internal,i})}{(k_{rad}+k_{bio})M} + (DCF_{ext,i})\right]}$$

Equation 32

where all terms have been defined.

It should be noted that Equations 31 and 32 could be condensed to the simpler form of Equations 9 and 10 by substitution of a single lumped parameter constant for the organism-specific variables. Also, it is possible to use Equation 31 to assess impacts to either carnivorous or herbivorous animals by substituting appropriate values of B_{iv} into this equation. The bioconcentration factor ($B_{iv,tp}$) in Equation 31, when multiplied by the soil concentration, provides a prediction of radionuclide concentration in the terrestrial animal's food. While B_{iv} values for animal:soil could be substituted, a more conservative approach is to use the existing ($B_{iv,tp}$) values provided for terrestrial plants. In this manner, biomagnification through higher trophic levels can be assessed.

3.5 Selection of Lumped Parameters for Riparian and Terrestrial Animals

Recall that the general screening phase of the graded approach utilizes lumped parameters to provide estimates of organism tissue concentration, and ultimately derive the nuclide, media, and organism–specific BCGs. While there is a relative abundance of data for aquatic animals and terrestrial plants, less information is found for terrestrial and riparian animals.

As noted in Sections 3.4.2 and 3.4.3, the kinetic/allometric equations can be condensed to a simpler form by substitution of a single lumped parameter in place of the organism-specific variables. The choice of a value for this lumped parameter becomes problematic, however, when considering the range of organism types meant to be covered by the method. Also, there is very limited data available in the literature on animal:water, animal:soil, and animal:sediment ratios. Two alternative approaches were evaluated:

Calculating Lumped Parameters by Multiplying Related Concentration Ratios (Product Approach). It is possible to calculate the lumped parameters by multiplying related concentration ratios; for example, the product of plant:soil and animal:plant concentration ratios yields a animal:soil ratio which may be substituted for the lumped parameter used in Equation 9. This approach must be used with caution, as the data used in the process are most likely from different sources. This approach also is hampered by the lack of environmental data.

Calculating Lumped Parameters by Using Uncertainty Analysis on the Kinetic/Allometric Method. An alternative method to developing lumped parameters for riparian and terrestrial animals was addressed by using uncertainty analysis on the kinetic/allometric method. A Monte-Carlo simulation was used to determine the effect of parameter variability on the calculation of maximal animal tissue concentrations relative to environmental media concentrations. The allometric equations shown for riparian and terrestrial animals in Section 3.4.2 and 3.4.3, respectively, were rearranged to predict lumped parameters resulting from exposure to a unit concentration of contaminant in water, sediment, or soil. The rearranged equations are shown below. Each of the variables has been previously defined.

$$LP(sediment)_{i,riparian\ animal} = \frac{C_{riparian\ animal\ sediment}}{C_{sediment}} \cdot \frac{f_1 f \left[10^{-8} \frac{a}{dc} 70M^{0.75}\right]\left(1 - e^{-(k_{rad}+k_{bio})(365.25)(1.02M^{0.3})}\right)}{(k_{rad}+k_{bio})M}$$

Equation 33

$$LP(water)_{i,riparian\ animal} = \frac{C_{i,riparian\ animal}}{C_{water}} \cdot \frac{f_1\left[B_{iv\ af}\left(10^{-8}\frac{a}{dc}70M^{0.75}\right) + 0.099M^{0.9}\right]\left(1 - e^{-(k_{rad}+k_{bio})(365.25)(1.02M^{0.3})}\right)}{(k_{rad}+k_{bio})M}$$

Equation 34

$$LP(soil)_{i,terrestrial\ animal} = \frac{C_{animal\ soil}}{C_{soil}} \cdot \frac{f_1\left[(B_{iv}+f)\left(10^{-8}\frac{a}{dc}70M^{0.75}\right) + \frac{PT}{IT}X\ 0.481M^{0.76}\right]\left(1 - e^{-(k_{rad}+k_{bio})(365.25)(1.02M^{0.3})}\right)}{(k_{rad}+k_{bio})M}$$

Equation 35

$$LP(water)_{i,riparian\ animal} = \frac{C_{animal,\ water}}{C_{water}} \cdot \frac{f_1\ 0.099M^{0.90}\left(1 - e^{-(k_{rad}+k_{bio})(365.25)(1.02M^{0.3})}\right)}{(k_{rad}+k_{bio})M}$$

Equation 36

A Monte Carlo uncertainty analysis was conducted on each equation, with parameters varied over their known ranges. The range of values assigned each variable used in the uncertainty analysis was taken from the technical literature. These values, and their accompanying distributions, are shown in Table 3.1.

Ten thousand simulations were run for each equation and nuclide. Results were generated for twenty-three radionuclides, and the 95th percentile value for each was compared with data (where it existed) from the technical literature. The results are tabulated in Table 3.2 (A-D). Based on analysis, the model predictions tracked reasonably well with the values observed in the scientific literature. The lumped parameter value selected (from a choice of available empirical data, product approach, and uncertainty analysis on the kinetic/allometric method) for use as the default lumped parameter for use in general screening is highlighted in each table. The preference was to use empirical data where available and of good quality, as was the case for many terrestrial animal:soil values. However, as previously discussed, data for riparian and terrestrial animals was generally limited. In most instances, the kinetic/allometric result was chosen over values taken from the technical literature. Generally, the kinetic/allometric calculation resulted in a higher estimate of the lumped parameter. This is expected, owing to the generally conservative nature of parameter values used in the kinetic/allometric method.

Table 3.1 Parameters Used in Kinetic/Allometric Method Uncertainty Analysis for Riparian and Terrestrial Animals

Equation and Parameter	Mean	Range (and distribution)[a]
Riparian animal: sediment and water lumped parameter assessment		
$r_{ra} = \frac{a}{dc} 70M^{b}$ r_{ra} = food intake rate in g/day		
$r_{ra,sediment} = \frac{a}{dc} 70M^{b} f$ $r_{ra,sediment}$ = sediment intake rate in g/day;		
a, ratio of active to maintenance metabolic rate (see equation 13)	2	0.5-3.0 (normal)
d, fraction of energy ingested that is assimilated (see equation 13)	0.65	0.3-0.9 (normal)
c, caloric value of food intake (see equation 13)	5	4 – 9 (normal)
b, exponent in allometric relationship detailing consumption as a function of body mass (see equation 13)	0.75	0.68-0.8 (normal)
f, fraction of diet that is soil (see equation 15)	0.1	0.01-0.55 (normal)
M, body mass in kilograms	1 kg	0.02 – 6000 (log normal)
$T_{ls} = 1.02\ M^{0.30}$ T_{ls} = maximum lifespan of the organism, years		
exponent (0.30), allometric relationship detailing lifespan as a function of body mass (see equation 20)	0.3	0.25 – 0.33 (normal)
constant (1.02), allometric relationship, detailing lifespan as a function of body mass (equation 20)	1.02	0.9 – 2.00 (normal)
$\lambda_{bio,i} = \frac{0.69315}{aM^{b}}$ $\lambda_{bio,i}$= biological decay constant of material in organism, per day		
b, exponent, allometric relationship detailing biological half-time as a function of body mass (equation 19)	Varies by nuclide 0.24 for Cs	0.15 – 0.3 (normal)
a, constant, allometric relationship, detailing biological half-time as a function of body mass (equation 19)	Varies by nuclide 3.5 for Cs	2 - 5 (normal)
$I_w = 0.099\ M^{0.9}$ I_w =water intake, L/d		
constant, allometric relationship, detailing water intake rate I_w(l/d) as a function of body mass, where $I_w = 0.099W^{0.90}$	0.099	0.07 - 0.13 (normal)
exponent, allometric relationship, detailing water intake rate as a function of body mass where $I_w = 0.099W^{0.90}$	0.9	0.63 - 1.17 (normal)

Table 3.1 (Continued) Parameters Used in Kinetic/Allometric Method Uncertainty Analysis for Riparian and Terrestrial Animals

Equation and Parameter	Mean	Range (and distribution)[a]
Terrestrial animal: soil and water lumped parameter assessment		
$r_{inhale,i} \bullet\bullet 0.481\ M^{0.76}$	$r_{iinhale,\ i}$ = inhalation rate of soil	
exponent (0.76), allometric relationship detailing inhalation rate as a function of body mass (equation 16)	0.76	0.64-0.86 (normal)
X Dust loading (equation 16)	0.001	0.0001 – 0.01 (log normal)
constant (0.481), allometric relationship, detailing inhalation rate as a function of body mass (equation 16)	0.481	0.001 – 0.66 (normal)
$r_{ta,soil} = r_{ra,sed}$ $r_{ta} = r_{ra}$ all other factors have been defined.	Varies	Varies
[a]The distributions used in this assessment were created by examination of the range of values of the input variables and, where possible, by testing using the forecasting and risk analysis software, Crystal Ball®.		

Table 3.2A A Comparison of Lumped Parameter Values Determined by Uncertainty Analysis on the Kinetic/Allometric Method, Product Approach, and Empirical Data (Literature Values): Riparian Animal to Sediment[a]

Element	Calculated as Product of Concentration Ratios (CR)	Animal to Sediment Value Kinetic/Allometric Method		Empirically Measured Lumped Parameter
		50th percentile	95th percentile	
Am	5.4E-05	3.6E-04	3.1E-03	1.4E-04
Ce	3.9E-02	1.5E-04	4.8E-04	
Cs	4.4E-01	1.2E-01	2.7E-01	
Co	4.0E-02	4.3E-03	1.0E-02	4.5E-01
Eu		5.9E-04	3.9E-03	
H	6.0E-01	1.2E-01	4.3E-01	
I	1.1E+00	1.3E-01	3.2E-01	
Pu	3.0E-06	3.6E-04	3.2E-03	5.0E-05
Ra	3.0E-02	1.4E-02	3.0E-02	
Sb	1.8E-03	1.8E-04	4.1E-04	
Sr	3.6E-01	1.1E+00	2.0E+00	
Tc	1.2E-02	1.7E-02	4.6E-02	
Th	2.4E-07	2.9E-04	1.9E-03	
U	1.0E-01	1.6E-03	3.8E-03	1.0E-03
Zn		7.2E-01	1.8E+00	
Zr	6.4E-03	1.1E-03	3.0E-03	

[a]The shaded cell indicates this value is used as the default lumped parameter in the general screening phase of the graded approach. Blank cells indicate data was unavailable.

Table 3.2B A Comparison of Lumped Parameter Values Determined by Uncertainty Analysis on the Kinetic/Allometric Method, Product Approach, and Empirical Data (Literature Values): Riparian Animal to Water[a]

Element	Calculated as Product of Concentration Ratios (CR)	Animal to Water Value Kinetic/Allometric Method		Empirically Measured Lumped Parameter
		50th Percentile	95th Percentile	
Am	2.2E-02	1.4E+00	1.2E+01	
Ce	3.9E+01	1.4E+01	3.5E+01	
Cs	1.5E+05	2.6E+04	4.7E+04	2.5E+05
Co	1.0E+03	8.6E+01	1.6E+02	9.0E+02[b]
Eu		3.6E+00	2.0E+01	
H	1.2E-01	2.4E-01	8.1E-01	
I	1.1E+02	2.9E+02	5.7E+02	2.1E+02
Pu	1.5E-02	3.6E+00	3.0E+01	6.7E+00
Ra	3.2E+01	4.6E+02	8.0E+02	
Sb	1.8E+00	1.7E-01	3.1E-01	
Sr	1.4E+03	3.5E+03	6.2E+03	9.0E+03[b]
Tc	1.0E+01	1.4E+01	2.9E+01	
Th	2.4E-01	2.4E-01	1.5E+00	
U	5.1E+00	1.6E+01	3.0E+01	
Zn		1.2E+05	2.5E+05	
Zr	5.0E+02	1.8E+01	4.0E+01	

[a] The shaded cell indicates this value is used as the default lumped parameter in the general screening phase of the graded approach. Blank cells indicate data was unavailable.

[b] These values are not directly measured lumped parameters but were derived from other parameters.

Table 3.2C A Comparison of Lumped Parameter Values Determined by Uncertainty Analysis on the Kinetic/Allometric Method, Product Approach, and Empirical Data (Literature Values): Terrestrial Animal to Soil[a]

Element	Calculated as Product of Concentration Ratios (CR)	Animal to Soil Value Kinetic/Allometric Method		Empirically Measured Lumped Parameter
		50th Percentile	95th Percentile	
Am	4.1E-07	3.7E-04	4.0E-03	1.0E-04
Ce	1.7E-04	2.0E-04	5.5E-04	5.5E-03
Cs	6.7E+01	1.1E+01	2.0E+01	1.0E+02
Co	1.1E-01	1.0E-02	3.0E-02	8.0E-02
Eu		7.9E-04	4.6E-03	
H	6.6E-01	1.3E+00	4.3E+00[b]	
I	2.0E-01	6.8E-01	1.4E+00	3.0E+00
Pu	2.2E-07	4.1E-04	3.0E-03	3.0E-03
Ra	1.1E-03	3.0E-02	6.0E-02	2.1E-01
Sb	1.8E-04	1.9E-04	4.3E-04	
Sr	1.7E+01	4.2E+01	7.6E+01	6.1E-01
Tc	1.0E+00	1.4E+00	3.1E+00	
Th	3.1E-06	2.9E-04	1.6E-03	1.0E-03
U	1.9E-05	1.7E-03	4.1E-03	1.0E-03
Zn		3.3E+00	7.0E+00	1.0E-02
Zr	9.1E-03	1.4E-03	3.5E-03	

[a]The shaded cell indicates this value is used as the default lumped parameter in the general screening phase of the graded approach. Blank cells indicate data was unavailable.

[b] The H lumped parameter value was set at a default of 1.0 for calculation of the generic BCG.

Table 3.2D A Comparison of Lumped Parameter Values Determined by Uncertainty Analysis on the Kinetic/Allometric Method, Product Approach, and Empirical Data (Literature Values): Terrestrial Animal to Water[a]

Element	Calculated as Product of Concentration Ratios (CR)	Animal to Water Value Kinetic/Allometric Method		Empirically Measured Lumped Parameter
		50th Percentile	95th Percentile	
Am		5.6E-03	8.6E-02	
Ce		2.4E-03	8.2E-03	
Cs	1.1E+01	2.0E+00	3.4E+00	
Co	7.9E-01	7.5E-02	1.3E-01	
Eu		9.2E-03	9.7E-02	
H		1.9E+00	1.7E+01[b]	
I		2.2E+00	3.4E+00	5.4E+00
Pu	1.5E-05	5.6E-03	9.3E-02	
Ra	1.8E+01	2.4E-01	4.0E-01	
Sb		3.0E-03	5.2E-03	
Sr	6.4E+02	1.8E+01	3.1E+01	
Tc		2.7E-01	8.4E-01	
Th		4.6E-03	4.5E-02	
U	1.9E-04	3.0E-02	5.0E-02	1.0E-03
Zn		3.7E+00	2.0E+01	1.0E-02
Zr	9.1E-03	1.8E-02	3.1E-02	

[a]The shaded cell indicates this value is used as the default lumped parameter in the general screening phase of the graded approach. Blank cells indicate data was unavailable.

[b] The H lumped parameter value was set at a default of 1.0 for calculation of the generic BCG.

4 Default Parameters and Their Sources

The following sections describe the source of parameter values used in the derivation of BCGs for aquatic animals, riparian animals, terrestrial plants, and terrestrial animals.

4.1 Bioaccumulation Factors (B_{iv}s)

The $B_{iv,aa,i}$ values for aquatic animals were selected from across all sampled aquatic taxa and include predatory fin fish, crustaceans, and other organisms. Typically the most limiting values come from crustaceans or molluscs. The specific source of default values used for the general screening phase of the graded approach for aquatic animal evaluations is shown in Table 4.1. Table 4.2 provides the values used for the general screening phase in the derivation of terrestrial plant BCGs.

Table 4.1 Default Bioaccumulation Factors (B_{iv}s) for Aquatic Animals

Radionuclide	$B_{iv,aa,i}$ Organism to Water (L/kg) fresh mass	Water $B_{iv,aa,i}$ Reference	Comment
^{241}Am	400	CRITR	Value for fresh water molluscs taken from CRITbiog.dat (generic bioaccumulation: 2000) and converted to wet weight basis by dividing by 5 (an arbitrary dry to wet weight conversion). Conversation with D Strenge and B Napier indicated the GENII and CRITR values are dry-weight basis.
^{144}Ce	9000	T&M T. 5.41	Maximum fresh weight value for molluscs.
^{135}Cs	22000	T&M T. 5.41	Maximum value for crustaceans, fresh weight, for ^{133}Cs, ^{134}Cs, ^{137}Cs.
^{137}Cs	22000	T&M T. 5.41	Maximum value for crustaceans, fresh weight, for ^{133}Cs, ^{134}Cs, ^{137}Cs.
^{60}Co	2000	T&M T. 5.41	Maximum fresh weight value for molluscs.
^{154}Eu	600	GENII	Value for fresh water molluscs taken from BIOAC1.dat (generic bioaccumulation: 3000) and converted to wet weight basis by dividing by 5 (an arbitrary dry to wet weight conversion). Conversation with D Strenge and B Napier indicated the GENII and CRITR values are dry-weight basis.
^{155}Eu	600	GENII	Value for fresh water molluscs taken from BIOAC1.dat (generic bioaccumulation: 3000) and converted to wet weight basis by dividing by 5 (an arbitrary dry to wet weight conversion). Conversation with D Strenge and B Napier indicated the GENII and CRITR values are dry-weight basis.
^{3}H	0.2	CRITR	Value for fresh water molluscs taken from CRITbiog.dat (generic bioaccumulation: 1) and converted to wet weight basis by dividing by 5 (an arbitrary dry to wet weight conversion). Conversation with D Strenge and B Napier indicated the GENII and CRITR values are dry-weight basis.
^{129}I	220	T&M T. 5.41	Maximum fresh weight value for molluscs.
^{131}I	220	T&M T. 5.41	Maximum fresh weight value for molluscs.
^{239}Pu	1000	T&M T. 5.41	Maximum fresh weight value for crustaceans.
^{226}Ra	3200	T&M T. 5.41	Freshwater gammarus.
^{228}Ra	3200	Ra-226	Freshwater gammarus.
^{125}Sb	100	T&M T. 5.41	Maximum fresh weight value for fish.
^{90}Sr	320	T&M T. 5.41	Maximum fresh weight value for molluscs.
^{99}Tc	78	T&M T. 5.41	Maximum fresh weight value for fish.
^{232}Th	80	T&M T. 5.41	Maximum fresh weight value for fish.
^{233}U	1000	T&M T. 5.41	Maximum fresh weight value for molluscs.
^{234}U	1000	T&M T. 5.41	Maximum fresh weight value for molluscs.
^{235}U	1000	T&M T. 5.41	Maximum fresh weight value for molluscs.
^{238}U	1000	T&M T. 5.41	Maximum fresh weight value for molluscs.
^{65}Zn	17000	T&M T. 5.41	Maximum fresh weight values for snails.
^{95}Zr	1600	T&M T. 5.41	Maximum fresh weight values for snails.

Table 4.2 Default Bioaccumulation Factors (B_{iv}s) for Terrestrial Plants

Radionuclide	$B_{iv,tp,i}$, Plant to Soil Bq/kg wet weight to Bq/kg soil (dry) mass	Plant $B_{iv,tp,i}$ Reference, Bq/kg plant (wet weight) per Bq/kg soil	Comment
^{241}Am	8.0E-03	T&M T5.16, T 5.18	Calculated from a CR value of 0.042 (dry wt/dry wt) for grasses. Converted to Biv using wet/dry ratio of 5.5. Note this also includes aerial deposition.
^{144}Ce	4.0E-02	T&M T5.16, T 5.17	Converted from a CR value (0.22 -dry wt/dry wt) for grasses in a soil with low pH content (<5.5). Converted to Biv using wet/dry ratio of 5.5
^{135}Cs	1.0E+01	T&M T5.16, T 5.17	Calculated from a CR value (42.6 - dry wt/dry wt) for legumes in Florida soils with low K content. Converted to Biv using wet/dry ratio of 4.5
^{137}Cs	1.0E+01	T&M, T5.16, T 5.17	Calculated from a CR value (42.6 - dry wt/dry wt) for legumes in Florida soils with low K content. Converted to Biv using wet/dry ratio of 4.5
^{60}Co	2.0E-01	T&M T5.16, T 5.17	Calculated from a CR value of 1 (dry wt/dry wt) for grasses in histosol soils. Converted to Biv using wet/dry ratio of 4.5
^{154}Eu	4.0E-02	Estimated from Ce value by KAH	
^{155}Eu	4.0E-02	Estimated from Ce value by KAH	
^{3}H	1.0E+00	NUREG 1.109	NUREG 1.109 and divided by a wet to dry conversion value of 4.5
^{129}I	4.0E-01	T&M T5.16, T 5.17	Calculated from a CR value of 1.84 (dry wt/dry wt) for legumes. Converted to Biv using wet/dry ratio of 4.5. Note this also includes aerial deposition.
^{131}I	4.0E-01	T&M T5.16, T 5.17	Calculated from a CR value of 1.84 (dry wt/dry wt) for legumes. Converted to Biv using wet/dry ratio of 4.5. Note this also includes aerial deposition.
^{239}Pu	1.0E-02	T&M T5.16, T 5.18	Calculated from a CR value of 0.066 (dry wt/dry wt) for legumes. Converted to Biv using wet/dry ratio of 4.5. Note this also includes aerial deposition.
^{226}Ra	1.0E-01	T&M T5.16, T 5.18	Calculated from a CR value of 0.49 (dry wt/dry wt) for legumes. Converted to Biv using wet/dry ratio of 4.5. Note this also includes aerial deposition.

Table 4.2 (Continued) Default Bioaccumulation Factors (B_{iv}s) for Terrestrial Plants

Radionuclide	$B_{iv,tp,i}$, Plant to Soil Bq/kg wet weight to Bq/kg soil (dry) mass	Plant $B_{iv,tp,i}$ Reference, Bq/kg plant (wet weight) per Bq/kg soil	Comment
^{228}Ra	1.0E-01	T&M, T5.16, T 5.18	Calculated from a CR value of 0.49 (dry wt/dry wt) for legumes. Converted to Biv using wet/dry ratio of 4.5. Note this also includes aerial deposition.
^{125}Sb	1.0E-02	GENII	Taken from GENII and converted to wet weight basis by dividing by 5 (an arbitrary wet to dry weight conversion). Conversation with D Strenge and B Napier indicated the GENII and CRITR ftrans values are on a dry-weight basis.
^{90}Sr	4.0E+00	T&M T5.16, T 5.17	Converted from a CR value (17.3 -dry wt/dry wt) for legumes in a soil with low Ca content. Converted to Biv using wet/dry ratio of 4.5
^{99}Tc	8.0E+00	GENII	Taken from GENII and converted to wet weight basis by dividing by 5 (an arbitrary wet to dry weight conversion). Conversation with D Strenge and B Napier indicated the GENII and CRITR ftrans values are on a dry-weight basis.
^{232}Th	1.0E-03	T&M T5.16, T 5.18	Calculated from a CR value of 0.0046 (dry wt/dry wt) for legumes. Converted to Biv using wet/dry ratio of 4.5. Note this also includes aerial deposition.
^{233}U	4.0E-03	T&M T5.16 T 5.18	Calculated from a CR value of 0.017 (dry wt/dry wt) for legumes. Converted to Biv using wet/dry ratio of 4.5. Note this also includes aerial deposition.
^{234}U	4.0E-03	Ti&M T5.16, T 5.18	Calculated from a CR value of 0.017 (dry wt/dry wt) for legumes. Converted to Biv using wet/dry ratio of 4.5. Note this also includes aerial deposition.
^{235}U	4.0E-03	T&M T5.16, T 5.18	Calculated from a CR value of 0.017 (dry wt/dry wt) for legumes. Converted to Biv using wet/dry ratio of 4.5. Note this also includes aerial deposition.
^{238}U	4.0E-03	T&M T5.16, T 5.18	Calculated from a CR value of 0.017 (dry wt/dry wt) for legumes. Converted to Biv using wet/dry ratio of 4.5. Note this also includes aerial deposition.
^{65}Zn	3.0E-01	T&M T5.16, T 5.17	Calculated from a CR value of 1.5 (dry wt/dry wt) for legumes. Converted to Biv using wet/dry ratio of 4.5. This value includes external (aerial) deposition in the value.
^{95}Zr	3.0E-02	T&M T5.16, T 5.17	Calculated from a CR value of 0.13 (dry wt/dry wt) for legumes. Converted to Biv using wet/dry ratio of 4.5.

4.2 Distribution Coefficients

Distribution coefficients describe the partitioning of a radionuclide between water and soil or sediment. Denoted by the variable K_d, these parameters were used in the absence of water (or sediment) data to estimate the missing radionuclide concentration data. Specific instructions on the use of this parameter are provided in Module 3, Section 3.2.3.

Table 4.3 Most Probable K_d Values for Use in Calculating BCGs for Sediment or Water for an Aquatic System Evaluation in the Absence of Co-Located Water and Sediment Data

Radionuclide	$K_{d,mp}$ Most Probable Values L/kg (mL/g)	Reference
^{241}Am	5.0E+03	Table 3.2, Till & Meyer, Median value for fresh water systems.
^{144}Ce	1.0E+03	RESRAD, Table E.3 page 202, "Manual for Implementing Residual Radioactive Material Guidelines Using RESRAD, Version 5.0" ANL/EAD/LD-2
^{135}Cs	5.0E+02	“”
^{137}Cs	5.0E+02	“”
^{60}Co	1.0E+03	“”
^{154}Eu	5.0E+02	Table 3.2, Till & Meyer, Median value for fresh water systems.
^{155}Eu	5.0E+02	“”
^{3}H	1.0E-03	Estimated by Higley
^{129}I	1.0E+01	Table 3.2, Till & Meyer, Median value for fresh water systems.
^{131}I	1.0E+01	“”
^{239}Pu	2.0E+03	Value taken from RESRAD, Table E.3 page 202, "Manual for Implementing Residual Radioactive Material Guidelines Using RESRAD, Version 5.0" ANL/EAD/LD-2.
^{226}Ra	7.0E+01	“”
^{228}Ra	7.0E+01	“”
^{125}Sb	1.0E+00	“”
^{90}Sr	3.0E+01	Value taken from RESRAD, Table E.3 page 202, "Manual for Implementing Residual Radioactive Material Guidelines Using RESRAD, Version 5.0" ANL/EAD/LD-2.
^{99}Tc	5.0E+00	Table 3.2, Till & Meyer, Median value for fresh water systems.
^{232}Th	6.0E+04	Value taken from RESRAD, Table E.3 page 202, "Manual for Implementing Residual Radioactive Material Guidelines Using RESRAD, Version 5.0" ANL/EAD/LD-2.
^{233}U	5.0E+01	“”
^{234}U	5.0E+01	“”
^{235}U	5.0E+01	“”
^{238}U	5.0E+01	“”
^{65}Zn	2.0E+01	“”
^{95}Zr	1.0E+03	“”

4.3 Coefficients Used in the Kinetic/Allometric Method

The following tables list the values of kinetic/allometric coefficients used in the derivation of lumped parameters using the kinetic/allometric method.

Table 4.4 Source of Default f_1 Values Used for Riparian and Terrestrial Animals

Radionuclide	f_1, (unitless)	Comment
^{241}Am	1.0E-03	ICRP 30 part 4 values for human and animal studies.
^{144}Ce	3.0E-04	ICRP 30 part 1 values for human and animal studies.
^{135}Cs	1.0E+00	ICRP 30 part 1 values for human and animal studies.
^{137}Cs	1.0E+00	ICRP 30 part 1 values for human and animal studies.
^{60}Co	5.0E-02	ICRP 30 part 1 values for human and animal studies.
^{154}Eu	1.0E-03	ICRP 30 Part 3 values.
^{155}Eu	1.0E-03	ICRP 30 Part 3 values.
^{3}H	1.0E+00	ICRP 30 part 1 values for human and animal studies.
^{129}I	1.0E+00	ICRP 30 Part 1 values.
^{131}I	1.0E+00	ICRP 30 Part 1 values.
^{239}Pu	1.0E-03	ICRP 30 part 4 values for human and animal studies.
^{226}Ra	2.0E-01	ICRP 30 part 1 values for human and animal studies.
^{228}Ra	2.0E-01	ICRP 30 part 1 values for human and animal studies.
^{125}Sb	1.0E-02	ICRP 30 part 3 values for human and animal studies.
^{90}Sr	3.0E-01	ICRP 30 part 1 values for human and animal studies.
^{99}Tc	8.0E-01	ICRP 30 part 2 values for human and animal studies.
^{232}Th-	2.0E-04	ICRP 30 part 1 values for human and animal studies.
^{233}U	5.0E-02	ICRP 30 part 1 values for human and animal studies.
^{234}U	5.0E-02	ICRP 30 part 1 values for human and animal studies.
^{235}U	5.0E-02	ICRP 30 part 1 values for human and animal studies.
^{238}U	5.0E-02	ICRP 30 part 1 values for human and animal studies.
^{65}Zn	5.0E-01	ICRP 30 part 2 values for human and animal studies.
^{95}Zr	2.0E-03	ICRP 30 part 1 values for human and animal studies.

Table 4.5 Source of Data Used in Estimating Biological Half-Times for Riparian and Terrestrial Animals (see Equation 19, Section 3.4.1.2)

Radionuclide	α (constant)	β (exponent)	Reference
^{241}Am	0.8	0.81	ICRP 30 Part 4
^{144}Ce	1.4	0.8	ICRP 30 Part 1
^{135}Cs	3.5	0.24	Whicker & Schultz
^{137}Cs	3.5	0.24	Whicker & Schultz
^{60}Co	2.6	0.24	Whicker & Schultz
^{154}Eu	1.4	0.8	ICRP 30 Part 3
^{155}Eu	1.4	0.8	ICRP 30 Part 3
^{3}H	0.82	0.55	Whicker & Schultz
^{129}I	6.8	0.13	Whicker & Schultz
^{131}I	6.8	0.13	Whicker & Schultz
^{239}Pu	0.8	0.81	ICRP 30 Part 4
^{226}Ra	2	0.25	Estimated by KAH
^{228}Ra	2	0.25	Estimated by KAH
^{125}Sb	0.5	0.25	ICRP 30 Part 3
^{90}Sr	107	0.26	Whicker & Schultz
^{99}Tc	0.3	0.4	ICRP 30 Part 2
^{232}Th	3.3	0.81	ICRP 30 Part 1
^{233}U	0.8	0.28	ICRP 30 Part 1
^{234}U	0.8	0.28	ICRP 30 Part 1
^{235}U	0.8	0.28	ICRP 30 Part 1
^{238}U	0.8	0.28	ICRP 30 Part 1
^{65}Zn	100	0.25	ICRP 30 Part 2
^{95}Zr	100	0.25	ICRP 30 Part 1

Table 4.6 Factors Used in Assessing the Relative Contribution to Internal Dose from Animal Inhalation versus Ingestion

Radionuclide	PT/IT[a] (Correction Factor)
^{241}Am	250
^{144}Ce	16
^{135}Cs	0.8
^{137}Cs	0.8
^{60}Co	7
^{154}Eu	30
^{155}Eu	30
^{3}H	1
^{129}I	0.7
^{131}I	0.7
^{239}Pu	4000
^{226}Ra	3
^{228}Ra	3
^{125}Sb	3.5
^{90}Sr	200
^{99}Tc	5
^{232}Th	750
^{233}U	7000
^{234}U	7000
^{235}U	3500
^{238}U	4000
^{65}Zn	1
^{95}Zr	10

[a] Based on ICRP 30, parts 1-3 and Zach's (1985) analysis of the relative contribution of inhalation to an equivalent amount of soil ingestion dose for animals.

Table 4.7 Allometric Equations and Parameter Values Used in Estimating Intake of Riparian Animal Organisms

Parameter	Equation	Descriptions	Value(s)	Reference
W		Body mass (g)	8800	default for raccoon or river otter
r	$r = \frac{a}{dc}\ 70\ M^{0.75}$	Food intake rate (g/d)	325.1377223	W&S, Vol. II, p. 43, equation 78
		a: ratio of active to basal metabolic rate	2	
		70: constant	70	
		d: fraction of energy ingested that is assimilated or oxidized	0.44	
		c: caloric value of food, kcal/g	5	
		M: body mass in kg	8.8	
		0.75: exponent in calculation	0.75	
$r_{sediment}$	$r_{sediment} = 0.1\ r$	Sediment Intake Rate (g/d)	32.51377223	EPA Wildlife Exposure Factor Handbook, Vol. 1, p. 4-22
		r: food intake rate, g/d	325.1377223	
		0.1: fraction of sediment in diet, expressed as % of food diet, dry	0.1	
T_{ls}	$T_{ls,\max} = 1.02M^{0.30}$	Maximum Lifespan	1.958	Calder, p. 316, Table 11-5
		1.02: constant in equation	1.02	
		see above equation, M: body mass in kg	8.8	
		0.30: exponent in calculation	0.30	
r_b	$r_b = 0.481M^{0.76}$	Inhalation rate (m^3/d)	2.511608286	Pedley, p. 15, Table V., adjusted to provide units of m^3/d
		0.481: constant in calculation to give m^3/d	0.481	
		see above equation, M: body mass in kg	8.8	
		0.76: exponent in equation	0.76	
$r_{inhalation}$	$r_{inhalation} = xr_b$	Sediment inhalation rate (g/d)	0.000251161	derived
		x: airborne dust loading, g/m^3	0.0001	
		r_b: inhalation rate (see above)	2.511608286	
I_w	$I_w = 0.099M^{0.90}$	Water consumption rate (L/d)	0.700921852	EPA Wildlife Exposure Factor Handbook, Vol. 1, p. 3-10, equation 3-17
		0.099: constant in equation	0.099	
		see above equation, M: body mass in kg	8.8	
		0.9: exponent in calculation	0.9	

Table 4.8 Allometric Equations and Parameter Values used in Estimating Intake of Terrestrial Animal Organisms

Parameter	Equation	Descriptions	Value(s)	Reference
W		Body mass (g)	22	default for deer mouse
r	$r = \frac{a}{dc}\ 70\ M^{0.75}$	Food intake rate (g/d)	3.635150245	W&S, Vol. II, p. 43, equation 78
		a: ratio of active to basal metabolic rate	2	
		70: constant	70	
		d: fraction of energy ingested that is assimilated or oxidized	0.44	
		c: caloric value of food, kcal/g	5	
		M: body mass in kg (=W*0.001)	0.022	
		0.75: exponent in calculation	0.75	
$r_{sediment}$	$r_{soil} = 0.1\ r$	Soil Intake Rate (g/d)	0.363515025	EPA Wildlife Exposure Factor Handbook, Vol. 1, p. 4-22
		r: food intake rate, g/d	3.635150245	
		0.1: fraction of sediment in diet, expressed as % of food diet, dry	0.1	
T_{ls}	$T_{ls,\max} = 1.02M^{0.3}$	Maximum Lifespan	.32	Calder, p. 316, Table 11-5
		1.02: constant in equation	1.02	
		see above equation, M: body mass in kg	0.022	
		0.30: exponent in calculation	0.30	
r_b	$r_b = 0.481M^{0.76}$	Inhalation rate (m^3/d)	0.026447603	Pedley, p. 15, Table V., adjusted to provide units of m^3/d
		0.481: constant in calculation to give m^3/d	0.481	
		see above equation, M: body mass in kg	0.022	
		0.76: exponent in equation	0.76	
$r_{inhalation}$	$r_{inhalation} = xr_b$	Soil inhalation rate (g/d)	2.64476E-06	derived
		x: airborne dust loading, g/m^3	0.0001	
		r_b: inhalation rate (see above)	0.026447603	
I_w	$I_w = 0.099M^{0.90}$	Water consumption rate (L/d)	0.003190183	EPA Wildlife Exposure Factor Handbook, Vol. 1, p. 3-10, equation 3-17
		0.099: constant in equation	0.099	
		see above equation, M: body mass in kg	0.022	
		0.9: exponent in calculation	0.9	

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Concluding Material

Review Activity:

DOE Programs

EH
EM
SC
DP
EE
FE
GC
IG
NE
NN
PO
RW

Operations and Field Offices

AL
CH
ID
NV
OR
RL
SR
BPO
CAO
FETC-PA
FETC-WV
OH
OK
RF

Laboratories

ANL
INEEL
ORNL
PNNL
SNL
SREL
AMES
ARG
BHG
BNL
EML
FNAL
LANL
LBNL
LLNL
NBL
SLAC
TJAF

Area Offices

Rocky Flats Area Office
West Valley Area Office
Amarillo Area Office
Fernald Area Office
Golden Field Office
Grand Junction Office
Kirtland Area Office
Kansas City Area Office
Miamisburg Area Office
Western Area Power Administration

Preparing Activity:

DOE-EH-412

Project Number:

ENVR-0011

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