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

MODULE 2

DETAILED GUIDANCE

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1 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.

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 are useful in defining the areas of contamination and the identification of localized areas of high concentration. Measurements of gross alpha and beta radiation could potentially be used in demonstrating compliance with biota dose limits. Derivation of BCGs for gross alpha and beta, for use in the general screening phase of the graded approach, is being considered by the BDAC. Appropriate BCGs may be included in the next issue of the technical standard.

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.

1.2 Receptors and Routes of Exposure

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 1.1, 1.2, 1.3, and 1.4, 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.

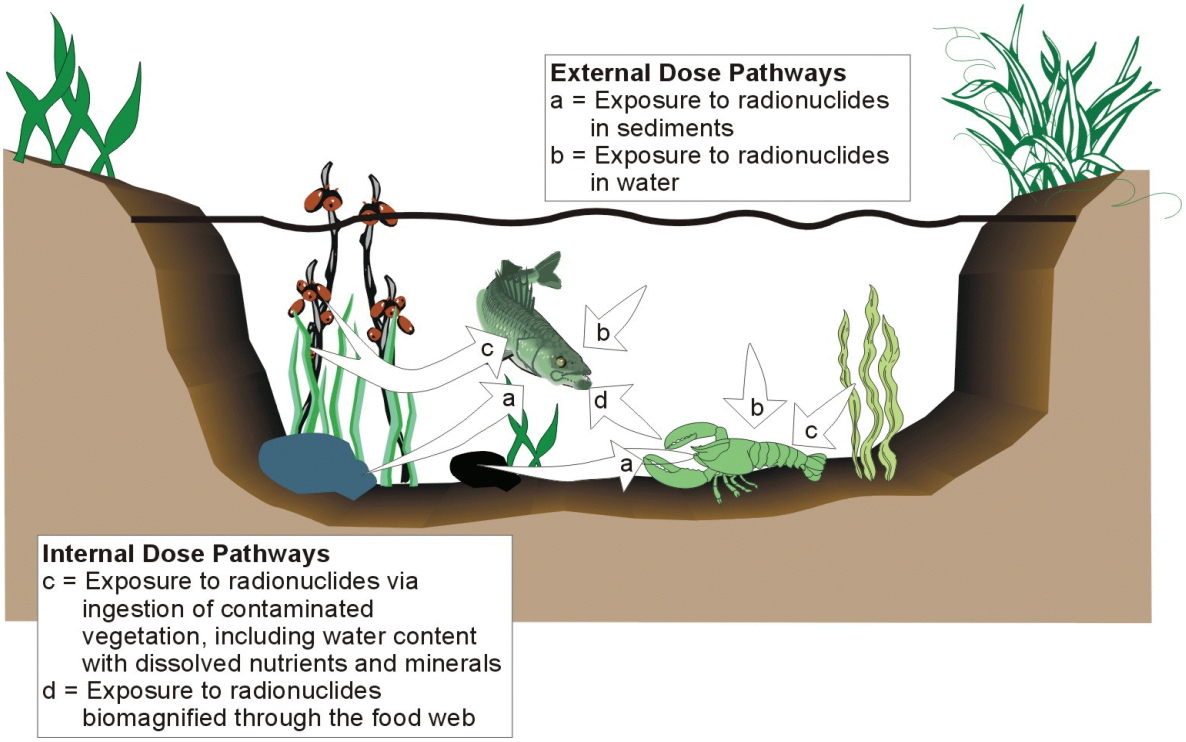

Figure 1.1 Exposure Pathways for Aquatic Animals

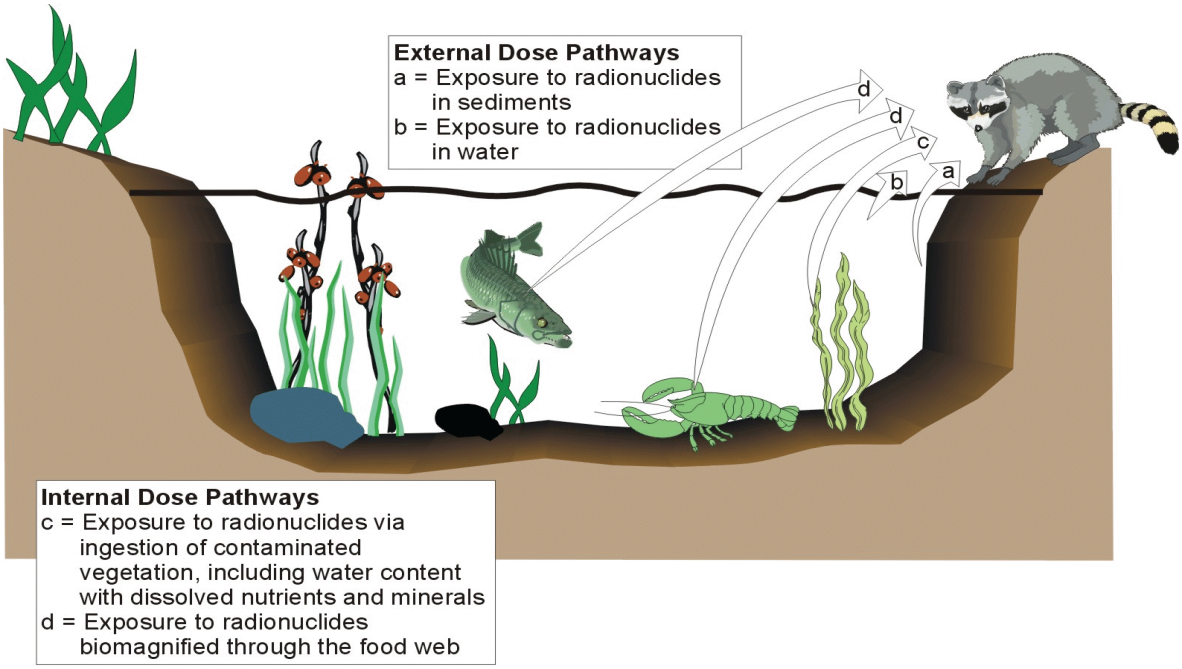

Figure 1.2 Exposure Pathways for Riparian Animals

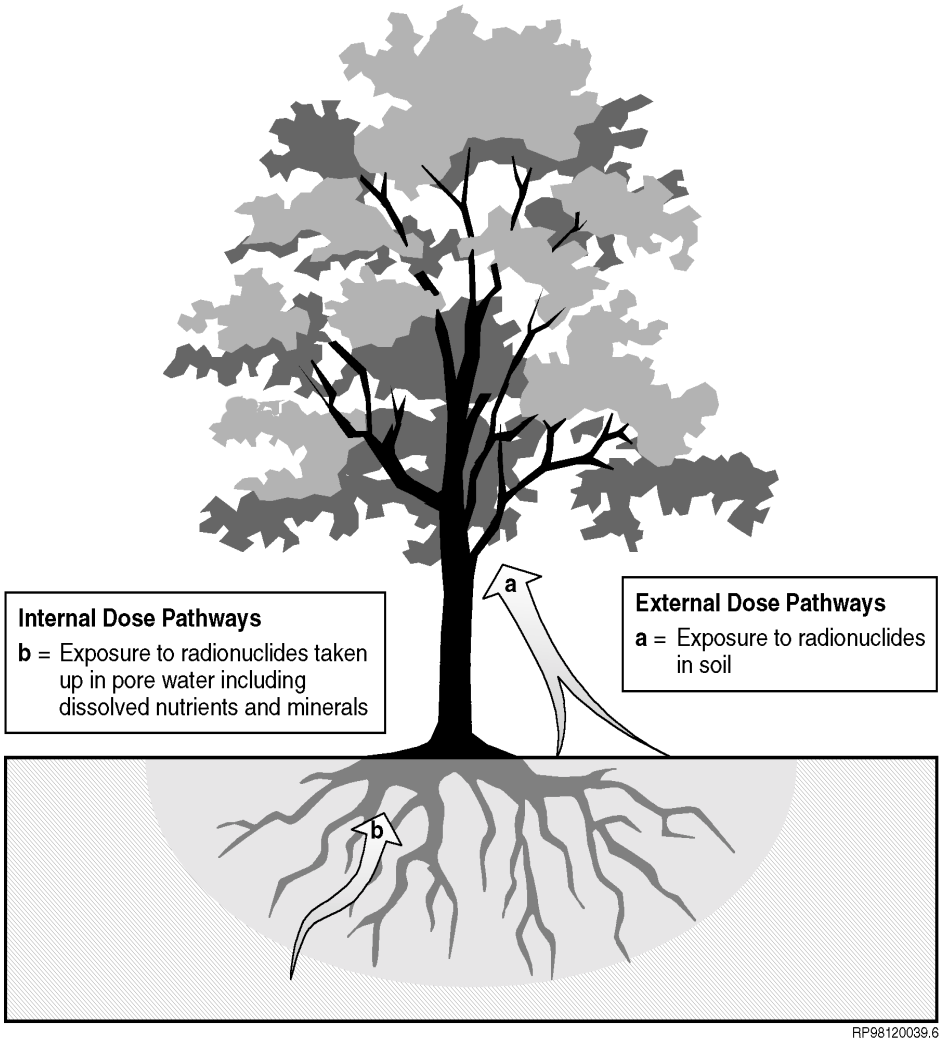

Figure 1.3 Exposure Pathways for Terrestrial Plants

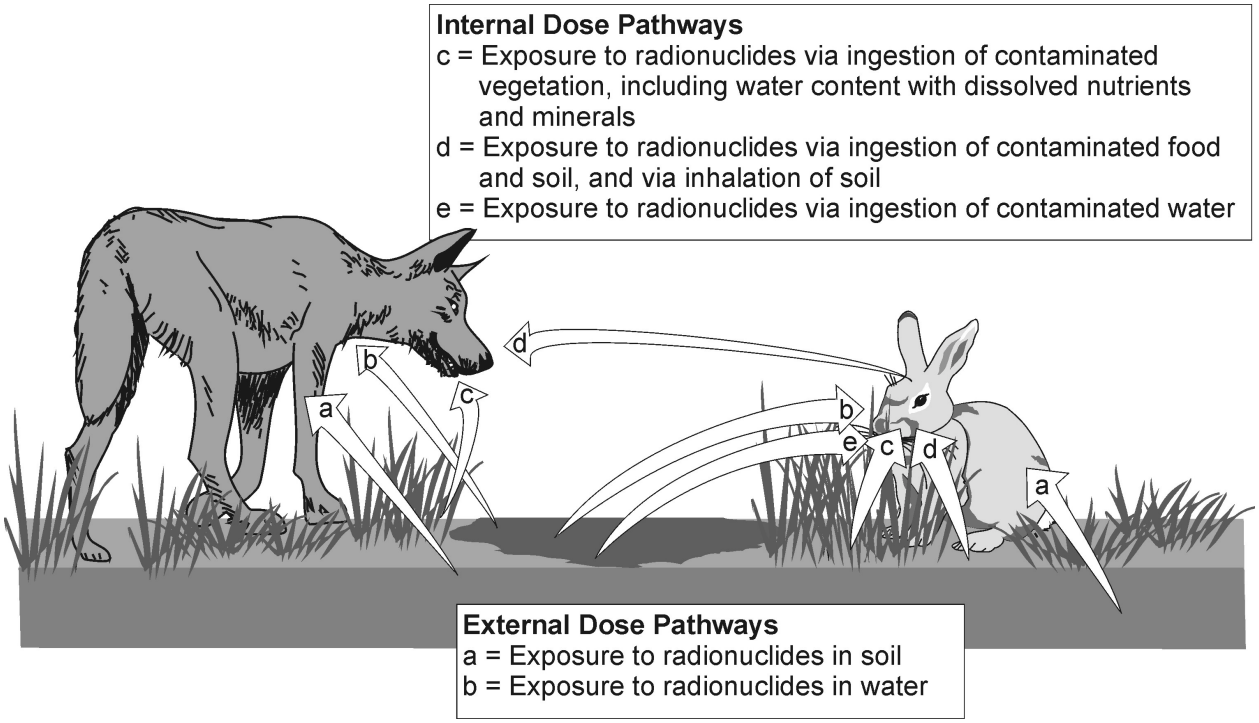

Figure 1.4 Exposure Pathways for Terrestrial Animals

1.2.1 Rationale for the Air Pathway as a Minor Source of Exposure

Airborne emissions of radionuclides are considered to be a minor source of exposure for animals and plants. Several points are offered which support this assumption:

- First, airborne emissions of radionuclides are limited to very small quantities to protect human health, specifically to an effective dose equivalent of 10 mrem/y (40 CFR 61.92) to members of the public. For all sources combined, the total effective dose equivalent to members of the public during onsite access is limited to 100 mrem/y (40 CFR 835.208). Second, terrestrial biota are exposed to approximately the same airborne concentrations of radionuclides as workers or members of the public and for approximately the same lengths of time.

- 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.

- 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. Although plants may also absorb carbon dioxide and water directly from the air, which would contribute to an internal dose, radioactive forms of these compounds (i.e., having the isotopes ^{14}C or ^{3}H) would not be taken up preferentially over non-radioactive forms. Moreover, we are not aware of any non-essential elements that are absorbed by plants from air rather than from soil. Hence, it is again reasonable to assume that the much lower maximum doses from airborne emissions that are specified for humans would provide an adequate level of protection for terrestrial plants.

- Internal dose to terrestrial animals is largely a function of ingestion rather than inhalation. For terrestrial plants, soils serve as the ultimate integrator of radionuclides originating and transported via the air pathway.

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.

1.2.2 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:

- 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, and Cummins 1994).

- 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 may 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.

1.3 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 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.

1.3.1 Biogeochemical Properties of the Released 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.

1.3.2 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.

1.3.3 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.

1.3.4 Ecology of the Receptors

The interactions of each receptor within its environment define the routes of its exposure. For a given area in which the soil is contaminated, 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 also determines how the receptor is exposed in time and space. For example, 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.

Considering the above and other relevant factors, and 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. Such models can be developed quickly and easily and are often useful for identifying exposure pathways. Helpful references include ASTM (1995), EPA (1998), and Suter (1996).

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2 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 3.

2.1 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 non-human 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).

The dose limits for radiation protection of non-human 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 non-human 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.

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2.1.1 Guidance on Time Averaging in Applying Daily Dose Limits

The guidance on the use of time averaging in applying the daily dose limits for non-human 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.

2.1.2 Rationale for Guidance on Time Averaging

The guidance on the use of time averaging in applying the daily dose limits for non-human 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 non-human 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 non-human 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.

- Based on studies involving short-term exposures, dose rates about 2-5 times higher than the daily limits for non-human 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 non-human 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 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 non-human 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 and off-normal releases at DOE sites is taken into account.

2.2 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 non-human 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 non-human 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 non-human 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 non-human 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.

2.3 Guidance on Estimating Mean Values

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 3, 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).

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 5, for related information.

2.3.1 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 3. 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 5, 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.

- 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 soil and sediment media should be estimated based on data for the same or similar 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 5, 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 1997). In cases where very little data are available on the distributions of radionuclide concentrations, a preliminary survey may be needed.

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3 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 2 for guidance on spatial and temporal averaging of radionuclide concentrations.

3.1 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.

3.2 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 ionizing radiation; 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 intersect a single habitat patch (Fig. 3.1 (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 (Fig. 3.1 (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 (Fig. 3.1 (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 (Fig. 3.2). 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.

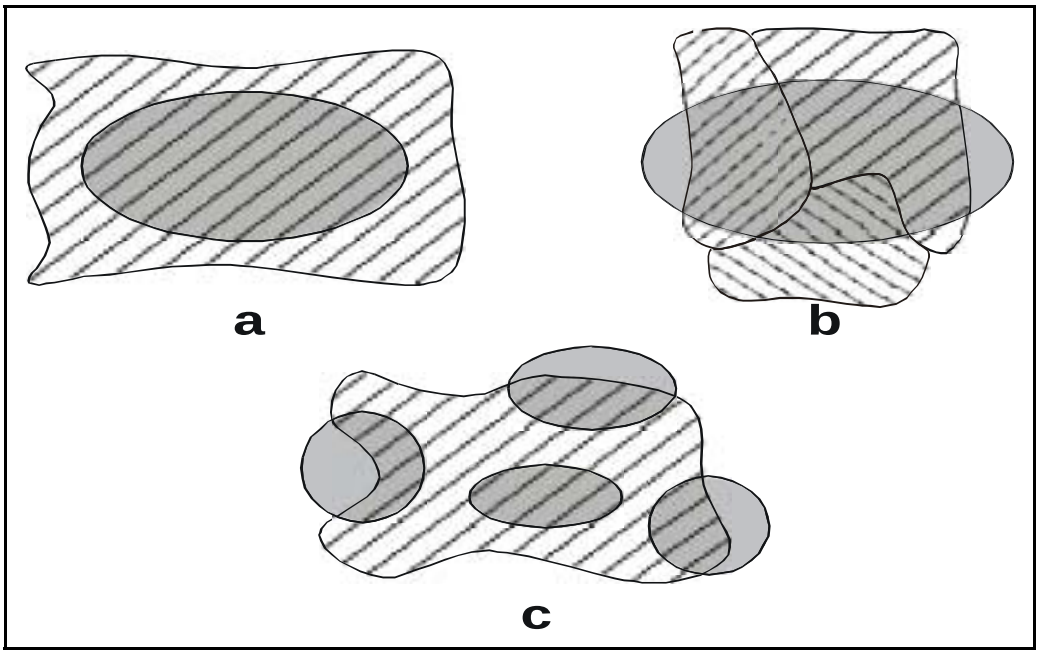

Figure 3.1 Hypothetical maps of contaminated areas and discrete habitat used to determine appropriately scaled assessment areas. Shading indicates contaminated areas. The cross-hatching indicates habitat types. Three cases are considered: **(a)** a single contaminated area intersects a single habitat patch; **(b)** a single contaminated area intersects multiple habitat patches; **(c)** multiple small contaminated areas intersect a single large habitat patch.

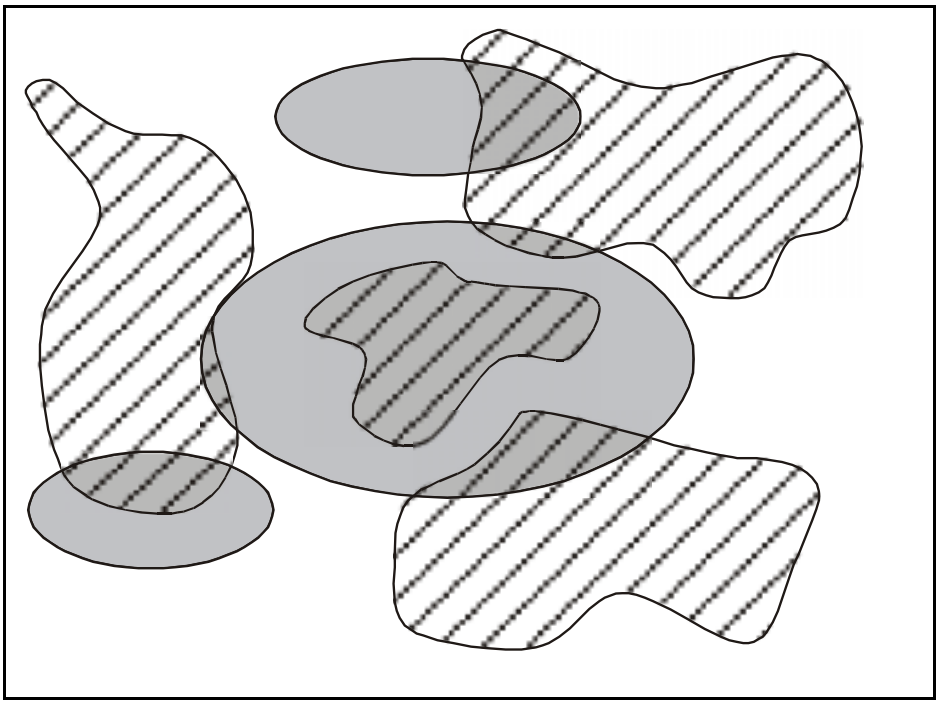

Figure 3.2 A hypothetical map of multiple areas with the same contamination intersecting multiple patches of the same discrete habitat type used to determine appropriately scaled assessment areas.

4 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.

4.1 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 U.S. 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 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 will 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).

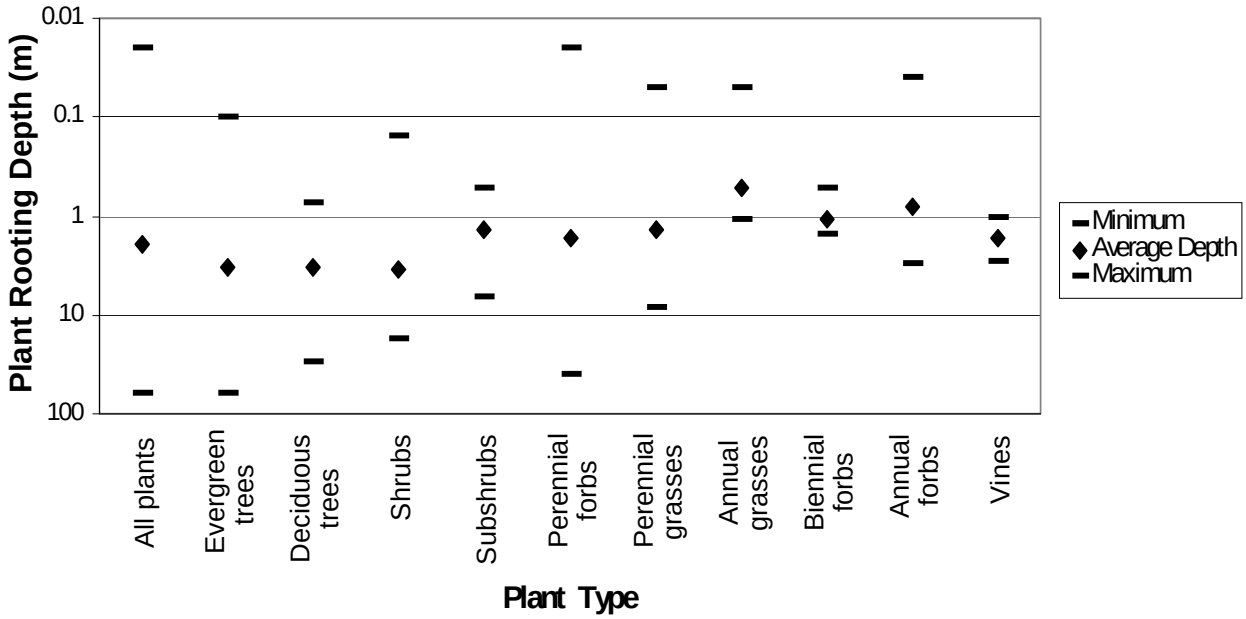

Figure 4.1 Average Rooting Depth by Plant Type
(source data from Foxx et al. 1984)

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.

4.2 Plant Rooting Depths

Plant roots can extend considerable depths into the subsurface, as indicated in Figure 4.1 and Table 4.1. Ranges of rooting depths vary considerably among plant types (Figure 4.1) and individual species. For these reasons, the data in Figure 4.1 and Table 4.1 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 4.1 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**

4.3 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.

4.4 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. [See 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. More mature plants will have better developed root systems with greater surface areas available for absorption of radioactive substances. They will typically 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. Compared with periods of normal rainfall, periods of heavy rain may dilute concentrations in leaves due to increased transpiration.

- **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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5 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 non-human receptors. Guidance is provided on sampling biota to estimate mean radionuclide body burdens in representative individuals of a population. The methods in this section are not intended to be used for assessing risks to protected species (e.g., threatened and endangered) or other receptors for which population-level assessments are insufficiently rigorous. Also, 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 non-human 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 non-human 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 5.1 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.

5.1 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.

5.2 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.

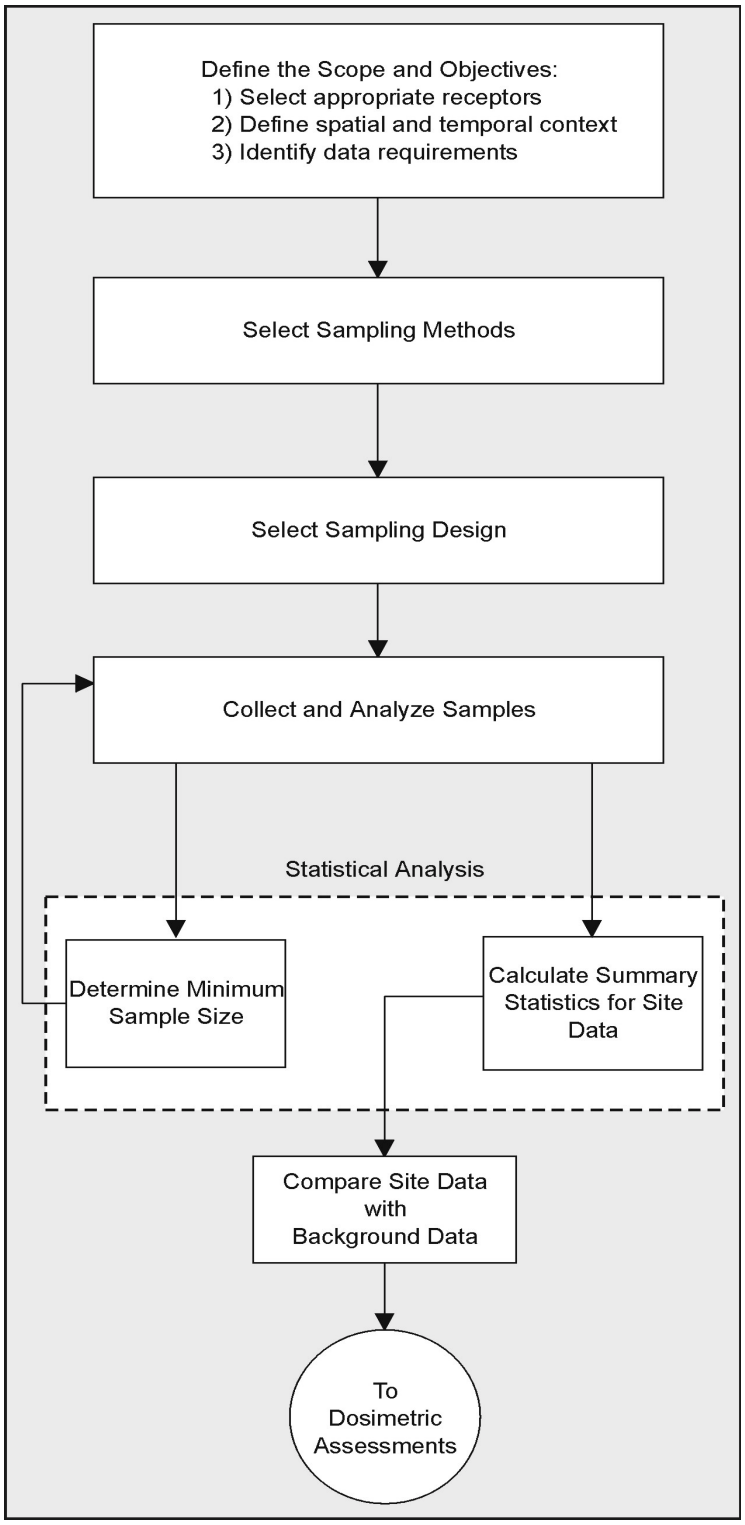

Figure 5.1 Flow Diagram for Collecting Biological Samples to Produce Data for Dosimetric Assessments of Non-Human Receptors

5.2.1 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.

5.2.2 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, relevance to management goals, and susceptibility to irradiation (EPA 1998). 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 non-human 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. They are likely to be an assessment endpoint when present at a site, but the methods discussed in this technical standard may be inappropriate for their evaluation, because the management goals may specify protection of individuals, rather than populations. Second, pine trees (*Pinus spp*.) are 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 typically accumulate more cesium than do fish feeding primarily in the water column (IAEA 1994).

5.2.3 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 2.

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. If contamination and suitable habitat do not overlap spatially, exposure is unlikely. Sampling designs that account for these issues are presented in Section 5.3 of this Module.

5.2.4 Representative Population Exposures

An important premise of this technical standard is that the acceptable dose limits 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). Clearly, it would not be appropriate to lump together all mammals at a site (e.g., herbivores, vermivores, and carnivores) in order to estimate the exposure to a representative mammal. 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.

5.2.5 Dosimetry Models

An important planning question is, "How will the internal concentrations be used to estimate dose rates?" The dosimetric models available for non-human 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.

5.2.5.1 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 DOE 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 non-human 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 non-human 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 non-human 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 non-human biota; wildlife generally consume the entire organism, not just the muscle tissue. Hence, whole-body concentrations are the appropriate measurements for estimating food chain transfers to non-human biota.

5.2.5.2 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.

5.2.5.3 Analytical Requirements

A general sample preparation issue that should be considered is whether or not external contamination is removed prior to analysis. Including deposited contamination in biota samples is 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.

5.2.5.4 Other Data Needs

Collecting biota is only one component of any sampling plan intended to refine the dose estimates produced by the DOE graded approach methods; biota concentrations can only be used to improve the estimated internal dose rate. 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.

5.3 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.

5.3.1 Sampling Considerations

For sampling, population definitions, sampling units, and sampling design must be considered.

5.3.1.1 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.

5.3.1.2 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.

5.3.1.3 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 5.4) 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.

5.3.1.4 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).

5.3.1.5 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 biota dose limits for the protection of biota do not differentiate between radionuclides originating from anthropogenic and natural sources. However, if remedial activities are to be conducted, it is important to differentiate between these sources 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 soils 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).

5.3.2 Statistical Considerations

Statistical concerns include underlying data distributions, summary statistics and confidence limits, and minimum sampling size.

5.3.2.1 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).

5.3.2.2 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. Readers are instructed to refer to the cited texts before 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 ith 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:

$$\hat{\mu} = \exp(\bar{y} + s_y^2/2)$$

$$\hat{\delta}^2 = \hat{\mu}^2\ [\exp(s_y^2) - 1]$$

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:

$$UCL_{1-\alpha} = \exp(\bar{y} + 0.5s_y^2 + \frac{s_y H_{1-\alpha}}{\sqrt{n-1}})$$

$$LCL_{\alpha} = \exp(\bar{y} + 0.5s_y^2 + \frac{s_y H_{\alpha}}{\sqrt{n-1}})$$

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.

5.3.2.3 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 ($\bar{x}$) 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 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), $\alpha = 0.05$, $s^2_y = 2$, 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 $\alpha = 0.05$, the minimum sample size declines to a more manageable 47.

5.3.3 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.

5.4 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.

5.4.1 Aquatic Biota

Aquatic biota include fish, benthic invertebrates, and amphibians and reptiles.

5.4.1.1 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).

5.4.1.2 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).

5.4.1.3 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.

5.4.2 Terrestrial Biota

Terrestrial biota taken for sampling include plants, mammals, birds, earthworms, and terrestrial arthropods.

5.4.2.1 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).

5.4.2.2 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).

5.4.2.3 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).

5.4.2.4 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.

5.4.2.5 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.

5.4.3 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.

5.4.3.1 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 5700.6C. Specifications and guidelines for quality systems for environmental data collection and environmental technology programs are presented in ASQC (1994).

5.4.3.2 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.

5.4.3.3 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.

5.4.3.4 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?

5.4.3.5 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 5.1 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.

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-recap-ture. 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.

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 arthro-pods. 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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6 Interim 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. Interim 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 interim guidance is summarized.

6.1 Statement of Issue

The limits on radiation dose to aquatic and terrestrial biota adopted in this manual 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.

6.2 Previous Assumptions About Radiation Weighting Factor

In radiation protection of humans, an average quality factor ($\overline{Q}$) 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 $\overline{Q}$ = 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 $\overline{Q}$ = 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 weighted absorbed dose to aquatic and terrestrial biota (IAEA 1992). This value also has been

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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).

used by other investigators (Blaylock et al., 1993; Whicker 1988).

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 $\overline{Q}$ = 20 used in radiation protection of humans, may not be appropriate for biota (Baker and Soldat 1992, and 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).

6.3 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.

6.4 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.

- The RBE for deterministic effects induced by 5-50 MeV neutrons varies from 1 to 10,

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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.

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.

6.5 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.

6.6 Interim Guidance on Radiation Weighting Factor for Alpha Particles

As an interim measure, 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 compliance 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 interim 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 interim 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.

7 Guidance on the Applicability of the Graded Approach for Evaluating Dose to Individual Organisms

7.1 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.

7.2 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.

7.3 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.

7.4 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.

7.5 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 concern 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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