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October 11, 1996

Joe Vozella, Chief
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Dear Mr. Vozella

Subject: Incorporation of radionuclides into the Ecological Risk Assessment (ERA) process

At the Ecological Risk Assessment workshop/course presented in September, 1996, discussions centered on the recent paper submitted by Los Alamos National Laboratory (LANL) to New Mexico Environment Department (NMED) and Environmental Protection Agency (EPA) titled *Ecological Risk Assessment Approach for Los Alamos National Laboratory*. At one point, the incorporation of radionuclides into the ERA process was discussed.

A DOE Environmental Restoration Group representative suggested that the DOE Oversight Bureau present a "white paper", justifying the inclusion of radionuclides into the ERA process. An attachment has been included with this letter, which documents available resources, experiences of other agencies and pertinent data-bases that you may wish to incorporate into your risk assessment process.

In his October 1, 1996, presentation to the Operations Working Group at LANL, Jim Piatt, Bureau Chief of the Surface Water Quality Bureau, expressed his concern that DOE/LANL needed to "adequately address" off-site discharges of radionuclides.

The incorporation of radionuclides into the ERA process will help insure that the ERAs produced by DOE are considered complete by NMED and other stakeholders. This should result in the considerable savings of time and money as slippage of schedules could occur if DOE needed to go back and re-assess their risk management decisions.

Data Quality Objectives (DQOs) should address data gaps and uncertainties early in the process

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to gather the information needed to complete an ERA. Characterization of stormwater quality along with sampling of additional ecologically pertinent media should be initiated at the onset.

Please feel free to contact Steve Yanicak at 672-0448 or Ralph Ford-Schmid of my staff at 827-1536 for further discussions regarding this issue.

Sincerely,

Neil Weber,
Chief, DOE Oversight Bureau

Enclosures

NW:rfs

cc: w/o enclosure
Ed Kelley, Director, NMED, WWMD

w/enclosure
Jim Piatt, Chief, NMED, SWQB
Benito Garcia, Chief, NMED, HRMB
Marcy Leavitt, Chief, NMED, GRPRB
Steve Yanicak, POC, DOE OB/LANL

Incorporation of Radionuclides into the Ecological Risk Assessment Process

Ralph Ford-Schmid
DOE Oversight Bureau

INTRODUCTION

Ecological Risk Assessment (ERA) methodology is currently under development at Department of Energy (DOE) facilities in New Mexico. During the early stages of this process is the appropriate time for dialog between the Laboratories, U. S. Environmental Protection Agency (EPA), and the New Mexico Environment Department (NMED) to determine appropriate measurement and assessment endpoints for effects of radionuclides and hazardous chemicals on ecological receptors.

Effective communication of the ecological risk posed by these constituents to NMED and other stakeholders (Indian tribes, government agencies and the general public) will in part, depend upon the thoroughness of the ERA process used. At a minimum, radioisotopes need to pass a risk screening process before being considered for elimination from further risk assessment.

BACKGROUND

An ERA workshop/course was presented by EPA Region 6 in Santa Fe, NM. in September, 1996. The workshop was centered on the recent "discussion" paper submitted by Los Alamos National Laboratory (LANL) to NMED and EPA titled *Ecological Risk Assessment Approach for Los Alamos National Laboratory* (R. W. Ferenbaugh, and others). The authors position concerning the assessment of risk posed by radionuclides was:

> For radionuclides, International Atomic Energy Agency guidance (IAEA 1992) will be followed. This guidance states that "the level of safety required for the protection of all human individuals is thought likely to protect other species, although not necessarily individual members of those species." "if man is adequately protected, then other living things are also likely to be sufficiently protected." Thus, ecological risk assessment at LANL will not address radionuclides except in situations where no human health risk assessment has been performed.

Though it is generally assumed that humans are comparatively radiosensitive and that guidelines that protect humans, probably also protect sensitive natural resources, this needs verification. The land use scenario used for the human-health risk assessments play an important role in the justification of the above assumption. A human health risk assessment using a residential land use scenario should better approximate ecological risk than an industrial land use scenario due to the additional exposure assumptions encountered with the more conservative approach.

Representatives of NMED expressed concern that the dismissal of radionuclides from the ERA

process would eliminate major Constituents of Concern (COCs) from the assessment process and could result in incomplete ERAs.

The radiation dose, below which is expected to have no adverse effects on aquatic organism, is 1-rad/day (NCRP 1991). For terrestrial organisms, a radiation exposure of 0.1 rad-day is not considered harmful (IAEA 1992). An estimate of radiation dose, expected to occur to terrestrial and aquatic organisms inhabiting DOE facilities (and off-site), in necessary to determine potential ecological risk. Source terms used for COCs in soils will be the same as those used for the human health risk assessment. The estimation of radiation doses to ecological receptors are calculated for any specified contaminant and/or receptor and in some cases can be directly derived from soil concentrations.

RECOMMENDATIONS

In consultation with other stakeholders, an ERA approach should be developed which addresses all constituents of concern, including radionuclides. The assessment of the effects of radionuclides on ecological receptors is not a new concept and has been addressed at other DOE facilities. The expertise developed by other DOE facilities (ORNL, INEL, Hanford, and PNL) should be incorporated into the ERA methodologies currently under development at DOE facilities in New Mexico.

The following documents should be used as a starting point to frame discussions on an appropriate ERA approach for radionuclides at LANL.

The *Draft Hanford Remedial Action Environmental Impact Statement and Comprehensive Land Use Plan* (HRA-EIS) (DOE 1996) outlines and describes the steps and calculations used at Hanford. This can be accessed electronically and Section C.0, Ecological Risk Assessment Methodology and Calculations is included with this paper as Appendix A.

The *Draft Guidance Manual for Conducting Screening Level Ecological Risk Assessments at the INEL* (EGG-ER-11321) outlines and describes the steps and calculations used at INEL. Section 3, pages 62 - 69, are included with this paper as Appendix B.

Another method that may be appropriate for consideration is the use of human risk Screening Action Levels (SALs) for radionuclides as Ecotoxicological Screening Action Levels (ESALs). This method has been used in an ecotoxicological screen of Potential Release Site 50-006(D) of Operable Unit 1147 of Mortandad Canyon (Gonzales and Newell 1996) and is effective as a conservative screening tool.

Additional sources for information include:

AQUATIC BIOTA:
Suter, G. W. 11, and C. L. Tsao. 1996. Toxicological Benchmarks for Screening of Potential Contaminants of Concern for Effects on Aquatic Biota on Oak Ridge Reservation-, 1996

Revision. Oak Ridge National Laboratory, Oak Ridge, TN. 104 pp, ES/ER/TM-96/R2

WILDLIFE:
Sample, B.E., D.M. Opresko, and G.W Suter 11. 1996. Toxicological Benchmarks for Wildlife-. 1996 Revision. Oak Ridge National Laboratory, Oak Ridge, TN. 227 pp, ES/ER/TM-86/R3

TERRESTRIAL PLANTS
Will, M.E. and G.W. Suter 11. 1995. Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Terrestrial Plants: 1995 Revision. Oak Ridge National Laboratory, Oak Ridge, TN. 123 pp, ES/ER/TM-85/R2

SEDIMENTS
Jones, D. S., R.N. Hull, and G.W. Suter 11. 1996. Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Sediment-Associated Biota: 1996 Revision. Oak Ridge National Laboratory, Oak Ridge TN. 34 pp, ES/ER/TM-95/R2

SOIL INVERTEBRATES AND MICROBIAL PROCESSES:
Will, M.E and G.W. Suter 11. 1995. Toxicological Benchmarks for Potential Contaminants of Concern for Effects on Soil and Litter Invertebrates and Heterotrophic Processes. Oak Ridge National Laboratory, Oak Ridge TN. 155 pp, ES/ER/TM-126/Rl

R. Eisler, 1994, *Radiation Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review* The U.S. Department of the Interior, National Biological Service, Biological Report 26

NCRP - National Counsel on Radiation Protection and Measurement - publications

Effects of Ionizing Radiation on Aquatics. NCR 109

Aspects of Sr Radiobiology. NCR 110

Health and Ecological Implications of Radioactively Contaminated Environment. Proceedings Annual Meeting # 12

CONCLUSIONS:

Effective communication of ecological risk posed by past and future operations of DOE facilities necessitates a consensus approach to the development of ecological risk assessment methodologies. This process should include input from stakeholders to develop appropriate measurement and assessment endpoints for the determination of potential effects of radionuclides and hazardous chemicals on ecological receptors.

REFERENCES

Ferenbaugh, R.W., O.B. Myers, M.H. Ebinger, A.F. Gallegos, and D.D. Breshears, 1996. Ecological Risk Assessment Approach for Los Alamos National Laboratory. LA-UR-96-766

Gonzales, G. J. and P. G. Newell, 1996. Ecotoxicological Screen of Potential Release Site 50-006(D) of Operable Unit 1147 of Mortandad Canyon and Relationship to the Radioactive Liquid Waste Treatment Facilities Project. LA - 13148 - MS, UC-940 and UC-7-21

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

NCRP, 1991. *Effects of Ionizing Radiation on Aquatic Organisms*, NCRP 109, National Council on Radiation Protection and Measurements, Bethesda, Maryland

APPENDIX A

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C.0 Ecological Risk Assessment Methodology and Calculations

This appendix describes the methods that were used to estimate radiation doses and hazard indices (HI) associated with the intake of hazardous chemicals by ecological receptors at the Hanford Site. Detailed results of the risk assessment are presented and are summarized by geographic area in Chapter 5.0 of the *Draft Hanford Remedial Action Environmental Impact Statement and Comprehensive Land Use Plan* (HRA-EIS) (DOE 1996).

C.1 Ecological Risk Assessment Methodology

The general method used in the HRA-EIS follows the approach recommended in the *Hanford Site Baseline Risk Assessment Methodology (HSBRAM)*[1] (DOE-RL 1993). The basic steps used in ecological risk assessment are (1) problem formulation, (2) characterization of potential exposures and associated ecological effects, and (3) comprehensive presentation of the results as a risk characterization (EPA 1992). This section describes these steps in terms specific to the HRA-EIS.

C.1.1 Problem Formulation

The Hanford Site supports a variety of arid terrestrial habitats; a major aquatic habitat (the Columbia River); and a number of threatened, endangered, or candidate species. For purposes of the HRA-EIS, these habitats and adjacent offsite habitats constitute the ecosystem potentially at risk. Contaminants of potential concern include a wide variety of radionuclides and hazardous chemicals. Ecological receptors may be exposed to these substances in surface and subsurface soils, ground and surface water, and uptake through the food chain. Effects from exposure can be chronic (i.e., long-term, low-level exposure that may cause latent damage that does not appear until later) or acute (i.e., direct mortality) depending on the magnitude and frequency of exposure.

Preparation of an ecological risk assessment requires selection of assessment and measurement endpoints. Assessment endpoints are the specific ecological characteristics to be protected, and measurement endpoints are characteristics that can be measured and correspond in some way to the assessment endpoints. In the HRA-EIS, the assessment endpoint for effects of radionuclides and hazardous chemicals is prevention of adverse effects from these substances on ecological receptors. The measurement endpoints that correspond to this assessment endpoint are (1) estimated radiation dose, which is compared to the 1-rad/day expected to have no adverse effects on aquatic organisms (NCRP 1991) and the 0.1-rad/day below which radiation exposure is not considered harmful to terrestrial organisms (IAEA 1992), and (2) the ratio of estimated intake of hazardous chemicals compared to the intake expected to have a No

the ratio of estimated intake of hazardous chemicals compared to the intake expected to have a No Observable Adverse Effect Level (NOAEL). The latter ratio is called the HI; a value greater than 1.0 indicates a potential for adverse effects.

C.1.2 Conceptual Model

Estimation of potential radiation and hazardous chemical effects requires a conceptual model that illustrates transport of constituents of potential concern to ecological receptors. The conceptual model used in this analysis is a simple food chain consisting of three wildlife species and a generic plant (Figure C-1). The wildlife species used are the Great Basin pocket mouse, the coyote, and the red-tailed hawk. The single exposure pathway considered is plant uptake from soil followed by ingestion of vegetation by the pocket mouse, and ingestion of mice by the coyote and the hawk. This model was designed to assess the effects at several trophic levels (primary producer, herbivore, and mammalian and avian carnivore), while being simple enough to efficiently assess potential effects at the many waste sites within the scope of the HRA-EIS. Hazards to aquatic receptors were assessed by estimating radiation doses from surface water radionuclide concentrations.

Figure C-1. Conceptual Model for Estimating Hazard Indices and Radiation Doses to Ecological Receptors.

C.1.3 Source Terms

Source terms for constituents of concern in soils were the same as those used for the human health risk assessment, which is described in Appendix B of the HRA-EIS. The maximum estimated concentration at time zero (1989) was used throughout the analysis. For a number of chemicals, data were provided for a compound where only one element in the compound was of concern for possible adverse effects. For example, the element of concern in sodium dichromate is chromium. In these cases, the concentration of the specific element was computed based on the atomic weights of the elements making up the compound. Chemical concentrations used to estimate HIs are listed in Table C-1; radionuclide concentrations used to estimate radiation doses are listed in Table C-2. Volume-based concentrations were converted to weight-based concentrations (mg/kg for chemicals, pCi/g for radionuclides), assuming a soil density of 1.76 g/cm^3.

C.1.4 Exposure Assessment

This section describes the general methods that were used to estimate intake of hazardous chemicals, the associated HIs, and the radiation doses resulting from radionuclide intake. This section also describes the equations typically presented in the risk assessment literature, and then describes the modifications used to calculate unit risk factors (URF) in order to simplify computations. Application of the URFs to ecological receptors results in an estimated radiation dose or chemical HI rather than a probability of some adverse effect as in the human health risk assessment. However, the term "URF" is maintained here for purposes of consistency with the similar methodology used for the human health risk assessment presented in Appendix B of the HRA-EIS.

C.1.4.1 Estimation of Hazardous Chemical Intake. Contaminant uptake by a generic plant was estimated by multiplying the contaminant concentration in the soil by the soil-to-plant concentration factors used in *GENII--The Hanford Environmental Radiation Dosimetry Software System* (GENII model) at the Hanford Site (Napier et al. 1988). The equation used was:

$$C_{vi} = (C_{si})(B_{vi})(0.4) \quad (1)$$

where:

C_{vi} = Contaminant concentration in plant (mg/kg wet weight)

C_{si} = Contaminant concentration in soil (mg/kg dry weight)

B_{vi} = Soil-to-plant concentration factor (unitless). (The factor for grain concentration was used, rather than for vegetative portions; the pocket mouse was assumed to consume seeds)

0.4 = Wet weight/dry weight conversion (DOE-RL 1994).

The intake rate of hazardous chemicals by a herbivore through consumption of plants was calculated using the following equation:

$$I_i = (C_{vi})(IR)(FI)/(BW) \quad (2)$$

where:

I_i = Intake rate of the i_{th} contaminant (mg/kg/day)

C_{vi} = Contaminant concentration in plant (mg/kg)

IR = Ingestion rate of food (kg/day wet weight)

FI = Fraction ingested from contaminated source (unitless)

BW = Body weight (kg wet weight).

Intake rates by carnivores are calculated similarly, substituting the contaminant concentrations in the herbivore for the concentrations in plants. Contaminant concentrations in herbivore muscle are estimated using Equation 3.

$$C_{mi} = (C_{vi})(IR)(FI)(B_{mi}) \quad (3)$$

where:

C_{mi} = Contaminant concentration in muscle (mg/kg wet weight)

C_{vi} = Contaminant concentration in plant (mg/kg wet weight)

IR = Ingestion rate of plants by herbivore (kg/day)

FI = Fraction ingested from a contaminated source (unitless)

B_{mi} = Plant-to-muscle transfer factor (day/kg). (Plant-to-beef transfer factors used in the GENII model at the Hanford Site were used where available [Napier et al. 1988], or from *A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides*

Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides Through Agriculture [Baes et al. 1984] where not). The transfer factors used to calculate URFs for chemicals are identified in Table C-3.

Ingestion rates and body weights used in calculating URFs for the HRA-EIS are listed in Table C-4.

C.1.4.2 Calculation of Hazard Indices. Hazard indices (the ratio of estimated contaminant intake rate to the exposure expected to have no adverse effect) were calculated by dividing the intake rate by the NOAEL (Equation 4).

$$HI = I/NOAEL \quad (4)$$

where I is calculated as described in Equation 2. NOAELs for contaminants of concern were obtained from the scientific literature, and were scaled to the body weight of the receptor as described below. Both factors are expressed as mg/kg body weight per day.

An HI greater than 1.0 for a given chemical indicates that estimated intake exceeds the threshold "safe" level. For sites with multiple chemicals present, the HIs may be summed, making the conservative assumption that the modes of action and target organs of the chemicals are similar. Therefore, a site may be said to present a hazard if the sum of HIs exceeds 1.0, even if the individual chemical HIs are less than 1.0. This approach was modified to allow calculation of HIs directly from soil concentrations, without the necessity of separate calculations of plant uptake from soil, plant consumption by the mouse, and so forth, for each Hanford Site location.

NOAELs were obtained, in order of preference, from the *Toxicological Benchmarks for Wildlife* (Opresko et al. 1993) or the U.S. Department of Energy (DOE-RL 1994). Wildlife NOAELs obtained (Opresko et al. 1993) were scaled to the body weight of the pocket mouse or coyote using the equation:

$$NOAEL_y = (NOAEL_x)[(bw_x)/(bw_y)]^{1/3} \quad (5)$$

where:

$NOAEL_y$ = NOAEL for the organism of interest

$NOAEL_x$ = NOAEL for the experimental animal available from the literature

bw_y = Body weight of the organism of interest

bw_x = Body weight of the experimental animal with the known NOAEL.

Wildlife NOAELs obtained from the *Remedial Investigation and Feasibility Study Report for the Environmental Restoration Disposal Facility* (DOE-RL 1994) were used directly, as listed, without attempting to scale them to body weight. Red-tailed hawk NOAELs were taken directly from the values for this species in the *Toxicological Benchmarks for Wildlife* (Opresko et al. 1993) or as the avian NOAELs from DOE-RL (1994). In some cases NOAELs were reported for compounds where a specific element in the compound was the one of primary concern for adverse effects (Opresko et al. 1993). In these cases, a NOAEL for the specific element was computed based on the atomic weights of the elements comprising the compound. This procedure was directly analogous to the procedure used to estimate element-specific contaminant concentration, as described in Section C.1.3. Scaling factors used to adjust

NOAELs are listed in Table C-5, (Opresko et al. 1993), and NOAELs and data sources are listed in Table C-6. NOAELs for plants were obtained as soil concentrations from the *Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Terrestrial Plants* (Suter et al. 1993), and HIs were calculated as the waste unit soil concentration divided by the NOAEL. It was, therefore, not necessary to calculate URFs for plants.

C.1.4.3 Calculation of Unit Risk Factors for Chemicals. Unit risk factors were calculated to allow computation of the HI for any specified contaminant and/or receptor that was directly derived from the soil concentration.

C.1.4.3.1 Pocket Mouse Unit Risk Factors. Expanding Equation 4 for estimation of the HI for the pocket mouse consuming the chemical of interest:

$$HI_i = I_i/NOAEL_i \quad (6)$$

$$= [(C_{vi})(IR)(FI)/(BW)]/NOAEL; \text{ expanding } C_{vi},$$

$$= [(C_{si})(B_{vi})(0.4)(IR)(FI)/(BW)]/NOAEL$$

The URF was defined as:

$$URF_i = NOAEL_i/[(B_{vi})(0.4)(IR)(FI)/(BW)] \quad (7)$$

such that:

$$HI_i = C_{si}/URF_i. \quad (8)$$

The fraction ingested (FI) was assumed to be 1.0 for all pocket mouse, coyote, and hawk calculations. This simplifies Equation 7, substituting in the ingestion rate and body weight for the pocket mouse (see Table C-4) to:

$$URF_i = NOAEL_i/[(B_{vi})(0.4)(0.0067 \text{ kg/d})(1.0)/(0.0235 \text{ kg})] \quad (9)$$

$$= NOAEL_i/[(B_{vi})(0.4)(0.2851/d)].$$

C.1.4.3.2 Coyote and Red-Tailed Hawk Unit Risk Factors. A similar process was used to calculate URFs for the coyote and red-tailed hawk. Calculation of factors for these receptors required using the chemical concentration in mouse muscle, C_{mi}, as an input variable; the calculations are presented in Equations 10 through 14.

$$HI_i = I_i/NOAEL_i \quad (10)$$

$$= [(C_{mi})(IR)(FI)/(BW)]/NOAEL; \text{ expanding } C_{mi} \text{ (see Equation 3)},$$

$$HI_i = [(C_{vi})(IR_{mouse})(B_{mi})(IR_{coyote})/(BW_{coyote})]/NOAEL; \text{ expanding } C_{vi},$$

$$HI_i = [(C_{si})(B_{vi})(0.4)(IR_{mouse})(B_{mi})(IR_{coyote})/(BW_{coyote})]/NOAEL.$$

As above, the URF was defined as:

$$URF_i = NOAEL_i/[(B_{vi})(0.4)(IR_{mouse})(B_{mi})(IR_{coyote})/(BW_{coyote})] \quad (11)$$

such that

$$HI_i = C_{si}/URF_i. \quad (12)$$

Incorporating the ingestion rate for the pocket mouse and the ingestion rate and body weight for the coyote (see Table C-4):

$$URF_i = NOAEL_i/[(B_{vi})(0.4)(0.0067\ kg/d)(B_{mi})(1.3\ kg/d)/(10\ kg)] \quad (13)$$

$$= NOAEL_i/[(B_{vi})(0.4)(B_{mi})(0.000871/d)].$$

The calculation for the hawk is similar, substituting the ingestion and body weights of 1.1 kg/d and 1.0 kg, respectively, into Equation 13,

$$URF_i = NOAEL_i/[(B_{vi})(0.4)(B_{mi})(0.00737/d)]. \quad (14)$$

The resulting URFs used to calculate HIs for the pocket mouse, coyote, and red-tailed hawk are listed in Table C-7.

C.1.4.4 Estimation of Radiation Doses to Ecological Receptors. Radiation doses to ecological receptors were calculated using URFs similar to those discussed for chemicals. The basic equation used was:

$$Dose\ (rad/day) = (5.11 \times 10^{-5})(MeV)(C) \quad (15)$$

where:

5.11×10^{-5} = Constant (rad d^{-1} pCi^{-1} g MeV^{-1} disintegration)

MeV = Mean energy of decay (MeV per disintegration)

C = Radionuclide concentration in organism (pCi/g dry weight).

The constant was derived as follows:

$$5.11 \times 10^{-5} = (A)(B)(C)(D)(E)(F)(G)$$

where:

$A = 1\ Ci/10^{12}\ pCi$

$B = 3.7 \times 10^{10}$ disintegrations/Ci-sec

C = 3,600 sec/hr

D = 24 hr/d

E = 10^6 eV/MeV

F = 1.6×10^{-12} erg/eV

G = 1 rad-g/100 ergs.

C.1.4.4.1 Derivation of Unit Risk Factors for Radiation Doses to Plants. Equation 15 written for a generic plant is:

$$D_{Pr}\ (\text{rad/d}) = (5.11 \times 10^{-5})(MeV_{Pr})(C_{Pr})\ (16)$$

where:

MeV_{Pr} = Average effective energy of decay of radionuclide r in the plant

C_{Pr} = Concentration of radionuclide r in plant (pCi/g = $[C_{sr}][B_{vr}]$), where

C_{sr} = Concentration of radionuclide r in soil (pCi/g)

B_{vr} = Soil-to-plant concentration factor (unitless) for radionuclide r.

Expanding Equation 16 to express radiation dose as a function of soil concentration,

$$D_{Pr}\ (\text{rad/d}) = (5.11 \times 10^{-5})(MeV_{Pr})(C_{sr})(B_{vr}).\ (17)$$

The URF (K_{Pr} to avoid confusion with the chemical URFs) for calculating the radiation dose to a plant caused by uptake from soil is:

$$K_{Pr} = (5.11 \times 10^{-5})(MeV_{Pr})(B_{vr}).\ (18)$$

The dose Equation 17 is simplified to:

$$D_{Pr}\ (\text{rad/d}) = (C_{sr})(K_{Pr}).\ (19)$$

C.1.4.4.2 Derivation of Unit Risk Factors for Radiation Doses to the Pocket Mouse. The derivation of the URF for radiation dose to the pocket mouse is similar to the derivation of URFs for the chemical HI for the pocket mouse and the radiation dose to the plant. Following Equation 16, the estimated dose is:

$$D_{Mr}\ (\text{rad/d}) = (5.11 \times 10^{-5})(MeV_{Mr})(C_{Mr}),\ (20)$$

where the terms are defined in the same manner as for the plant (Section C.1.4.4.1) and C_{Mr} (the radionuclide concentration in the mouse) is calculated using Equation 3, but expressing the soil concentration in pCi/g and multiplying the right side of the equation by 0.001 kg/g to adjust for the

difference in units (chemical concentrations are expressed in mg/kg), yielding Equation 21.

$$C_{Mr} = (C_{Pr})(IR)(FI)(B_{Mr})(0.4)(0.001) \quad (21)$$

where:

C_{Mr} = Contaminant concentration in mouse (pCi/g wet weight)

C_{Pr} = Contaminant concentration in plant (pCi/g dry weight)

IR = Ingestion rate of plants by herbivore (g/day)

FI = Fraction ingested from a contaminated source (unitless) = 1.0

B_{Mr} = Plant-to-muscle transfer factor (day/kg). (Plant-to-beef transfer factors used in the GENII model at the Hanford Site were used where available [Napier et al. 1988], or from *A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides Through Agriculture* [Baes et al. 1984] where not)

0.4 = wet weight/dry weight conversion (DOE-RL 1994)

0.001 = kg/g (conversion factor for plant-to-beef transfer factors, which have units of days/kg).

Expanding Equation 20,

$$D_{Mr} = (C_{Pr})(IR_M)(FI_M)(B_{Mr})(5.11 \times 10^{-5})(MeV_{Mr}), \quad (22)$$

and expanding C_{Pr} using Equation 1,

$$D_{Mr} = (C_{sr})(B_{vr})(0.4)(IR_M)(FI_M)(B_{Mr})(5.11 \times 10^{-5})(MeV_{Mr}). \quad (23)$$

Following the reasoning in Equations 17 through 19, the URF K_{Mr} for calculating the radiation dose to a pocket mouse from consumption of plants growing in contaminated soil is:

$$K_{Mr} = (5.11 \times 10^{-5})(MeV_{Mr})(B_{vr})(0.4)(IR_M)(FI_M)(B_{Mr}). \quad (24)$$

The dose equation is simplified to:

$$D_{Mr}\ (rad/d) = (C_{sr})(K_{Mr}). \quad (25)$$

As with chemicals, FI was assumed to be 1.0 for the pocket mouse, coyote, and hawk.

C.1.4.4.3 Derivation of Unit Risk Factors for Radiation Doses to the Coyote and Hawk. The derivation of risk factors for the coyote and the hawk was similar to derivation of risk factor for radiation dose for the pocket mouse and the chemical HI for the coyote. The concentration of radionuclides in the coyote was calculated as:

$$C_{Cr} = (C_{Mr})(IR)(FI)(B_{mmr}) \quad (26)$$

where:

C_{Cr} = Radionuclide concentration in coyote (pCi/g wet weight)

C_{Mr} = Contaminant concentration in pocket mouse (pCi/g wet weight, calculated in Equation 21)

IR = Ingestion rate of mice by coyote (kg/day)

FI = Fraction ingested from a contaminated source (unitless)

B_{mmr} = Muscle-to-muscle transfer factor (assumed to be 1.0).

The resulting equation for radiation dose, expanding Equation 21 is:

$$D_{Cr} = (C_{sr})(B_{vr})(0.4)(IR_M)(B_{Mr})(5.11 \times 10^{-5})(MeV_{Cr})(IR_C) \quad (27)$$

The URF K_C is:

$$K_C = (B_{vr})(0.4)(IR_M)(B_{Mr})(5.11 \times 10^{-5})(MeV_{Cr})(IR_C). \quad (28)$$

The URF for the hawk was calculated using the same equation, substituting MeV_{Hr} and IR_H for MeV_{Cr} and IR_C, respectively.

Unit risk factors for radiation doses calculated in this manner are listed in Table C-8. The soil-to-plant transfer factors used in the GENII model for the Hanford Site were used in the analysis (Napier et al. 1988). Organism-specific MeVs were obtained from the *Methods for Estimating Doses to Organisms from Radioactive Materials Released Into the Aquatic Environment* (Baker and Soldat 1992), assuming radii of 1.4, 2, 30, and 5 cm (0.6, 0.8, 12, and 2 in.) for the generic plant, pocket mouse, coyote, and red-tailed hawk, respectively.

C.1.4.4.4 Estimation of Radiation Dose to Aquatic Receptors. Mean and maximum radiation doses to aquatic receptors were estimated for a generic aquatic plant, invertebrate, fish, and the muskrat using the constants provided in *A Methodology for Calculating Radiation Doses from Radioactivity Released to the Environment* (Killough and McKay 1976) and the same source terms used in the human health risk assessment presented in Appendix B of the HRA-EIS. Concentrations used in the analysis are listed in Tables C-9 and C-10.

C.2 Results

Estimated HIs and radiation doses to ecological receptors are presented in Tables C-11 through C-23. Results are presented for four geographic areas (Columbia River, Central Plateau, Reactors on the River, and All Other Areas) and for each grid cell within the geographic areas. Division of the Hanford Site into grid cells for purposes of risk assessment is discussed in Appendix B of the HRA-EIS. The results are presented for each individual chemical or radionuclide, by cell, within each geographic area. Tables C-12,

C-14, C-16, C-18, C-20, and C-22 provide the results summed by cell for all chemicals or radionuclides within the cell (i.e., the total HI or radiation dose resulting from contaminants within that grid cell). Table C-23 presents the estimated radiation doses to aquatic receptors. This appendix does not assess the significance of these calculations.

C.3 Uncertainties

A quantitative uncertainty analysis was not undertaken because of the level of effort required to assess and present the analysis for each grid cell on the Hanford Site. Source data is probably the primary contributor to uncertainty in calculating both the HIs and the radiation doses, as described in Appendix B of the HRA-EIS. Additional source data could either increase or decrease the estimated hazards; the maximum chemical and radionuclide concentrations were used for all analyses so that the estimate would represent an upper-bound value. Secondary contributors to uncertainty are considered to be the conceptual model and the transfer factors used to estimate plant uptake and assimilation in the pocket mouse. The conceptual model does not include potential dose contributions from water ingestion, inhalation, or direct radiation exposure. Inclusion of these pathways would increase the estimated radiation doses and chemical HIs. The model also assumes that organisms obtain all food from contaminated sources. Accounting for home range size relative to grid cell size would decrease the estimated risks for the coyote and hawk, which range over a larger area than the 1-km^2 (0.39-mi^2) grid cell area.

Additional data on transfer factors could either increase or decrease the estimated hazards; again, use of the maximum concentrations, essentially assuming direct uptake from the waste, was intended to ensure that the hazard estimates are an upper bound. Additional secondary contributors to uncertainty are the NOAELs for hazardous chemicals, and the assumption that radionuclides have infinite half-lives in the environment and are perfectly assimilated by the hawk and coyote. Although NOAEL estimates could be too high (making the HI too low), additional data (or calculations accounting for radiological decay) in either of these areas probably would reduce the estimates of hazard. Data regarding body weight and ingestion rates are considered to be minor contributors to uncertainty, because estimates of these parameters are better than estimates of the other variables. The assumption that the fraction ingested from a contaminated source equals 1.0 is a contributor to the conservatism of the hazard estimates rather than a source of uncertainty. Reduction of this value for a given receptor would reduce the HI or radiation dose.

Table C-1. Chemical Concentrations Used as Input to Hazard Index Calculations.

Table C-2. Radionuclide Source Term Concentrations Used to Estimate Radiation Dose.

Table C-3. Transfer Factors Used to Calculate Unit Risk Factors for Chemicals.

Table C-4. Body Weights and Ingestion Rates of Ecological Receptors.

Table C-5. Scaling Factors for Adjusting Wildlife No Observed Adverse Effect Levels for Body Size.

Table C-6. No Observed Adverse Effect Levels for Chemical Intake by Ecological Receptors.

Table C-7. Unit Risk Factors for Calculating Chemical Hazard Indices.

Table C-8. Unit Risk Factors and Associated Parameters Used to Calculate Radiation Doses.

Table C-9. Water Concentrations Used to Calculate Maximum Radiation Doses to Aquatic Organisms.

Table C-10. Water Concentrations Used to Calculate Mean Radiation Doses to Aquatic Organisms.

Table C-11. Hazard Index, Chemical by Cell, Reactors on the River.

Table C-12. Hazard Index, Summed by Cell, Reactors on the River.

Table C-13. Estimated Radiation Dose, Radionuclide by Cell, Reactors on the River.

Table C-14. Estimated Radiation Dose, Total by Cell, Reactors on the River.

Table C-15. Hazard Index, Chemical by Cell, Central Plateau.

Table C-16. Hazard Index, Summed by Cell, Central Plateau.

Table C-17. Estimated Radiation Dose, Radionuclide by Cell, Central Plateau.

Table C-18. Estimated Radiation Dose, Total by Cell, Central Plateau.

Table C-19. Hazard Index, Chemical by Cell, All Other Areas.

Table C-20. Hazard Index, Summed by Cell, All Other Areas.

Table C-21. Estimated Radiation Dose, Radionuclide by Cell, All Other Areas.

Table C-22. Estimated Radiation Dose, Total by Cell, All Other Areas.

Table C-23. Estimated Radiation Doses to Aquatic Organisms.

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1. At the time of this writing, the *Hanford Site Baseline Risk Assessment Methodology (HSBRAM)* was used to conduct the ecological risk assessment. Since that time, it has been updated and revised in the *Hanford Site Risk Assessment Methodology (HSRAM)* (DOE-RL 1995).

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APPENDIX B

INEL-94/0190
Revision 1

Guidance Manual for Conducting Screening Level Ecological Risk Assessments at the INEL

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BW = Receptor-specific body weight (kg)

SUF = Site use factor (unitless).

When a surface water exposure pathway exists, exposure estimates due to ingestion of surface water for each receptor must be included in the exposure assessment. The equation for estimating exposure for drinking water is

$$EE_{water} = C \times EPC \tag{6}$$

where

EE_{water} = Estimated exposure from drinking water ingestion (mg/day)

C = Water consumption (liters/day) as function of body weight in kilograms (see Table 3-18)

EPC = Exposure point concentration of surface water (mg/liter).

Exposure Formulas for Radioactive Contaminants

The IAEA report (IAEA 1992) states that there is little doubt that radionuclides in the environment can produce doses to certain organisms similar to or even substantially higher than doses to people living in and deriving sustenance from the same environment. Therefore, the risk of effects for natural biota (discounting variations in radiosensitivity, life span, etc.) would appear to be as high, or higher than that for humans. However, there is a basic difference in the manner that risk assessment for humans is performed. For humans, the risk assessment is directed at the individual, while other species are viewed and valued more as populations than as individuals.

The assumption will be made that radionuclides emitting alpha and beta particles will not present a external dose risk since the basic rule of thumb (Sclein 1992) is that it requires an alpha particle of at least 7.5 MeV or a beta particle of at least 70 keV to penetrate a protective layer of skin (0.07 mm thick). Therefore, only the internal dose for these emitters will be assessed. Gamma emitters can produce a dose rate to tissues from both external and internal exposure and will be included in both assessments. Radionuclides can potentially cause both internal and external exposure since they emit particles with different energies and characteristics.

External Dose

The rule of thumb calculation for external dose (Shleien 1992) is recommended for SLERA to calculate the external dose to tissue. The following equation is used to calculate the dose rate to tissue in an infinite medium uniformly contaminated by a gamma emitter:

$$DR_{external} = \frac{2.12EC}{\rho} \tag{7}$$

where

$DR_{external}$	=	External radiation dose estimate (rad/hour)
E	=	Average decay energy (MeV)
C	=	The concentration of the radionuclide ($\mu Ci/cm^3$)
ρ	=	The density of the medium (g/cm^3).

The average energy per disintegration is assumed to be 100% of the gamma emitters. This is assumed to be a conservative estimate of the dose to burrowing functional groups. Exposure to nonburrowing functional groups is assumed to be 50% (a hemisphere).

Internal Dose

Internal radiation exposure dose estimates should be calculated using the approach presented by IAEA (1992). The technical report (IAEA 1992) provides valuable information on the estimated doses to both plants and animals under current radiation protection standards. The dose estimates in this technical report (IAEA 1992) have been calculated for three different scenarios: (1) controlled releases of radionuclides to the atmosphere, (2) controlled releases of radionuclides to a freshwater aquatic system, and (3) uncontrolled releases of radionuclides from a shallow land nuclear waste repository. The last scenario is considered applicable for use with contaminated media at the INEL and it is recommended the SLERA analysis use the equations presented in this paper to calculate radiation dose estimates. It is also important to review the information found in the technical report (IAEA 1992) during the SLERA analysis phase, since some simplifying (and conservative) assumptions concerning decay energy and absorption are presented.

For terrestrial receptors (either plant or animal), the dose from radionuclide contaminants is estimated by assuming the internal radiation dose estimate (calculated from the steady-state whole body concentration) is equivalent to the steady-state concentration of radionuclides in reproductive organs. The equation of interest is

$$DR_{internal} = \frac{TC \times E \times FA \times 3200\ dis/day\text{-}pCi}{6.24x10^9\ MeV/g\text{-}Gy} \tag{8}$$

where

$DR_{internal}$	=	Internal radiation dose estimate (Gy/day)
TC	=	Tissue radionuclides concentration (pCi/g)
E	=	Average decay energy (MeV/dis)

FA = Fraction of decay energy absorbed (unitless).

Since tissue levels (TCs) for radionuclides are derived by multiplying the concentration of radionuclide in soil by a radionuclide-specific concentration factor (CF) for all terrestrial animals or terrestrial plants, Equation (12) can be rewritten as

$$DR_{internal} = \frac{CS \times CF \times ADE \times FA \times 3200\ dis/day\text{-}pCi}{6.24x10^{9}\ MeV/g\text{-}Gy} \qquad (9)$$

MeV

where

$DR_{internal}$ = Internal radiation dose estimate (Gy/day)

CS = Concentration of contaminant in soil ingested (pCi/g)

CF = Concentration factor (unitless)

ADE = Average decay energy (MeV/dis)

FA = Fraction of decay energy absorbed (unitless).

Assumptions used in the calculation of the ADE values were (a) for ß or α radiations from a radionuclide the FA was set equal to 1 (100%) and (b) for γ the FA was set equal to 0.3 (30%). Only emissions with an intensity of 1% or greater were considered, and Auger and conversion electrons were not considered. The ADE values were calculated using the following equation (Kocher 1981):

$$ADE = \sum_{i}^{1} Y_i\, E_i \qquad (10)$$

where

ADE = Average disintegration energy (MeV/dis)

Y_i = Yield or intensity

E_i = Energy of radiation, for ß = average energy.

Table 3-25 presents an example of the calculation of exposure dosage to selected functional groups.

3.3.1.4 Development of Ecologically-Based Screening Levels. Development of ecologically-based screening levels (EBSLs) for contaminated media at each WAG allows a rational, consistent approach for (1) screening of sites that may require further investigation or remedial action, and (2) prioritization of sites based on comparison of concentrations of contaminants with EBSLs. It also allows inclusion of additional data as sites are subsequently

Table 3-25. Example table showing the estimated exposure dose to functional groups for radionuclide (Sr-90).

Functional group	Level in food or soil[a] (pCi/kg)	CF[b]	SUF[c]	ED[c]	TC[d] (pCi/kg)	Internal dose rate[e] (Gy/day)
Vegetation	534,000	1.11	1	1	592,740	5.94E-05
Mammalian/omnivores (M122A)	592,740	0.006	1	1	3,556	3.57E-07
Avian carnivores (AV310)	3,556	0.003	1	1	10.7	1.07E-09
Mammalian herbivores (M122)	592,740	0.006	0.5	1	1,778	1.78E-07

a. For those functional groups representing vegetation the value is the level in soil; for the functional groups representing herbivores and omnivores the value is the level in plants; for functional groups representing predators the value is the level in small mammals.

b. CF = concentration factor. For functional groups representing vegetation the value is the plant uptake factor; for the other functional groups the value is the food chain transfer factor.

c. SUF = site use factor, ED = exposure duration. See Table 3-19.

d. Tissue concentration = TC. The concentration in soil or food times the concentration factor times the site use factor times the exposure duration.

e. See text for explanation of calculation of internal dose rate from prey concentration.

sampled. It is a basic modification of the equations previously presented and is the recommended method for those WAGs that have not yet sampled all the sites identified in the FFA/CO.

The approach includes the following elements: compilation of ecological-based toxicity criteria to generate appropriate toxicity reference values (TRVs); rearrangement of exposure equations to calculate EBSLs in media from target intakes and default exposure assumptions; development of general (site-wide) EBSLs for each functional group by contaminant using best available estimates for species-specific exposure parameters and TRVs these general EBSLs are modified for application at a WAG by dividing by the SUF); calculation of screening level quotients (SLQs; the ratio of [contaminant] to EBSL) for each contaminant; and evaluation of risks from multiple contaminants. Since conservative assumptions are inherent in the process of EBSL development, if no exceedance occurs and the existence of multiple contaminants does not appear to be a contributor to potential risk, then the contaminant can be eliminated from further consideration as potential sources of risk to the receptors. Exceedance of the EBSLs indicate that further investigation of potential risks to ecological receptors is warranted, depending on the magnitude of the exceedance, the uncertainty involved in TRV and EBSL development, and other considerations. If EBSLs are used in the SLERA for a WAG, the results of this effort need to be presented as shown in Table 3-26. Summarization of the EBSL provides a useful tool that can be incorporated into further SLERA work as applicable.

be presented as shown in Table 3-26. Summarization of the EBSL provides a useful tool that can be incorporated into further SLERA work as applicable.

EBSL Formulas for Nonradiological Contaminants

The following example is presented to show the development of EBSLs for screening against nonradiological soil contamination concentrations. The toxicity quotient (TQ), which represents a quantitative method for evaluating potential adverse impacts to exposed populations, is defined as

$$TQ = \frac{EE}{TRV} \tag{11}$$

where

TQ	=	Toxicity quotient (unitless)
EE	=	Estimated exposure (mg/kg body weight-day)
TRV	=	Contaminant-specific toxicity reference value (mg/kg-day).

Thus, solving for the concentration of the nonradionuclide contaminant in the soil (CS) and assuming that when TQ equals 1 that EE_{soil} = TRV, the equation becomes

$$EBSL_{soil} = \frac{TRV \times BW}{[(PP \times BAF) + (PV \times PUF) + PS] \times IR \times ED} \tag{12}$$

where

$EBSL_{soil}$	=	INEL-specific ecologically-based screening level for non-radiological contaminants in soil (mg/kg)
TRV	=	Toxicity reference value (mg/kg-day)
PP	=	Percentage of diet represented by prey ingested (unitless)
PV	=	Percentage of diet represented by vegetation ingested (unitless)
PS	=	Percentage of diet represented by soil ingested (unitless)
IR	=	Ingestion rate (kg/day)
ED	=	Exposure duration (fraction of year spent in the assessment area) (unitless)
BW	=	Receptor-specific body weight (kg).

Table 3-26. Example table showing the calculated EBSLs for nonradionuclides (mg/kg) for a WAG with assessment area of 1000 ha.

Functional groups	Antimony	Chromium VI	Chromium III	Mercury
Avian herbivores (AV122)	—	2.37E+03	—	1.87E-01
Avian insectivores (AV210)	—	4.45E+03	—	2.01E-01
Avian insectivores (AV222)	—	2.30E+02	—	1.60E-01
Avian carnivores (AV310)	—	5.41E+01	—	1.07E+00
Avian carnivores (AV322)	—	5.05E+01	—	9.98E-01
Avian omnivores (AV422)	—	5.10E+00	—	2.13E-02
Mammalian herbivores (M122)	2.50E+01	2.44E+03	1.79E+05	7.40E+00
Mammalian/herbivores (M122A)	2.04E+01	3.50E+03	2.60E+05	3.84E-01
Mammalian insectivores (M210)	2.14E+01	3.45E+03	6.05E+03	4.00E-01
Mammalian carnivores (M322)	1.20E+00	6.05E+03	6.85E+04	4.80E+01
Mammalian/omnivores (M422)	2.37E+00	2.90E+03	2.16E+05	1.84E-01
Mammalian omnivores (M422A)	1.62E+01	5.77E+02	9.23E+05	6.47E+02
Reptiles/insectivores (R222)	—	—	—	—
Reptiles/carnivores (R322)	—	—	—	—
Peregrine Falcon and Northern Goshawk (AV312)[a]	—	2.71E+01	—	5.35E-01
Bald Eagle, Ferruginous Hawk, and Loggerhead Shrike (AV322)[a]	—	2.53E+01	—	1.99E-01
Pygmy Rabbit (M122A)[a]	1.02E+01	1.75E+03	1.30E+05	1.92E-01
Townsend's big-eared bat (M210)[a]	1.07E +01	1.72E+03	3.03E+03	2.00E-01

a. A factor of 2 was included in the calculation of the WAG-specific EBSL for these T/E species.

Note: A dash (—) indicates that the ecosystem parameters were not available to calculate the EBSL for this functional group/contaminant combination.

EBSL Formulas for Radiological Contaminants

The same concept used to develop the EBSLs formulas for nonradionclides can be used for radionuclides. In this case, TQs are defined as:

$$TQ = \frac{DR}{TRV} \tag{13}$$

where

TQ = Toxicity quotient (unitless)

DR = Estimated dose rate (rad/hour)

TRV = Toxicity reference value (rad/hour).

External EBSL

First solving for radionuclide concentration (C) and assuming that when TQ equals 1, then DR = TRV, Equation (7) becomes

$$(C \times 10^6 \rho Ci/\mu Ci) \div (1.68\ g/cm^3) = \frac{(TRV \times \rho \times 10^6 \rho Ci/\mu Ci)}{(2.12 \times E \times 1.68\ g/cm^3)} \tag{14}$$

where

C = The concentration of the radionuclide ($\mu Ci/cm^3$)

TRV = Toxicity reference value (rad/hour)

ρ = The density of the medium (1.68 g/cm^3)

E = Average decay energy (MeV).

The density of the medium is eliminated, on the right side of the equation and the concentration of the radionuclide is converted from uCi to pCi by multiplying by a factor of 10^6. Next, the equation is simplified, and the left side of the equation becomes the EBSL. In other words the concentration of radionuclide in the soil that results when the TQ = 1. The EBSL equation for external dose becomes

$$EBSL_{external} = (TRV_{external} \times 10^6 pCi/uCi) \div 2.12 \times E \tag{15}$$

where

$EBSL_{external}$ = Ecologically-based screening level value for external dose (pCi/g)

TRV = Toxicity reference value (rad/hour)

E = The average gamma energy per disintegration (MeV).

Internal EBSL

Solving for the concentration of radionuclide in soil (C) and setting DR = TRV in Equation (9), the concentration for radiological contaminants in soil is redefined as an EBSL for internal dose

$$EBSL_{internal} = \frac{TRV \times 6.24x10^9\ MeV/g\text{-}Gy}{CF \times ED \times ADE \times FA \times 3200\ dis/day\text{-}pCi} \quad (16)$$

where

$EBSL_{internal}$	=	Ecological based screening level for radionuclides in soil (pCi/g)
TRV	=	Toxicity reference value (Gy/day)
CF	=	Concentration factor (unitless)
ED	=	Exposure duration (unitless)
ADE	=	Average decay energy (MeV/dis)
FA	=	Fraction of decay energy absorbed (unitless).

3.3.2 Effects Assessment

The purpose of the effects (or, stressor-response) assessment is to characterize the toxicity of stressors to selected measurement endpoints. In this section effects of the contaminants on those functional groups identified as potential receptors will be quantified as TRVs. This process relies on professional judgment, especially when few data are available or when choices among several sources of data are required. If available data are inadequate, this will be identified as a data gap and will be addressed in the screening evaluation.

There are numerous sources of these data, including:

- Primary literature sources (veterinary science literature, journal articles, and scientific publications)
- Registry of Toxic Effects of Chemical Substances
- Hazardous Substances Database
- Integrated Risk Information System
- Agency for Toxic Substances and Disease Registry
- Phytotox Database
- Aquatic Information Retrieval
- Chemical Evaluation Search and Retrieval System