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SUBJECT: Methods and Tools for Estimation of the Exposure of Terrestrial Wildlife to Contaminants

TO: EH-41 Web site users

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METHODS AND TOOLS FOR ESTIMATION OF THE EXPOSURE OF TERRESTRIAL WILDLIFE TO CONTAMINANTS

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EXECUTIVE SUMMARY

A critical component in ecological risk assessment is the evaluation of exposure experienced by endpoint receptors. Exposure can be defined as the coincidence in both space and time of a receptor and a stressor, such that the receptor and stressor come into contact and interact. Without sufficient exposure of the receptor to the contaminants, there is no ecological risk.

Unlike some other endpoints considered in ecological risk assessments, terrestrial wildlife are significantly exposed to contaminants in multiple media. They may drink or swim in contaminated water, ingest contaminated food and soil, and breathe contaminated air. Exposure models for terrestrial wildlife must therefore include multiple media. In addition, because most wildlife are mobile, moving among and within habitats, exposure is not restricted to a single location. They may integrate contamination from several spatially discrete sources. As a consequence, the accurate estimation of wildlife exposure requires the consideration of habitat requirements and spatial movements.

This report presents methods for estimating exposure of terrestrial wildlife to both chemical (Sect. 2.1) and radionuclide (Sect. 2.2) contaminants. Approaches for probabilistic exposure estimation (Sect. 2.3) and extrapolation from individual-level exposures to population-level effects (Sect. 2.4) are reviewed. Finally, methods and models to estimate contaminant concentrations in selected food types consumed by wildlife (Sect. 3.2) and life history parameters (Sect. 3.3) needed to accurately estimate exposure are presented.

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1. INTRODUCTION

Exposure can be defined as the coincidence in both space and time of a receptor and a stressor such that the receptor and stressor come into contact and interact (Risk Assessment Forum 1992). In the context of ecological risk assessment, receptors include all endpoint species or communities identified for a site [see Suter (1989) and Suter et al. (1995) for discussions of ecological endpoints for waste sites]. In the context of hazardous waste site assessments, stressors are chemical contaminants and the contact and interaction are represented by the uptake of the contaminant by the receptor. Without sufficient exposure of the receptor to the contaminants, there is no ecological risk.

Unlike some other endpoint assemblages, terrestrial wildlife are significantly exposed to contaminants in multiple media. They may drink or swim in contaminated water, ingest contaminated food and soil, and breathe contaminated air. Exposure models for terrestrial wildlife must therefore include multiple media. In addition, because most wildlife are mobile, moving among and within habitats, exposure is not restricted to a single location. They may integrate contamination from several spatially discrete sources. As a consequence, the accurate estimation of wildlife exposure requires the consideration of habitat requirements and spatial movements.

The purpose of this report is to present generalized methods for the estimation of exposure of terrestrial wildlife, focusing primarily on methods and models for birds and mammals. Reptiles and amphibians are not considered because few data exist with which to assess exposure to these organisms. In addition, because toxicological data are scarce for both classes, evaluation of the significance of exposure estimates is problematic. The general exposure estimation procedure developed for birds and mammals, however, is applicable to reptiles and amphibians (EPA 1993).

Methods are presented for estimating exposure to both chemical (Sect. 2.1) and radionuclide (Sect. 2.2) contaminants. Approaches for probabilistic exposure estimation (Sect. 2.3) and extrapolation from individual-level exposures to population-level effects (Sect. 2.4) are reviewed. In addition to exposure models, methods and models to estimate contaminant concentrations in selected food types consumed by wildlife (Sect. 3.2) and life history parameters (Sect. 3.3) needed to accurately estimate exposure are presented.

2. METHODS FOR ESTIMATION OF EXPOSURE

Contaminants to which terrestrial wildlife may be exposed may be grouped into two broad classes: chemical (e.g., heavy metals, organics) and radionuclide. Because the mode of action differs greatly between these two general classes of contaminants, methods for estimation of exposure also differ. Methods for estimation of exposure to both chemical and radionuclide contaminants are presented below.

2.1 ESTIMATION OF EXPOSURE TO CHEMICAL CONTAMINANTS

As terrestrial wildlife move through the environment, they may be exposed to contamination via three pathways: oral, dermal, and inhalational. Oral exposure occurs through the consumption of contaminated food, water, or soil. Dermal exposure occurs when contaminants are absorbed directly through the skin. Inhalational exposure occurs when volatile compounds or fine particulates are respired into the lungs. The total exposure experienced by an individual is the sum of exposure from all three pathways or

$$E_{total} = E_{oral} + E_{dermal} + E_{inhal} , \qquad (1)$$

where

E_{total} = total exposure from all pathways,
E_{oral} = oral exposure,
E_{dermal} = dermal exposure,
E_{inhal} = exposure through inhalation.

Dermal exposure is assumed to be negligible for birds and mammals on most U.S. Department of Energy (DOE) hazardous waste sites. While methods are available to assess dermal exposure to humans (EPA 1992), data necessary to estimate dermal exposure are generally not available for wildlife (EPA 1993). Additionally, many contaminants found at DOE facilities (e.g., metals and radionuclides) are unlikely to be absorbed through skin (Camner et al. 1979; Watters et al. 1980). Feathers and fur of birds and mammals further reduce the likelihood of significant dermal exposure by limiting the contact of skin with contaminated media. Therefore, dermal exposure is expected to be negligible relative to other routes in most cases. If contaminants that have a high affinity for dermal uptake are present (e.g., organic solvents and pesticides) and an exposure scenario for an endpoint species is likely to result in significant dermal exposure (e.g., burrowing mammals or swimming amphibians), dermal exposure may be estimated using the model for terrestrial wildlife presented by Hope (1995).

Inhalation of contaminants is also assumed to be negligible at most DOE facilities. This is for two reasons. First, because most contaminated sites are either capped or vegetated, exposure of contaminated surface soils to winds and resulting aerial suspension of contaminated dust particulates are minimized. Second, most volatile organic compounds (VOCs), the contaminants most likely to present a risk through inhalation exposure, rapidly volatilize from soil and surface water to air, where they are rapidly diluted and dispersed. Paterson et al. (1990) suggest that organic compounds with soil half-lives of <10 days are generally lost from soil before significant exposure can occur. As a consequence, significant exposure to VOCs through inhalation is unlikely. In situations where inhalation exposure of endpoint species is believed to be occurring or is expected to occur, models for vapor or particulate inhalation (Hope 1995) may be employed. In these cases, EPA (1993) recommends consulting an inhalation toxicologist.

Because contaminant exposure experienced by wildlife through both the dermal and inhalation pathways is negligible, the majority of exposure must be attributed to the oral exposure pathway. Equation 1 can therefore be simplified to

$$E_{total} \approx E_{oral} \ . \tag{2}$$

2.1.1 Estimation of Oral Exposure

Oral exposure experienced by wildlife may come from multiple sources. They may consume contaminated food (either plant or animal), drink contaminated water, or ingest soil. Soil ingestion may be incidental while foraging or grooming or purposeful to meet nutrient needs. The total oral exposure experienced by an individual is the sum of the exposures attributable to each source and may be described as

$$E_{oral} \approx E_{food} + E_{water} + E_{soil} \ , \tag{3}$$

where

E_{food}	=	exposure from food consumption,
E_{water}	=	exposure from water consumption,
E_{soil}	=	exposure from soil consumption.

For exposure estimates to be useful in the assessment of risk to wildlife, they must be expressed in terms of a body weight-normalized daily dose or milligrams of contaminant per kilograms body weight per day (mg/kg/d). Exposure estimates expressed in this manner may then be compared to toxicological benchmarks for wildlife, such as those derived by Sample et al. (1996a), or to doses reported in the toxicological literature. Models for the estimation of exposure from oral ingestion have been reported in the literature (EPA 1993, Sample and Suter 1994, Hope 1995, Pastorok et al. 1996, Freshman and Menzie 1996) and are generally of the form

$$E_j = \sum_{i=1}^{m} (I_i \ x \ C_{ij}) \ , \tag{4}$$

where

E_j	=	total oral exposure to contaminant (j) (mg/kg/d),
m	=	total number of ingested media (e.g., food, water, or soil),
I_i	=	ingestion rate for medium (i) (kg/kg body weight/d or L/kg body weight/d),
C_{ij}	=	concentration contaminant (j) in medium (i) (mg/kg or mg/L).

Very few wildlife consume diets that consist exclusively of one food type. To meet nutrient needs for growth, maintenance, and reproduction, most wildlife consume varying amounts of multiple food types. Because it is unlikely that all food types consumed will contain the same contaminant concentrations, dietary diversity is of one of the most important exposure modifying factors.

To account for differences in contaminant concentrations of different food types, exposure estimates should be weighted by the relative proportion of daily food consumption attributable to each food type and the contaminant concentration in each food type. In addition, wildlife may drink from different water sources

and consume soils that differ in contaminant concentrations. These differences must also be accounted for. This may be done by modifying Eq. 4 as follows

$$E_j = \sum_{i=1}^{m} \sum_{k=1}^{n} p_{ik}(I_i \ x \ C_{ijk}) \ , \tag{5}$$

where

n	=	number of types of medium (i) consumed (unitless),
p_{ik}	=	proportion of type (k) of medium (i) consumed (unitless),
C_{ijk}	=	concentration of contaminant (j) in type (k) of medium (i) (mg/kg or mg/L).

If the site is spatially heterogeneous with respect to either contamination or wildlife use, the model must be modified to include spatial factors. The most important spatial consideration is the movement of wildlife. Animals travel varying distances, on a daily to seasonal basis, to find food, water, and shelter. The area encompassed by these travels is defined as the home range (we use the term here to include territories). If the site being assessed is larger than the home range of an endpoint species and provides the habitat needs of the species, then the previously listed models are adequate. However, endpoint species often have home ranges that are larger than contaminated sites, or the contaminated site may not supply all of a species' habitat requirements. In those cases, the wildlife exposure model must be modified.

If the contaminated site has similar habitat quality to the surrounding area but is smaller than the home range, use of the contaminated site is simply a function of its area. That is, one can assume that for wildlife that use the entire contaminated area, exposure is proportional to the ratio of the size of the contaminated site to home range size. Eq. 5 can be modified as follows as follows:

$$E_j = \frac{A}{HR}[\sum_{i=1}^{m} \sum_{k=1}^{n} p_{ik}(I_i \ x \ C_{ijk})] \ , \tag{6}$$

where

A	=	area (ha) contaminated,
HR	=	home range size (ha) of endpoint species.

Note that *A* is the area contaminated, not the entire area that has been designated a hazardous waste site (e.g., an operable unit). Because boundaries are often drawn conservatively, they may contain a considerable uncontaminated area.

The previous equation (6) implies that all of the habitat within a contaminated area is suitable and that use of all portions of the contaminated area is equally likely. Because many waste sites are industrial or highly modified in nature, it is unlikely that all areas within their bounds will provide habitat suitable for endpoint species. If it is assumed that use of a waste site will be proportional to the amount of suitable habitat available on the site, Eq. 6 may be modified to read

$$E_j = P_h(\frac{A}{HR}[\sum_{i=1}^{m} \sum_{k=1}^{n} p_{ik}(I_i \ x \ C_{ijk})]) \ , \tag{7}$$

where

P_h = proportion of suitable habitat in the contaminated area.

One complication is the spatial heterogeneity of contaminants on waste sites. These models (Eqs. 4-7) are based on the assumption that either contaminants are evenly distributed on the site, or wildlife forage randomly with respect to contamination on the portion of the site that constitutes habitat so that they are exposed to mean concentrations. However, if contaminant levels are related to habitat quality, that assumption would not hold. For example, contaminant concentrations might be greatest near the center of a site, but the habitat quality might be highest near the edges. In such cases, it might be necessary to model the proportional contribution of each area with a distinct combination of contaminant level and habitat quality

$$E_j = \sum_{l=1}^{o} \left(\frac{A_l}{HR} \left[\sum_{i=1}^{m} \sum_{k=1}^{n} p_{ik} (I_i \ x \ C_{ijkl}) \right] \right) , \tag{8}$$

where

o = number of distinct contaminated habitat areas,

A_l = area (ha) of a distinct contaminated habitat area,

C_{ijkl} = concentration of contaminant (j) in type (k) of medium (i) from the lth area (mg/kg or mg/L).

As can be seen, if the distribution of contamination and habitat quality is complex, this approach to exposure estimation rapidly becomes ungainly. In such cases, it is advisable to implement the exposure in a Geographic Information System (GIS). Using a GIS, maps displaying the spatial distribution of habitat types may be overlaid with maps of contaminant distribution to accurately determine the degree to which habitat is contaminated. Furthermore, if information on the distribution or movements of wildlife (generated by radiotelemetry or censuses) are available, these data may be combined with the habitat and contaminant data to provide a more accurate visualization of exposure. Examples of the application of GIS to wildlife exposure and risk assessments can be found in Clifford et al. (1995), Banton et al. (1996), Henriques and Dixon (1996) and Sample et al. (1996b).

2.1.2 Exposure-Modifying Factors

Factors other than those described in these models modify contaminant exposure experienced by wildlife endpoint species. These factors include age, sex, season, and behavior patterns.

The models above imply that the endpoint species have uniform body size, metabolism, diet, home ranges, and habitat requirements. However, these properties may differ between juveniles and adults and between males and females. For example, because they are actively growing, metabolism (and therefore food consumption) is generally greater for juveniles of most endpoint species. Diet composition may also differ dramatically between juveniles and adults of the same species. Similarly, the food requirements of females during reproduction are greater than that for males for many endpoint species. These factors may serve to make certain age classes or a particular sex experience greater contaminant exposure than other segments of the population. Because of their greater exposure, contamination may present a greater risk to these segments of the population. If it is known that a particular lifestage or sex is sensitive to contamination, that lifestage should be emphasized in the exposure assessment.

Behavior may modify exposure by increasing or decreasing the likelihood of contact with contaminated media. Wildlife behaviors are frequently seasonal in nature. Some foods may be available and consumed only at certain times of the year. Similarly, some habitats and certain parts of the home range may

be used only in certain seasons. In addition, many species hibernate or migrate; by leaving the area or restricting their activity to certain times of year, their potential exposure may be dramatically reduced. All of these factors should be considered when evaluating contaminant exposure experienced by wildlife, and exposure models should be adjusted accordingly. The simplest approach to modifying the exposure estimates to take into account some of these exposure-modifying factors is to generate multiple exposure estimates. For example, if diet differs by season or by sex, calculate exposure estimates for each sex or season. Comparison of exposure estimates generated for differing exposure scenarios will aid in identifying the segments of population at greatest risk or times of year when risk is greatest.

2.2 ESTIMATION OF EXPOSURE TO RADIONUCLIDES

Estimation of exposure and effects from radionuclides is both qualitatively and quantitatively different from estimation of exposure to chemical contaminants. Exposures to radionuclides may be internal or external, and effects are caused by energetic particles or rays released as part of the decay of atoms. Decay energies of particles or rays emitted by each radionuclide must be accounted for. Unlike chemical exposures where effects of chemicals are generally evaluated individually, the internal and external doses from all radionuclides present must be summed to arrive at the appropriate exposure dose for a given organism. In addition, a number of radionuclides have daughter products that must also be included in the exposure calculations.

Internal exposures result from ingestion of contaminated food, soil, or water or inhalation of contaminated soil or dust (Templeton et al. 1971, IAEA 1976, Blaylock and Trabalka 1978, Woodhead 1984). External exposures result from direct exposure to radiation from the soil and may occur either above or below ground (or a combination of both), depending on the habits of the receptor (e.g., fossorial vs nonfossorial). Evaluation of the resulting radiation doses received by biota requires quantitative information on the radionuclides to which they are exposed. In all cases, the radiation source must be known in terms of the quantity of each specific radionuclide (pCi/g) and the corresponding energy released per disintegration (MeV/dis). Conversions for units of dose and activity generally reported in the literature are presented in Table 1.

Table 1. Comparison of units of activity and absorbed dose of ionizing radiation under the international and conventional systems of measure

Measure	International system	Conventional system	Relationship
Activity	Becquerel (Bq) = one nuclear disintegration/s	Curie (Ci) = 3.7×10^{10} nuclear disintegrations/s	1 Bq = 2.7×10^{-11} Ci 1 Ci = 3.7×10^{10} Bq
Absorbed dose	Gray (Gy) = 1 Joule/kg	rad = 0.01 Joule/kg	1 Gy = 100 rad 1 rad = 0.01 Gy

Models for estimating radiation dose rates (mrad/d) for plants, earthworms, and terrestrial wildlife species are based on methodology from Blaylock et al. (1993) and Baker and Soldat (1992). The general methodology and the equations specific to each exposure route used in estimation of dose rates for biota are described below. In practice, doses from alpha (α), beta (β), and gamma (γ) emissions (only β and γ for external exposures of earthworms and plants and only γ for external exposures of wildlife receptors) should be calculated for each radionuclide of concern, including the dose rates from all short-lived daughter products for the radionuclides. Doses from each radionuclide (plus daughters) should then be summed over all

exposure routes and all radionuclides to arrive at the overall estimate of the dose received for each receptor. Alpha particles have low penetration energy and are not considered for external exposures. Beta particles are unlikely to penetrate the epidermis of larger organisms, so they are only considered in external exposures to plants and earthworms.

2.2.1 External Exposures: Direct Radiation from Soil

The equation for estimating aboveground external dose rates (mrad/d) for terrestrial receptors exposed to contaminated soil uses dose coefficients published by Eckerman and Ryman (1993). These dose coefficients relate the doses to organs and tissues in the body to concentrations of radionuclides in soil and are available for soil contaminated to depths of 1, 5, and 15 cm or soil assumed to be contaminated to an infinite depth. A dose rate reduction factor is used to account for the fraction of time the receptor spends aboveground. This factor is necessary because a different model is used to estimate below-ground exposures to soil radionuclides. The fraction of time spent above or below ground by each receptor species should be estimated based on knowledge of the species' life history and behavior patterns. Dose coefficients assume that the source region is a smooth plane (Eckerman and Ryman 1993), but this is rarely the case in a terrestrial habitat. A representative average dose reduction factor for ground roughness is 0.7, although recommended values range from essentially unity for paved areas to about 0.5 for a deeply plowed field (Eckerman and Ryman 1993). For relatively small mammals (e.g., mice, voles, and shrews) that are effectively much closer than 1 m to the source, an elevation correction factor (ECF) of 2 should be applied to account for the increased dose expected at ground level relative to the effective height of a standard human used to derive the dose coefficients. For large animals the ECF may be set at 1. If desired, more complex modeling may be conducted to arrive at ECFs for organisms of any given effective height above the ground. For plants it may be assumed that the dose represents that to the reproductive part of the plant with an effective height similar to that of the standard human. An ECF of 2 may be appropriate for evaluating low-growing plant species. The equation for aboveground dose from external exposures for a plant or wildlife receptor is

$$D_{abovegrd} = F_{above} \quad F_{ruf} \quad \sum \quad C_{soil,i} \quad DF_{grd,i} \; CFb \quad ECF \;, \tag{9}$$

where

$D_{above\ grd}$	=	external dose rate to receptor from aboveground exposures to contaminated soil (mrad/d),
F_{above}	=	dose rate reduction factor accounting for the fraction of time the receptor spends aboveground (unitless),
F_{ruf}	=	dose rate reduction factor accounting for ground roughness (unitless) [Representative average of 0.7 (Eckerman and Ryman 1993) is reasonable default],
$C_{soil,i}$	=	activity of radionuclide *i* in surface soil (pCi/g),
$Df_{grd,i}$	=	dose coefficient for radionuclide *i* in soil contaminated to given depth (Eckerman and Ryman 1993) (Sv/s per Bq/m^3),
CFb	=	conversion factor to change Sv/s per Bq/m^3 to mrad g/pCi d (Equals 5.12 x 10^{14}),
ECF	=	elevation correction factor to adjust dose coefficients to value representative of effective height of animal aboveground.

Dose from alpha radiation is not a concern for external sources, as alpha radiation lacks penetrating power. The effective dose coefficients from Eckerman and Ryman (1993) incorporate both high-energy β and γ emissions. Radionuclide-specific parameters for selected radionuclides are provided in Table 2. These include dose coefficients assuming soil contaminated to a depth of 15 cm. Coefficients for soil contaminated to depths of 1, 5, and 15 cm and to an infinite depth are available in Eckerman and Ryman (1993).

Below-ground exposures are calculated assuming immersion in a continuous soil medium. Dose coefficients are unavailable for the immersion scenario, so exposures can be modeled as dose to soil adjusted for absorption by a small volume of tissue. The exposure fraction reflects the fraction of time the receptor spends below ground. Receptors that do not go below ground (e.g., nonfossorial wildlife: deer, hawks, turkey, etc.) do not receive a dose via this exposure route. Only γ radiations with energies greater than 0.01 MeV were evaluated for wildlife receptors as those with lower energies are unlikely to penetrate skin. Both β and γ radiations were evaluated for earthworms. The equation for below-ground external exposures of earthworms and wildlife receptors is

$$D_{belowgrd} = 1.05 \quad F_{below} \sum C_{soil,i} \quad \epsilon_i \quad CFa \ , \tag{10}$$

where

$D_{below\ grd}$	=	external dose rate to earthworm or wildlife receptor in burrow from contaminated soil (mrad/d),
F_{below}	=	dose rate reduction factor accounting for the fraction of time the receptor spends below ground (unitless),
$C_{soil,\ i}$	=	activity of radionuclide *i* in surface soil (pCi/g),
ϵ_i	=	energy for γ emissions by nuclide *i* (MeV/nt),
1.05	=	conversion factor to account for immersion in soil vs water (estimated value; Keith Eckerman, Health Sciences Research Division, Oak Ridge National Laboratory, personal communication, June 1996),
CFa	=	conversion factor to go from MeV/nt to g mrad/pCi d. (5.12×10^{-2}).

Note that the conversion factor of 1.05 used to account for the difference between immersion in soil vs water was meant for small volumes of tissue. While it can be roughly applied to large animals, it may be more appropriate to consult a health physicist and conduct more complex calculations of dose from below-ground exposures for large animals expected to spend significant time below ground.

Table 2. Average energy of decay and absorbed fractions for select radionuclides

Radionuclide	Average energy of decay[a]			Absorbed fractions (gamma)[b]					$DF_{grd\ 0\text{-}15}$ (Sv m3/s Bq)[c]
	alpha	beta	gamma	A	B	C	D	E	
Actinium-228		0.475	0.971	0.01	0.0127	0.04	0.06	0.14	2.76e-17
Americium-241	5.479	0.052	0.033	0.04	0.05	0.12	0.16	0.3	1.23e-18
Antimony-126		0.283	2.834	0.01	0.01	0.03	0.04	0.11	8.13e-17
Antimony-126m		0.591	1.548	0.085	0.0123	0.03	0.05	0.12	4.44e-17
Astatine-218	6.697	0.04	0.007	0.63	0.79	0.94	0.94	0.94	3.13e-20
Barium-137m		0.065	0.597	0.011	0.015	0.04	0.06	0.15	1.71e-17
Beryllium-7			0.049	0.012	0.017	0.06	0.09	0.2	1.40e-18
Bismuth-210		0.389							1.86e-20
Bismuth-211	6.55	0.01	0.047	0.027	0.04	0.11	0.15	0.29	1.28e-18
Bismuth-212	2.174	0.472	0.186	0.01	0.011	0.04	0.06	0.14	5.36e-18
Bismuth-214		0.659	1.508	0.085	0.0123	0.03	0.05	0.12	4.36e-17
Cadmium-109		0.083	0.026	0.09	0.126	0.16	0.21	0.36	7.88e-20
Calcium-45		0.077							3.35e-22
Carbon-14		0.049							7.20e-23
Cesium-134		0.164	1.555	0.085	0.0123	0.03	0.05	0.12	4.47e-17
Cesium-137		0.187							3.94e-21
Cobalt-57		0.019	0.125	0.01	0.012	0.04	0.06	0.15	2.66e-18
Cobalt-60		0.097	2.504	0.01	0.01	0.03	0.04	0.11	7.25e-17
Curium-242	6.102	0.01	0.002	0.63	0.79	0.94	0.94	0.94	9.07e-22
Curium-243	5.797	0.138	0.134	0.01	0.0105	0.04	0.06	0.15	3.02e-18
Curium-244	5.795	0.009	0.002	0.63	0.79	0.94	0.94	0.94	6.74e-22
Europium-152		0.139	1.155	0.085	0.0123	0.03	0.05	0.12	3.75e-17
Europium-154		0.292	1.242	0.085	0.0123	0.03	0.05	0.12	4.11e-17
Europium-155		0.063	0.061	0.012	0.017	0.06	0.09	0.2	9.75e-19
Iodine-129		0.064	0.025	0.09	0.126	0.16	0.21	0.36	6.93e-20
Lead-212		0.176	0.148	0.01	0.011	0.04	0.06	0.15	3.62e-18
Lead-214		0.293	0.25	0.01	0.01	0.04	0.06	0.14	6.70e-18
Neptunium-237	4.769	0.07	0.035	0.027	0.04	0.11	0.15	0.29	4.16e-19
Plutonium-238	5.487	0.011	0.002	0.63	0.79	0.94	0.94	0.94	8.07e-22

Table 2. (continued)

Radionuclide	Average energy of decay[a]			Absorbed fractions (gamma)[b]					$DF_{grd\ 0\text{-}15}$ (Sv m3/s Bq)[c]
	alpha	beta	gamma	A	B	C	D	E	
Plutonium-239	5.148	0.007							1.52e-21
Plutonium-239/240	5.148	0.007	0.002	0.63	0.79	0.94	0.94	0.94	1.52e-21
Plutonium-240	5.156	0.011	0.002	0.63	0.79	0.94	0.94	0.94	7.84e-22
Polonium-210		0.038	0.005	0.63	0.79	0.94	0.94	0.94	2.45e-22
Polonium-211	7.442		0.008	0.63	0.79	0.94	0.94	0.94	2.24e-19
Polonium-212	8.785								3.62e-18
Polonium-214	7.687								2.40e-21
Polonium-216	6.779								4.87e-22
Polonium-218	6.001								2.63e-22
Potassium-40		0.523	0.156	0.01	0.0115	0.04	0.06	0.14	4.57e-18
Protactinium-233		0.196	0.204	0.01	0.01	0.04	0.06	0.14	5.16e-18
Protactinium-234		0.494	1.919	0.085	0.0123	0.03	0.05	0.12	5.38e-17
Protactinium-234m		0.822	0.012	0.55	0.63	0.93	0.93	0.93	4.20e-19
Radium-223	5.667	0.076	0.134	0.01	0.0105	0.04	0.06	0.15	3.10e-18
Radium-224	5.674	0.002	0.01	0.63	0.79	0.29	0.35	0.52	2.62e-19
Radium-226	4.774	0.004	0.007	0.63	0.79	0.94	0.94	0.94	1.65e-19
Radium-228		0.017							0.00e+00
Radon-220	6.288								1.10e-20
Radon-222	5.489								1.14e-20
Sodium-22		0.194	2.193	0.085	0.0123	0.03	0.05	0.12	6.31e-17
Strontium-90		0.196							3.72e-21
Technetium-99		0.101							6.70e-22
Thallium-207		0.493	0.002	0.63	0.79	0.94	0.94	0.94	9.48e-20
Thallium-208		0.598	3.375	0.01	0.01	0.03	0.04	0.11	9.68e-17
Thorium-228	5.4	0.021	0.003	0.63	0.79	0.94	0.94	0.94	4.17e-20
Thorium-230	4.671	0.015	0.002	0.63	0.79	0.94	0.94	0.94	6.39e-21
Thorium-231		0.165	0.026	0.09	0.126	0.16	0.21	0.36	1.94e-19
Thorium-232	3.996	0.012	0.001	0.63	0.79	0.94	0.94	0.94	2.78e-21

Table 2. (continued)

Radionuclide	Average energy of decay[a]			Absorbed fractions (gamma)[b]					$DF_{grd\ 0\text{-}15}$ (Sv m3/s Bq)[c]
	alpha	beta	gamma	A	B	C	D	E	
Thorium-234		0.06	0.009	0.63	0.79	0.94	0.94	0.94	1.29e-19
Tin-126		0.172	0.057	0.012	0.017	0.06	0.09	0.2	7.90e-19
Tritium		0.006							0
Uranium-232	5.302	0.017	0.002	0.63	0.79	0.94	0.94	0.94	4.83e-21
Uranium-233	4.817	0.006	0.001	0.63	0.79	0.94	0.94	0.94	7.24e-21
Uranium-233/234	4.817	0.006	0.001	0.63	0.79	0.94	0.94	0.94	7.24e-21
Uranium-234	4.758	0.013	0.002	0.63	0.79	0.94	0.94	0.94	2.14e-21
Uranium-235	4.396	0.049	0.156	0.01	0.0115	0.04	0.06	0.14	3.75e-18
Uranium-235/236	4.396	0.049	0.156	0.01	0.0115	0.04	0.06	0.14	3.75e-18
Uranium-236	4.396	0.049	0.156	0.01	0.0115	0.04	0.06	0.14	3.75e-18
Uranium-238	4.187	0.01	0.001	0.63	0.79	0.94	0.94	0.94	5.52e-22
Yttrium-90		0.935							1.20e-19
Zirconium-89		0.101	1.165	0.085	0.0123	0.03	0.05	0.12	3.85e-17

[a] Values were obtained from ICRP (1983).

[b] Absorbed fractions for worms, plants, and mouse were derived from data in Blaylock et al. (1993).
Absorbed fraction for other receptors were derived following methodology of Cristy and Eckerman (1987).
Absorbed fractions for beta radiation were 100% for all radionuclides listed.
A = Plants and soil invertebrates. Derived from large insect values presented in Blaylock et al. (1993).
B = Small mammals and birds <<1 kg (e.g., pine vole). Derived from small fish values in Blaylock et al. (1993).
C = Small- to medium-sized mammals and birds (e.g., mink). Derived from values for ~0.76kg human infant after Cristy and Eckerman (1987).
D = Medium-sized mammals and birds (e.g., red fox). Derived from values for ~2.5kg 1-year old human after Cristy and Eckerman (1987).
E = Large mammals (e.g., white-tailed deer). Derived from values for ~28kg human after Cristy and Eckerman (1987).

[c] DF_{grd} is the dose coefficient for soil contaminated to a depth of 15 cm (Eckerman and Ryman 1993).

2.2.2 Internal Exposures: Ingestion

Wildlife receptors may receive internal radiation doses after ingesting contaminated prey, soil, or water or after inhaling contaminated dust. Blaylock et al. (1993) provide an equation for estimating the internal dose to fish contaminated with radionuclides. This equation can be modified to address consumers eating a variety of prey types, ingesting soil, and drinking water, as well as plants and invertebrates taking up contaminants directly from the soil

$$D_{ing} = \sum QF \quad C_{tissue} \quad \epsilon_i \quad CFa \quad AF \ , \tag{11}$$

where

D_{ing}	=	internal dose rate received after ingestion of contaminated prey and soil (mrad/d),
QF	=	quality factor to account for the greater biological effectiveness of α particles (20 for α; 1 for β and γ emissions; unitless),
C_{tissue}	=	activity (pCi/g) of radionuclide *i* in tissue of organism,
ϵ_i	=	energy for α, β, or γ emissions by nuclide *i* (MeV/nt),
CFa	=	conversion factor to go from MeV/nt to g mrad/pCi d (5.12×10^{-2}),
AF	=	absorption factor (unitless).

Radionuclide activity in tissue may be determined a number of ways, depending on data availability. Measured data should be used, if available. In the absence of measured data, soil-to-tissue uptake factors may be used. Uptake factors for selected radionuclides in plants, soil invertebrates, and small mammals are presented in Table 3; additional discussion of uptake factors is presented in Sect. 3.2.

Absorbed energy fractions for α radiations are assumed to equal one for all receptors. While absorption fractions for β radiations are assumed to be one for wildlife receptors, β absorption fractions for plants and earthworms are assumed to equal those for large insects from Blaylock et al. (1993) (assuming small reproductive parts of greatest concern). This is because β radiations are unlikely to have sufficient energy to pass through the wildlife tissues; however, some fraction may have sufficient energy to pass through smaller organisms such as earthworms and plants. Absorption fractions for γ radiations for plants and earthworms were also assumed to be equivalent to those for large insects presented in Blaylock et al. (1993). Absorption fractions for γ radiations derived for infant, 1-yr old, and adult humans using the methodology described in Cristy and Eckerman (1987) were used for wildlife receptors of similar sizes. Table 2 presents absorption factors used for several receptor-radionuclide combinations.

Energies (α, β, and γ) for selected radionuclides were obtained from Eckerman and Ryman (1993) and are provided in Table 2. Because different types of radiation differ in their relative biological effectiveness per unit of absorbed dose, a quality factor derived from data on humans is normally applied (NCRP 1987). The quality factor is determined by the linear energy transfer of radiation, and linear energy transfer for α particles is substantially higher than that for β or γ emissions. A quality factor of 1 should be used for β and γ radiation and 20 for α radiation (Blaylock et al. 1993).

Table 3. Radionuclide-specific soil-tissue uptake factors for plants, soil invertebrates, and small mammals and bioaccumulation factors for birds and mammals

Radionuclide	UF_{plants} [a]				UF_{invert} [a]	BAF_{bird} [b]	BAF_{mamm} [b]	UF_{mamm} [a]
	All plants	Grass	Herb. plants	Tree/shrubs				
228Ac	8.75e-04 c	8.75e-04 d	8.75e-04 d	8.75e-04 d	1.25e-03 e	1.25e-03 f	1.25e-03 c,g	
241Am						4.20e-03 g,j	2.00e-03 j	
212Bi	8.75e-03 c,g	8.75e-03 d	8.75e-03 d	8.75e-03 d	2.00e-02 e	2.00e-02 f	2.00e-02 c	
214Bi	8.75e-03 c,g	8.75e-03 d	8.75e-03 d	8.75e-03 d	2.00e-02 e	2.00e-02 f	2.00e-02 c	
45Ca						2.80e-02 g,j	1.00e-01 j	
244Cm						1.00e-03 f	1.00e-03 k	
57Co						1.40e+00 g,j	5.00e-03 g,j	
60Co						1.40e+00 g,j	5.00e-03 g,j	
134Cs	1.27e-03 l	1.27e-03 l	1.27e-03 l	1.27e-03 l		7.00e+00 g,j	2.56e+00 l	1.62e-02 l
137Cs	1.27e-03 d	1.27e-03 m	1.27e-03 d	1.27e-03 d		7.00e+00 g,j	2.56e+00 m	1.62e-02 m
152Eu	1.05e-02 d	1.05e-02 d	1.05e-02 g,n	1.05e-02 d	1.00e-01 e	1.00e-01 f	1.00e-01 k	
154Eu	1.05e-02 d	1.05e-02 d	1.05e-02 g,n	1.05e-02 d	1.00e-01 e	1.00e-01 f	1.00e-01 k	
155Eu	1.05e-02 d	1.05e-02 d	1.05e-02 g,n	1.05e-02 d	1.00e-01 e	1.00e-01 f	1.00e-01 k	
129I	3.40e-04 d	3.40e-04 g,j	3.40e-04 d	3.40e-04 d	2.00e+00 e	7.00e-03 g,j	2.00e+00 g,j	
40K						1.00e+00 f	1.00e+00 j	
22Na						4.00e+00 f	4.00e+00 j	
237Np	9.00e-03 d	9.00e-03 d	9.00e-03 o	9.00e-03 d	9.00e-03 e	3.84e-03 f	3.84e-03 g,n	
234mPa	6.25e-04 c,g	6.25e-04 d	6.25e-04 d	6.25e-04 d	5.00e-02 e	5.00e-02 f	5.00e-02 c	
210Pb						2.00e-02 f	2.00e-02 j	
212Pb						2.00e-02 f	2.00e-02 j	
214Pb						2.00e-02 f	2.00e-02 j	
238Pu	3.00e-04 l	6.00e-05 l	3.00e-04 l	6.00e-05 l		2.10e-03 g,j	5.00e-04 g,j	
239Pu	3.00e-04 d	6.00e-05 p	3.00e-04 p	6.00e-05 p	9.12e-03 q	2.10e-03 g,j	5.00e-04 g,j	
239/240Pu						2.10e-03 g,j	5.00e-04 g,j	
223Ra	7.50e-02 d	7.50e-02 d	7.50e-02 l,r	7.50e-02 d	7.50e-02 e	4.50e-02 f	4.50e-02 g,j	
224Ra	7.50e-02 d	7.50e-02 d	7.50e-02 l,r	7.50e-02 d	7.50e-02 e	4.50e-02 f	4.50e-02 g,j	
226Ra	7.50e-02 d	7.50e-02 d	7.50e-02 r	7.50e-02 d	7.50e-02 e	4.50e-02 f	4.50e-02 g,j	
228Ra	7.50e-02 d	7.50e-02 d	7.50e-02 l,r	7.50e-02 d	7.50e-02 e	4.50e-02 f	4.50e-02 g,j	
90Sr	4.95e-01 d	1.60e-01 s	4.95e-01 s	4.95e-01 d		5.60e-02 g,j	4.00e-01 g,j	
228Th	9.00e-04 d	4.00e-04 l,p	9.00e-04 g,r	9.00e-04 g,r	5.00e-03 e	5.00e-03 f	5.00e-03 k	3.20e-05 l

Table 3. (continued)

Radionuclide	UF_{plants} [a]				UF_{invert} [a]	BAF_{bird} [b]	BAF_{mamm} [b]	UF_{mamm} [a]
	All plants	Grass	Herb. plants	Tree/shrubs				
230Th	9.00e-04 d	4.00e-04 l,p	9.00e-04 g,r	9.00e-04 g,r	5.00e-03 e	5.00e-03 f	5.00e-03 k	3.20e-05 l
232Th	9.00e-04 d	4.00e-04 p	9.00e-04 g,r	9.00e-04 g,r	5.00e-03 e	5.00e-03 f	5.00e-03 k	3.20e-05 p
234Th	9.00e-04 d	4.00e-04 l,p	9.00e-04 g,r	9.00e-04 g,r	5.00e-03 e	5.00e-03 f	5.00e-03 k	3.20e-05 l
208Tl	1.00e-03 c,g	1.00e-03 d	1.00e-03 d	1.00e-03 d	2.00e+00 e	2.00e+00 f	2.00e+00 c	
232U	1.97e+00 d	9.00e-04 l	3.75e-03 l,r	1.97e+00 l,t		7.00e-01 g,j	1.50e-02 j	3.20e-04 l
233U	1.97e+00 d	9.00e-04 l	3.75e-03 l,r	1.97e+00 l,t		7.00e-01 g,j	1.50e-02 j	3.20e-04 p
234U	1.59e+00 d	9.00e-04 l	3.75e-03 l,r	1.59e+00 t		7.00e-01 g,j	1.50e-02 j	3.20e-04 l
235U	1.97e+00 d	9.00e-04 l	3.75e-03 l,r	1.97e+00 l,t		7.00e-01 g,j	1.50e-02 j	3.20e-04 l
235/236U	1.97e+00 d	9.00e-04 l	3.75e-03 l,r	1.97e+00 l,t		7.00e-01 g,j	1.50e-02 j	3.20e-04 l
238U	1.97e+00 d	9.00e-04 p	3.75e-03 r	1.97e+00 t		7.00e-01 g,j	1.50e-02 j	3.20e-04 p

[a] Soil-tissue uptake factors (UF) for plants, soil invertebrates, and small mammals were obtained from available literature. When necessary, values originally reported on a dry-weight basis were converted to a wet-weight basis based on tissue water content.
[b] Bird and mammal bioaccumulation factors (BAFs, ratio of tissue activity to activity in food) were obtained from available literature. Values originally reported as biotransfer factors (d/kg) were converted to BAFs by multiplying d/kg by the ingestion rate of the test species. When necessary, values originally reported on a dry-weight basis were converted to a wet-weight basis based on tissue water content.
[c] Baes et al. (1984).
[d] Assumed the same as other plant types.
[e] Uptake factor for earthworms was unavailable. Used the larger of the plant and mammal values.
[f] Assumed mammal BAF because of lack of bird-specific values.
[g] Elemental form of the analyte was used for isotope.
[j] IAEA (1994).
[k] NCRP (1989).
[l] Assumed uptake same as reported for other isotope of the radionuclide (i.e., ^{137}Cs values used for ^{134}Cs).
[m] Garten (1980a).
[n] Trabalka and Garten (1983).
[o] Garten et al. (1986).
[p] Garten et al. (1987).
[q] Garten and Dahlman (1978).
[r] Bondietti et al. (1979).
[s] Garten and Lomax (1987).
[t] Garten (1980b).

2.2.3 Internal Exposures: Inhalation

Wildlife species using burrows may receive an additional internal dose from inhalation of dust originating from contaminated soil. Intake of radionuclide *i* by inhalation is estimated as (DOE 1995b)

$$D_{inh} = QF \quad F_{below} \sum C_{soil,i} \quad A \quad \frac{1}{AD} \quad \epsilon_i \; CFa \quad AF \;, \tag{12}$$

where

D_{inh}	=	internal dose rate from inhalation of contaminated soil (mrad/d),
F_{below}	=	dose reduction factor for fraction of time receptor spends below ground (unitless),
A	=	mass of respirable dust per volume of air breathed (0.1 g/m^3; DOE 1995b),
AD	=	air density (1200 g/m^3; Eckerman and Ryman 1993),
ϵ_i	=	α, β, or γ radiation energies for radionuclide *i* (MeV),
CFa	=	conversion factor to go from MeV to mrad g/pCi/d (5.12 x 10^{-2}),
AF	=	absorption factor (unitless).

Healy (1980) suggests that 0.0001 g/m^3 would be a conservative value when addressing human exposures to dust. Because burrowing animals are likely to spend a greater portion of their time in a confined space (burrow) than humans and are physically closer to the soil surface, an air mass loading of 0.1 g/m^3 is suggested as a conservative estimate of the mass of respirable dust (A) to which these animals may be exposed.

Total internal exposures are obtained by adding ingestion and inhalation dose rates over all radionuclides, including all short-lived daughter products.

2.2.4 Effects Levels for Radionuclides

The discharge of radioactive waste into the environment results in long-term, low-dose exposure to organisms. In most cases, acute mortality can be discounted. Any potential increase in morbidity and mortality that might result from the exposure to chronic irradiation above background is unlikely to be detected because of natural fluctuations in the size of populations.

The International Atomic Energy Agency (IAEA) recommends limiting the dose for terrestrial organisms to 100 mrad/d (IAEA 1992). Studies evaluating reproductive success and survival were used to determine the dose limit. Species-specific effects data were not available, so 100 mrad/d was selected as the threshold dose for all representative wildlife receptors. A dose rate of this magnitude is unlikely to cause observable changes in terrestrial animal populations (IAEA 1992). Higher dose rates may result in impaired reproduction or reduced survivorship. A dose rate of 1 rad/d is generally considered protective of plant and invertebrate populations (IAEA 1992, Barnthouse 1995) based on studies of productivity and community characteristics. This dose rate or less is unlikely to cause observable changes in terrestrial plant populations (IAEA 1992). Higher dose rates may result in reduced productivity or changes in species composition within communities. Therefore, 1 rad/d was selected as the threshold dose for effects on plant and invertebrate populations. Invertebrates tend to be less radiosensitive than plants or vertebrates, and indirect responses to radiation-induced vegetation changes (e.g., habitat alteration) appear more critical than direct effects (e.g., mortality, etc.) from radiation (IAEA 1992).

2.2.5 Uncertainties in Radiological Risk Assessment

A number of areas of uncertainty exist in the estimation of exposure and risks to terrestrial biota from exposure to radionuclides. The methodology outlined above is likely to overestimate dose rates that endpoints may receive. Whereas some of the information needed to implement the methodology is well known, much is unknown or unspecified statistically. A conservative but reasonable approach to model assumptions and radiological exposure scenarios was adopted to avoid underestimating risks to biota. Specific uncertainties identified in the radionuclide models are listed below.

- It is assumed that uptake of radionuclides from soil, food, and water are similar. Radionuclides bound to soil may be less available than those in tissue or water. Many radionuclides are poorly absorbed from soils (e.g., ^{137}Cs bound to clay minerals). Therefore, assuming uptake from soil equal to uptake from food may result in a conservative estimate of actual uptake.
- The dose coefficients obtained from Eckerman and Ryman (1993) used to estimate dose rates from external exposures are developed for application in determining dose rates to humans. These dose coefficients were applied directly for wildlife receptors or adjusted based on the effective height of the receptors, but the actual dose coefficients for wildlife, given differences in size, behavior, and general morphology, may be greater or less than those developed for humans.
- The air mass loading factor of 0.1 g/m^3 used in estimating exposures from inhalation of radionuclide-contaminated dust was selected as a conservative value. Healy (1980) suggested that 0.0001 g/m^3 would be a conservative value for estimating human exposures from inhalation of dust.
- The conversion factor used in the model for below-ground exposures was derived for small volumes of tissue (e.g., a mouse or shrew) immersed in soil assumed to be contaminated to an infinite depth. The actual dose for large animals or in cases where only the first few centimeters of soil are contaminated may be higher or lower. The simplifying assumptions used in the models presented here are generally applicable, but a health physicist could be consulted to develop specific dosimetry models where a more detailed evaluation is desired.
- Absorption factors are not available for many terrestrial organisms. The approach used here was to apply values developed for similar-sized aquatic organisms (Blaylock et al. 1993) or humans (Cristy and Eckerman 1987) to wildlife species. Because size and geometry of wildlife species do not exactly match those of aquatic organisms or humans, actual absorption fractions for wildlife species may be higher or lower than those suggested here.

2.3 PROBABILISTIC EXPOSURE ESTIMATION

Contaminant exposure estimates for wildlife are frequently generated using single, conservative values (e.g., upper 95% confidence limits on the mean, maximum observed value) to represent parameters (e.g., contaminant concentration in soil, food, water, or air; ingestion rates; or diet composition) in the exposure model. These single parameter values, known as point estimates, are selected because they are believed to be protective of most individuals and their use simplifies the calculation of an exposure estimate. While the use of conservative assumptions is suitable in a screening-level assessment, the use of point estimates is not recommended in a baseline or definitive assessment. Employing point estimates for the input parameters in the exposure model does not take into account the variation and uncertainty associated with the

parameters. Contaminant exposure that endpoints may receive in any given area may therefore be either over or underestimated. Consequently, remediation may be recommended for areas where it is unnecessary, or significant risks may be overlooked. Calculation of the exposure model using point estimates also produces only a point estimate of exposure. This exposure estimate provides no information concerning the distribution of exposures or the likelihood that individuals within an area will actually experience potentially hazardous exposures. To incorporate the variation in exposure parameters and to provide a better estimate of the potential exposure experienced by wildlife, it is highly recommended that exposure modeling be performed using probabilistic methods such as Monte Carlo simulation.

A detailed discussion of Monte Carlo simulation is beyond the scope of this report. General discussion of Monte Carlo techniques are provided by Rubenstein (1981) and Law and Kelton (1982). Briefly, Monte Carlo simulation is a resampling technique frequently used in uncertainty analysis in risk assessment (Hammonds et al. 1994). In practice, distributions are assigned to input parameters in a model, and the model output is recalculated many times to produce a distribution of output parameters (e.g., estimates of contaminant exposure). Each time the model is recalculated, a value is selected from within the distribution assigned for each input parameter. As a result, a distribution of exposure estimates is produced that reflects the variability of the input parameters. To determine which input parameters most strongly influence the final exposure estimate, a sensitivity analysis may be performed (Hammonds et al. 1994). Detailed discussions of sensitivity and uncertainty analysis, and the use of Monte Carlo simulations in risk assessment, are provided by Hammonds et al. (1994) and EPA (1996). Burmaster and Anderson (1994) outline 14 principles of good practice for the use of Monte Carlo techniques in risk assessment. Initial guidance for the use and interpretation of Monte Carlo analysis in risk assessment have been developed by the EPA Risk Assessment Forum (EPA 1997) and EPA Region 8 (EPA Region 8 1995). Examples of the application of Monte Carlo techniques in wildlife exposure and risk assessment are presented in MacIntosh et al. (1994), Sample et al. (1996b), and Moore et al. (In Press). Finally, a special issue of the journal *Human and Ecological Risk Assessment* (Vol. 2, No. 4, 1996) has recently been published to commemorate the 50th anniversary of the development of Monte Carlo methods. This issue will contain multiple papers on the application and interpretation of Monte Carlo methods. Software for conducting Monte Carlo simulations include @Risk (Palisade Corporation, Newfield, New York) and Crystal Ball (Decisioneering, Inc., Denver, Colorado).

2.4 EXTRAPOLATION FROM INDIVIDUALS TO POPULATIONS

Exposure models used in a risk assessment must be appropriate for the assessment endpoints considered. The models presented in previous sections are for estimation of exposure of individual organisms, but except for threatened and endangered species, wildlife endpoints are generally considered at the population level (Suter et al. 1995). Because exposure estimates must be integrated with exposure-response information, which is expressed as organism-level responses, the use of these organism-level exposure models is appropriate.

The conversion of individual-level exposure estimates to population-level effects occurs in the risk characterization and can be made in several ways. First, it may be assumed that there is a distinct population on the site so that the exposure of the population is represented by the exposure of all of the individuals. All individuals at the site are assumed to experience equivalent exposure. This assumption is appropriate for small organisms, with limited home ranges, on large sites, particularly if the site constitutes a distinct habitat that is surrounded by inappropriate habitat. For example, a grassy site surrounded by forest or industrial development might support a distinct population of voles. The risks to that population can be estimated directly from the exposures of the individual organisms.

Another approach is to assume that a certain number of individuals are exposed to contaminants out of a larger population. The proportion of the local population exposed at levels that exceed toxic thresholds represents the proportion of the population potentially at risk. This was the logic underlying the preliminary assessment for wide-ranging wildlife on the Oak Ridge Reservation (ORR; Sample et al. 1996b). On the ORR, while most habitat for wide-ranging wildlife species exists outside of source operable units (OUs; contaminated areas), some suitable habitat is present within source OUs. The proportion of the ORR-wide population potentially at risk is represented by the number of individuals that may use habitat on source OUs. The degree to which a source OU is used (and therefore the risk that it may present) is dependent upon the availability of suitable habitat on the OU. An estimate of risks to reservation-wide populations was estimated as follows.

1. Individual-based contaminant exposure estimates are generated for each source OU using the generalized exposure model (Eq. 5). Contaminant data, averaged over the entire OU, were used in the exposure estimate.

2. Contaminant exposure estimates for each OU were compared to Lowest Observed Adverse Effects Levels (LOAELs) from Sample et al. (1996a) to determine the magnitude and nature of effects that may result from exposure at the OU. If the exposure estimate >LOAEL, then individuals at the OU may experience adverse effects.

3. Availability and distribution of habitat on the ORR and within each OU, suitable for each species considered, was determined using a satellite-generated landcover map for the ORR (Washington-Allen et al. 1995).

4. Habitat requirements for the endpoint species of interest are compared to the ORR habitat map to determine the area of suitable habitat on the ORR and within OUs.

5. The area of suitable habitat on the ORR and within OUs was multiplied by species-specific population density values (ORR-specific or obtained from the literature) to generate estimates of the ORR-wide population and the numbers of individuals expected to reside within each OU.

6. The number of individuals for a given endpoint species expected to be receiving exposures >LOAELs for each measured contaminant was totaled. This is performed using the OU-specific population estimate from step 5 and the results from step 2. This number is then compared with the ORR-wide population to determine the proportion of the ORR-wide population that is receiving hazardous exposures.

This approach provides a very simple estimate of population-level effects. It is biased because it does not take wildlife movement into account. Wide-ranging species may travel among and use multiple OUs, therefore receiving exposures greater than that estimated for a single OU. In addition, the proportion of reservation-wide population potentially at risk is limited by the proportion of suitable habitat present in source OUs. For example, if 5% of the suitable habitat for a given species is located within OUs, the proportion of the population potentially at risk cannot exceed 5%.

A third approach is to combine the results of Monte Carlo simulation of exposure with literature-derived population density data to evaluate the likelihood and magnitude of population-level effects

on wildlife. The number of individuals within a given area likely to experience exposures >LOAELs can be estimated using cumulative binomial probability functions (Dowdy and Wearden 1983). Binomial probability functions are estimated using the following equation

$$b(y;n;p) = \binom{n}{y} p^{y} (1-p)^{n-y} , \qquad (13)$$

where

y	=	the number of individuals experiencing exposures >LOAEL,
n	=	total number of individuals within the watershed,
p	=	probability of experiencing an exposure in excess of the LOAEL,
b (y; n; p)	=	probability of y individuals out of a total of n, experiencing an exposure >LOAEL, given the probability that exceeding the LOAEL = p.

By solving Eq. 13 for y = 0 to y = n, a cumulative binomial probability distribution may be generated that can be used to estimate the number of individuals within an area that are likely to experience adverse effects. This approach was used to estimate the risks that PCBs and mercury in fish presented to the population of piscivores in watersheds on the ORR (Sample et al. 1996b). Monte Carlo simulations were performed to estimate watershed-wide exposures. It was assumed that wildlife were more likely to forage in areas where food is most abundant. Density or biomass of fish at or near locations where fish bioaccumulation data were collected were assumed to represent measures of food abundance. (Biomass data were preferred but were unavailable for all watersheds. Where unavailable, density data were used.) The relative proportion that each location contributed to overall watershed density or biomass data was used to weight the contribution to the watershed-level exposure. The watershed-level exposure was estimated to be the weighted average of the exposure at each location sampled within the watershed. In this way, locations with high fish densities or greater fish biomass contribute more to exposure than do locations with lower density or biomass. Because the watersheds were large enough to support multiple individuals, the weighted average exposure estimate was assumed to represent the exposure of all individuals in each watershed. While simplistic, this approach is believed to provide a better estimate of population-level effects than the previously described method. Use of this method, however, requires exposure data from multiple, spatially disjunct areas and data suitable to weight the potential exposure at each area.

Freshman and Menzie (1996) present an additional approach for extrapolating to population-level effects. Their Population Effects Foraging (PEF) model estimates the number of individuals within a local population that may be adversely affected. The PEF model is an individual-based model that allows animals to move randomly over a contaminated site. Movements are limited by species-specific foraging areas and habitat requirements. The model estimates exposures for a series of individuals, and then sums the number of individuals that receive exposures in excess of toxic thresholds (Freshman and Menzie 1996).