

SCREENING RISKS TO TERRESTRIAL VERTEBRATES FROM RADIONUCLIDE CONTAMINATION IN SOIL AND WATER

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ABSTRACT

In 1995, we published a screening-level risk assessment methodology for use on the Idaho National Engineering and Environmental Laboratory (INEEL).¹⁻⁴ Few other methodologies for screening-level ecological risk assessments of radionuclides are available. Because radionuclides are increasingly common as contaminants of concern in ecological risk assessments, there is a need to refine existing methodologies to address them. Compliance with regulatory limits will also require screening methodologies. The U.S. Department of Energy's Biota Dose Assessment Committee is developing simple screening methods for compliance with the proposed biota dose limits in 10 CFR Part 834. In response to those needs, we propose a methodology for screening risks to terrestrial vertebrates from radionuclide-contaminated soil and drinking water. The methodology is founded on basic principles of radiation dosimetry with conservative, but realistic, assumptions where appropriate. It makes maximum use of available data, deriving basic parameters primarily from the literature. In most cases, the only field measurements required are radionuclide concentrations in soil and water. The methodology incorporates simple food web modeling and screens multiple radionuclides simultaneously. The proposed methodology is a modification of methods currently used at the INEEL and is intentionally similar to INEEL methodologies for non-radionuclide contaminants. This results in increased acceptability and ease of use. The methodology is easily adaptable for assessing risks to terrestrial vertebrates on many sites different from the INEEL.

INTRODUCTION

In 1995, VanHorn, Hampton, and Morris published a methodology for conducting screening-level ecological risk assessments on the Idaho National Engineering and Environmental Laboratory (INEEL).¹⁻⁴ This methodology included methods for assessing ecological risks from radionuclide contamination and has been used to screen most Waste Area Groups (WAGs) on the INEEL. The methods developed at that time were useful and acceptable to regulators and stakeholders. However, because radionuclides are increasingly common as contaminants of concern on the INEEL and elsewhere, there is a need to refine the existing methodologies so they can be, on the one hand, more site-specific and, on the other hand, more easily generalized to other sites. In addition to ecological risk assessment, compliance with environmental regulations requires screening-level methodologies. For example, the U.S. Department of Energy's (DOE) Biota Dose Assessment Committee is charged with developing methods by which facilities can show compliance with the proposed biota dose limits in 10 CFR Part 834. This and other compliance related efforts will require screening-level methodologies.

In this paper, we show the mathematical basis and discuss the application of a proposed methodology for screening risks to terrestrial vertebrates from radionuclide - contaminated soil and drinking water. The methodology is founded on basic principles of radiation dosimetry with conservative, but realistic, assumptions where appropriate. It makes maximum use of available data, deriving basic parameters primarily from the literature. In most cases, the only field

measurements required are radionuclide concentrations in soil and water. The methodology incorporates simple food web modeling and screens multiple radionuclides simultaneously. The proposed methodology is a modification of methods currently used at the INEEL and is intentionally similar to INEEL methodologies for non-radionuclide contaminants. This results in increased acceptability and ease of use. The methodology is easily adaptable for assessing risks to terrestrial vertebrates on many sites different from the INEEL.

MATHEMATICAL DEVELOPMENT

Internal Dosimetry

Under our working assumptions (Table 1) and parameter definitions (Table 2), a given radionuclide is taken up and eliminated from an organism by a first order model (Fig. 1). The differential equation describing this model is:

Eq. 1

$$\frac{dC_{Ti}}{dt} = \frac{I_i}{M} - (\lambda_{pi} + \lambda_{bi})C_{Ti}$$

which, under an assumption of equilibrium ($t \rightarrow \infty$), solves to:

Eq. 2

$$C_{Ti} = \frac{I_i}{M(\lambda_{pi} + \lambda_{bi})}$$

For any terrestrial vertebrate, the total intake rate of radionuclide i (I_i) is the sum of the intake from the animal tissue ingested (I_{pi}), the plant material ingested (I_{vi}), the water ingested (I_{wi}), and the soil ingested (I_{si}). Each of these depends on the concentration of the radionuclide in the respective medium, the fraction of that medium in the diet of the organism, and the rate at which the organism consumes food or water.

Eq. 3

$$I_i = I_{pi} + I_{vi} + I_{wi} + I_{si}$$

Table 1. Assumptions of a method for estimating dose rates to terrestrial vertebrates from radiation contaminated soil and water.

- | | |
|-----|--|
| 1) | The system is at equilibrium |
| 2) | The vertebrates exhibit a constant intake rate |
| 3) | All intake is absorbed and distributed immediately throughout the entire organism |
| 4) | Single target organism |
| 5) | Direct contact with contaminated soil |
| 6) | Subsurface animals are surrounded by an infinite "cloud" of contaminated soil |
| 7) | For external dose, the fluence rate for surface animals is half that of buried animals |
| 8) | α and β radiations do not contribute to external dose |
| 9) | For internal dose, all α and β energy is absorbed in the organism, 30% of γ energy is absorbed |
| 10) | No significant external radiation from water |
| 11) | Inhalation of radioactivity is negligible |

Table 2. Definitions of parameters for a method of estimating dose rates and risk to terrestrial vertebrates from radiation contaminated soil and water.

Parameter	Definition	Units
$\left(\frac{\mu_{en}}{\rho}\right)_j^i$	mass energy absorption coefficient in animal tissue of radiation j from radionuclide i	$\text{cm}^2 \text{g}^{-1}$
Λ_{bi}	biological loss constant of radionuclide i	d^{-1}
λ_{pi}	physical decay constant of radionuclide i	d^{-1}
μ_{ij}	Attenuation coefficient in soil for radiation j of radionuclide i	cm^{-1}
ρ_s	Soil density	g cm^{-3}
ϕ_{ij}	Particle fluence rate of radiation j from soil contaminated with radionuclide i	Bq cm^{-2}
C_{Si}^E	Ecologically-based screening level for radionuclide i in soil	Bq g^{-1}
C_{Wi}^E	Ecologically-based screening level for radionuclide i in water	Bq l^{-1}
C_{Pi}	Concentration of radionuclide i in prey food items	Bq g^{-1}
C_{Si}	Concentration of radionuclide i in soil	Bq g^{-1}
C_{Ti}	Whole-body tissue concentration of radionuclide i	Bq g^{-1}
C_{Vi}	Concentration of radionuclide i in vegetation	Bq g^{-1}
C_{Wi}	Concentration of radionuclide i in water	Bq l^{-1}
D_{Ei}	Dose rate due to external radiation from radionuclide i	Gy d^{-1}
D_{Ii}	Dose rate due to internal contamination with radionuclide i	Gy d^{-1}
D_T	Dose rate due to external and internal radiation from all radionuclides	Gy d^{-1}
D_{Ti}	Dose rate due to external and internal radiation from radionuclide i	Gy d^{-1}
E_i	Average absorbed energy equivalent of radionuclide i	MeV
E_{ij}	Energy of radiation j from radionuclide i = E_{avg} for β spectra, actual energy for others	MeV
F_{Aij}	Fraction of radiation j from radionuclide i absorbed in organism = 1 for α or β radiation, 0.3 for γ radiation ⁵	unitless
F_B	Burrowing factor = 1 if animal spends most of its time underground 2 if animal spends most of its time on the ground surface	unitless
F_{Bi}	Bioaccumulation factor for radionuclide i	unitless
F_d	Fraction of diet from contaminated area	unitless
F_P	Fraction of diet as prey	unitless
F_S	Fraction of diet as soil	unitless
F_{Ui}	Plant uptake factor for radionuclide i	unitless
F_V	Fraction of diet as vegetation	unitless
F_w	Fraction of water from contaminated water source	unitless
I_i	Intake rate of radionuclide i	Bq d^{-1}
I_{Pi}	Intake rate of radionuclide i from animal tissue ingested	Bq d^{-1}
I_{Si}	Intake rate of radionuclide i from soil ingested	Bq d^{-1}
I_{Vi}	Intake rate of radionuclide i from plant material ingested	Bq d^{-1}
I_{Wi}	Intake rate of radionuclide i from water ingested	Bq d^{-1}
J_1	Summary variable (see derivation in text)	$\text{Gy g Bq}^{-1} \text{d}^{-1}$

J_2	Summary variable (see derivation in text)	$\text{Gy l Bq}^{-1} \text{d}^{-1}$
k	unit conversion factor	$\text{Gy g MeV}^{-1} \text{Bq}^{-1} \text{d}^{-1}$
M	Body mass of organism	g
Q_{ij}	Quality factor for radiation j from radionuclide i = 1 for β or α radiation, 20 for α radiation	unitless
R_{PSV}	Food ingestion rate	g d^{-1}
R_{W}	Water ingestion rate	l d^{-1}
S_{Vi}	Volumetric soil concentration of radionuclide i	Bq cm^{-3}
TQ	Toxicity quotient	unitless
TRV	Toxicity reference value	Gy d^{-1}
Y_{ij}	Intensity (yield) for radiation j from radionuclide i	unitless

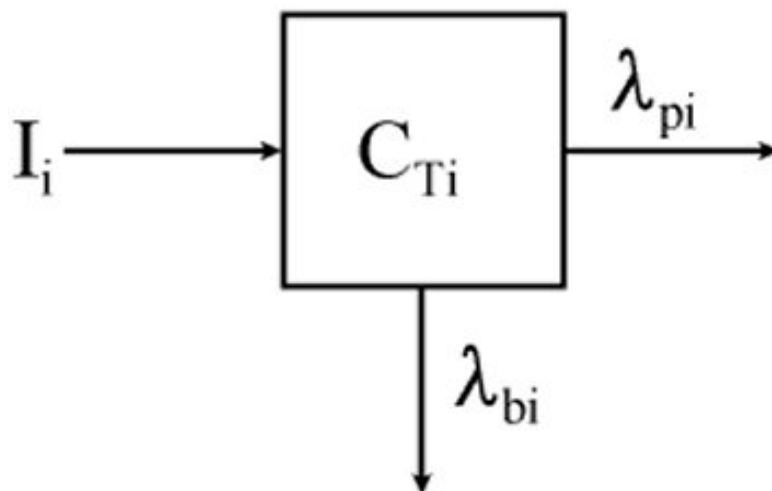


Fig. 1. Diagram of a conceptual model for ingestion and elimination of radionuclide i from an organism. The model assumes instantaneous, uniform mixing. Parameter definitions are in Table 1.

$$I_{\text{Pi}} = C_{\text{Pi}} F_{\text{P}} R_{\text{PSV}} \quad \text{Eq. 4}$$

$$I_{\text{Vi}} = C_{\text{Vi}} F_{\text{V}} R_{\text{PSV}} \quad \text{Eq. 5}$$

$$I_{\text{Si}} = C_{\text{Si}} F_{\text{S}} R_{\text{PSV}} \quad \text{Eq. 6}$$

$$I_{wi} = C_{wi} R_w$$

Eq. 7

All food and water ingested does not necessarily come from a contaminated source. Therefore, we modify the intake by a coefficient representing the fraction of food and water that is contaminated. Combining Eqs. 3 - 7 and including this coefficient results in:

$$I_i = (C_{Pi} F_P + C_{Si} F_S + C_{Vi} F_V) R_{PSV} F_d + C_{Wi} R_W F_W.$$

Eq. 8

The most commonly available contamination data on the INEEL are soil and water concentrations. Thus we estimate concentrations in vegetation and prey animals using bioaccumulation factors and plant uptake factors.

$$C_{Pi} = F_{Bi} C_{Si}$$

Eq. 9

$$C_{Vi} = F_{Ui} C_{Si}$$

Eq. 10

$$I_i = (F_{Bi} F_P + F_{Ui} F_V + F_S) C_{Si} R_{PSV} F_d + C_{Wi} R_W F_W$$

Eq. 11

Combining Eq. 2 and Eq. 11 :

$$C_{Ti} = \frac{(F_{Bi} F_P + F_{Ui} F_V + F_S) C_{Si} R_{PSV} F_d + C_{Wi} R_W F_W}{M(\lambda_{pi} + \lambda_{bi})}.$$

Eq. 12

A dose rate to the terrestrial vertebrate from internal contamination can be estimated using an "average energy equivalent deposited." This value is the sum over all radiations j from radionuclide i of the energy of the radiation (the average β energy or the actual energy of α or γ radiations), the fraction of decays yielding that radiation (its intensity or yield), the quality factor of the radiation (accounting for the greater ionization in a shorter path length from α radiation), and a factor accounting for the different penetrating abilities of the different radiations.)

Eq. 13

$$D_{\text{Ei}} = kC_{\text{Ti}}\bar{E}_i = kC_{\text{Ti}} \sum_j E_{ij} Y_{ij} Q_{ij} F_{Aij}$$

Combining , Eq. 12 and Eq. 13, and using a unit conversion factor, results in an estimate of radiation dose rate to a terrestrial vertebrate from eating contaminated food and drinking contaminated water.

Eq. 14

$$D_{\text{Ei}} = k \left[\frac{(F_{\text{Bi}}F_{\text{p}} + F_{\text{Ui}}F_{\text{v}} + F_{\text{S}})C_{\text{Si}}R_{\text{psv}}F_{\text{d}} + C_{\text{wi}}R_{\text{w}}F_{\text{w}}}{M(\lambda_{\text{pi}} + \lambda_{\text{bi}})} \right] \sum_j E_{ij} Y_{ij} Q_{ij} F_{Aij}$$

External Dosimetry

The external dose rate to a terrestrial vertebrate from contaminated soil can be determined by summing over all radiations, j, from radionuclide i, the product of the particle fluence rate of the radiation, its energy, and its mass energy absorption coefficient in animal tissue. This process is only valid for γ radiation. However, we assume that external radiation from α and β radiation are negligible because they will not usually penetrate the skin of the animal.

Eq. 15

$$D_{\text{Ei}} = k \sum_j \phi_{ij} E_{ij} \left(\frac{\mu_{\text{en}}}{\rho} \right)_{ij}$$

The particle fluence rate is composed of the volumetric source strength of radionuclide i, the intensity of radiation j from radionuclide i, the attenuation coefficient in soil of radiation j from radionuclide i, and a burrowing factor, accounting for whether the organism lives primarily on or under the soil surface. The volumetric source strength is the product of the mass concentration of radionuclide i in soil and the soil bulk density.

Eq. 16

$$\phi_{ij} = \frac{S_{\text{wi}} Y_{ij}}{F_{\text{B}} \mu_{ij}}$$

Eq. 17

$$S_{\text{wi}} = C_{\text{Si}} \rho_{\text{S}}$$

Eq. 18

$$D_{\text{Ei}} = \frac{kC_{\text{Si}}\rho_{\text{S}}}{F_{\text{B}}} \sum_j \frac{E_{ij} Y_{ij}}{\mu_{ij}} \left(\frac{\mu_{\text{en}}}{\rho} \right)_{ij}$$

Total Dose Rate

The estimated total dose rate from radionuclide i is the sum of the external and internal dose rates.

$$D_{Ti} = k \left[\frac{(F_{Bi}F_P + F_{Ui}F_V + F_S)C_{Si}R_{PSV}F_d + C_{Wi}R_W F_W}{M(\lambda_{pi} + \lambda_{bi})} \right] \sum_j [E_{ij} Y_{ij} Q_{ij} F_{Aj}]$$

$$+ \frac{kC_{Si}\rho_s}{F_B} \sum_j \left[\frac{E_{ij} Y_{ij}}{\mu_{ij}} \left(\frac{\mu_{en}}{\rho} \right)_{ij} \right]$$

Eq. 19

For simplicity, we define two summary variables:

$$J_{ii} = \frac{k(F_{Bi}F_P + F_{Ui}F_V + F_S)R_{PSV}F_d}{M(\lambda_{pi} + \lambda_{bi})} \sum_j [E_{ij} Y_{ij} Q_{ij} F_{Aj}] + \frac{k\rho_s}{F_B} \sum_j \left[\frac{E_{ij} Y_{ij}}{\mu_{ij}} \left(\frac{\mu_{en}}{\rho} \right)_{ij} \right]$$

Eq. 20

Then, combining , and , and rearranging:

$$J_{2i} = \frac{kR_W F_W}{M(\lambda_{pi} + \lambda_{bi})} \sum_j [E_{ij} Y_{ij} Q_{ij} F_{Aj}]$$

Eq. 21

$$D_{Ti} = J_{ii}C_{Si} + J_{2i}C_{Wi}$$

Eq. 22

The total dose rate from all radionuclides present is estimated by summing over all radionuclides.

$$D_T = \sum_i [J_{ii}C_{Si} + J_{2i}C_{Wi}]$$

Eq. 23

Toxicity Quotient

The total dose rate from all radionuclides to the target organism can be compared to accepted dose rate limitations (Toxicity Reference Values; TRV) to determine whether the

population of organisms is at risk from the radiation. If the resulting Toxicity Quotient (TQ) is greater than 1, the population may be at risk.

Eq. 24

$$TQ = \frac{D_T}{TRV}$$

The TRV commonly used at the INEEL is 10 mGy d^{-1} .⁵

Ecologically-based Screening Levels

Risk assessors and WAG managers at the INEEL find it most convenient to screen their contaminated sites based on soil concentrations derived from acceptable limits. These concentrations, known as Ecologically-based Screening Levels (EBSL), can be calculated by determining the soil and/or water concentrations (C_{Si}^E and C_{Wi}^E) that would result in $TQ = 1$. At $TQ = 1$, $D_T = TRV$.

If a single radionuclide, i , is present in the contaminated soil and/or water, then:

$$TRV = J_{1i}C_{Si}^E + J_{2i}C_{Wi}^E$$

Eq. 25

$$C_{Si}^E = \frac{TRV - J_{2i}C_{Wi}^E}{J_{1i}}$$

Eq. 26

$$C_{Wi}^E = \frac{TRV - J_{1i}C_{Si}^E}{J_{2i}}$$

Eq. 27

For a given radionuclide and target organism, all parameters except C_{Si} and C_{Wi} are constant. Therefore, although there are an infinite number of C_{Si}^E and C_{Wi}^E that would solve Eq. 25, they define a line with a negative slope (Fig. 2). Any combination of soil and water concentration below and left of the line will result in a dose rate to members of the population less than the TRV. If either variable equals zero, i.e., there is no soil or water concentration, any value of the other variable less than the EBSL will result in a dose rate to members of the population less than the TRV.

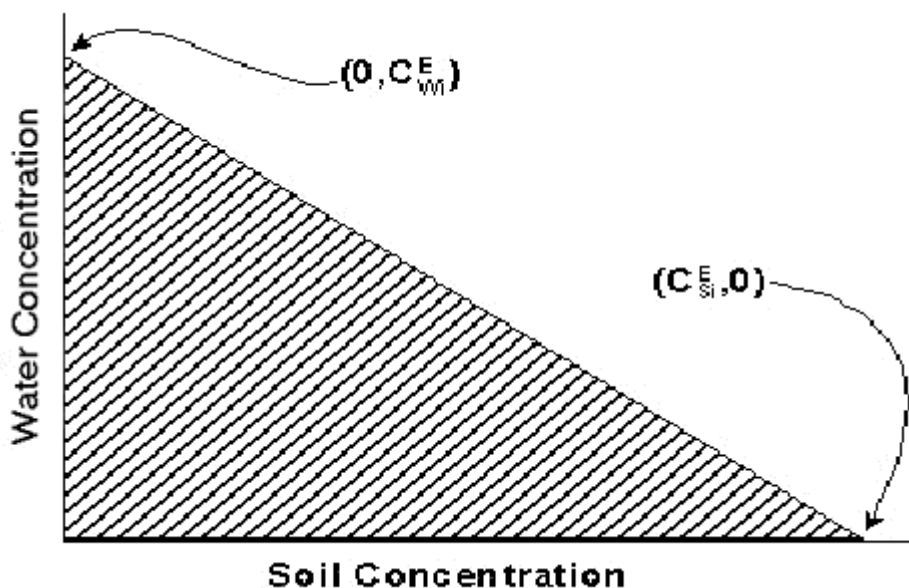


Fig. 2. Relationship between soil concentration and water concentration for a single radionuclide where the estimated dose rate to a terrestrial vertebrate is equal to the Toxicity Reference Value. Any combination of soil and water concentrations in the shaded area will result in a Toxicity Quotient less than one.

For multiple radionuclides, the situation is somewhat more complex. At $TQ = 1$:

$$TRV = \vec{C}_S^E \cdot \vec{J}_1 + \vec{C}_W^E \cdot \vec{J}_2$$

Eq. 28

where $\vec{C}_S^E$ and $\vec{C}_W^E$ are the vectors of soil and water concentrations of i radionuclides that solve Eq. 28, $\vec{J}_1$ is a vector of $\vec{J}_{1i}$ for i radionuclides, and $\vec{J}_2$ is a vector of $\vec{J}_{2i}$ for i radionuclides.

Similar to the single radionuclide cases, $\vec{C}_S^E \cdot \vec{J}_1$ and $\vec{C}_W^E \cdot \vec{J}_2$ define a line with a negative slope (Fig. 3). Any combination of these parameters below and left of the line will result in a dose rate to members of the population less than the TRV. If either variable equals zero, i.e., there is no soil or water concentration, any value of the other variable less than the EBSL will result in a dose rate to members of the population less than the TRV. However, this observation may be of little practical benefit. In this case, there are an infinite number of vectors, each of length i , that would solve Eq. 28. The magnitude of the calculation would not likely justify the approach. The most straightforward approach would be to measure the soil and water concentrations and calculate a TQ on a site-by-site basis using Eq. 24.

APPLICATION

This methodology is intended to fit into the ecological risk assessment process as defined by the U.S. Environmental Protection Agency⁶ and applied at the site. Endpoints and Contaminants of Potential Concern (COPCs) are selected by any appropriate and agreed upon

method. If COPC selection normally includes screening against background and/or regulatory criteria (e.g. water quality criteria), that screening should still take place. If the remaining COPCs include one or more radionuclides and the endpoints include terrestrial vertebrates, this methodology may be applied. The nature of its application depends on the number of radionuclides. If a single radionuclide is present, a useful EBSL can be

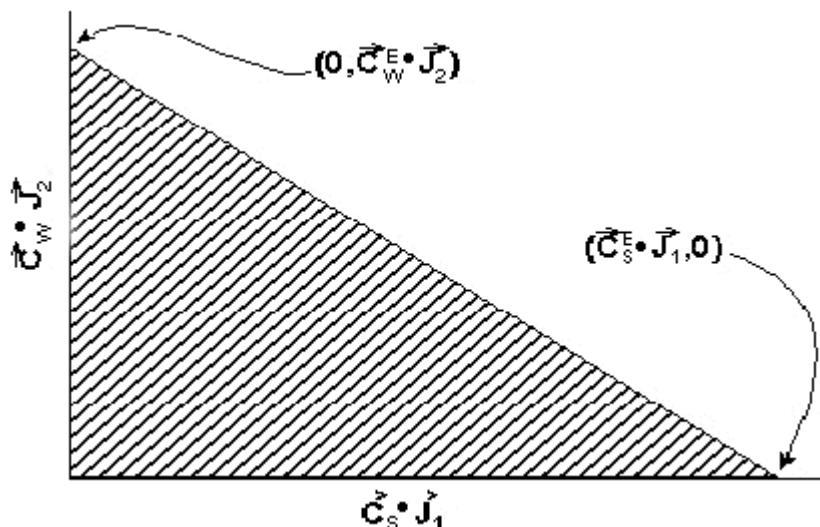


Fig. 3. Relationship between soil concentration and water concentration for i radionuclides where the estimated dose rate to a terrestrial vertebrate is equal to the Toxicity Reference Value. Any combination of soil and water concentrations in the shaded area will result in a Toxicity Quotient less than one.

calculated by determining the line defining the “safe area” (Fig. 2) for each endpoint species. If these lines are plotted on the same axes, the EBSL or “safe area” is defined by the line or line segments closest to the origin (Fig. 4). This EBSL can then be used to screen any site for which one is willing to assume the same characteristics.

As noted above, if multiple radionuclides are present, the magnitude of the calculation will probably make the determination of an EBSL too cumbersome to be useful. In this case, we recommend that screening be carried out on a site-by-site basis using Eq. 24.

CASE STUDY

To demonstrate its utility and compare its results to the current method, this methodology was applied to screen the radionuclides at WAG 2, a site on the INEEL for which a screening-level ecological risk assessment was already completed using current methodologies.⁷

At the INEEL, endpoints are defined using a functional grouping method.¹ This case study determined risks from all radioactive COPCs at WAG 2 for functional group M422, small omnivorous mammals. All assumptions about the biological characteristics of the functional group and the physical characteristics of the site are consistent with those assumed in the existing assessment (Table 3). We used the reported surface soil concentrations in our analysis.

In the original assessment, radiations with intensity (Y_{ij}) less than 1% were excluded from the analysis. We made the same assumption. However, we did not exclude auger and conversion electrons or x-rays, as did the original assessment. Dose contributions from decay

series were not considered for this case study although, for a true analysis we would recommend their consideration.

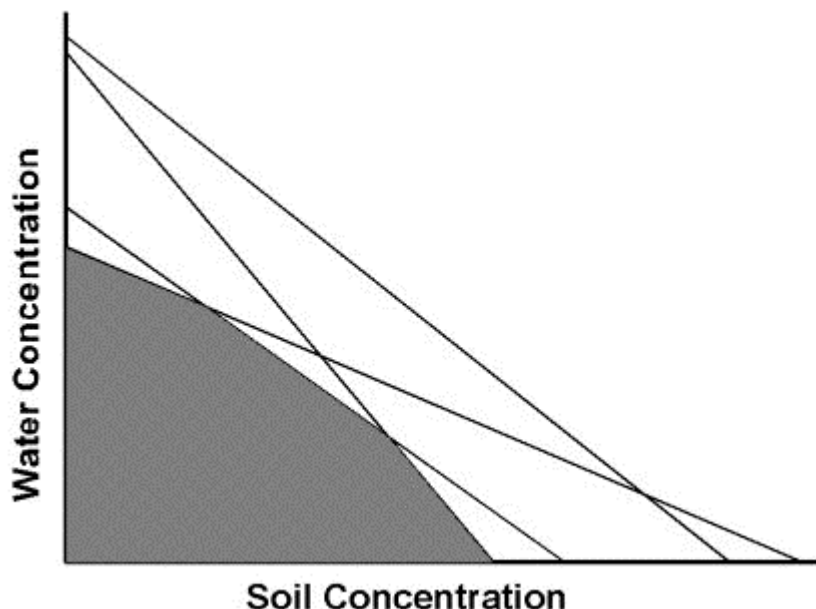


Fig. 4. Relationship between soil concentration and water concentration for a single radionuclide and multiple terrestrial vertebrates where the dose rate is equal to the Toxicity Reference Value. Any combination of soil and water concentrations in the shaded area will result in a Toxicity Quotient < 1 for the most sensitive species and will thus be protective of all species.

Using this methodology, we concluded that functional group M422 was potentially at risk from radionuclides in soil at WAG 2. As can be seen from the doses due to individual radionuclides (the last column in Table 3), the largest contributors to this risk are ^{90}Sr and, to a lesser extent ^{238}Pu , ^{239}Pu , and ^{241}Am . This is similar to results obtained in the original assessment, where the Screening-level Quotient (SLQ) for internal radiation from ^{90}Sr was 2.5, an indication of potential risk from this radionuclide. However, the transuranic elements were not previously identified as important contributors to risk.

We must emphasize that the assumptions used in this analysis, based on the assumptions from the original assessment, are quite conservative. This indication of risk should only be considered an indication that further work should be considered.

SUMMARY AND CONCLUSION

The methodology for screening risks to terrestrial vertebrates presented here will, if the input data are adequate, provide a realistic, moderately conservative, assessment of risks to terrestrial vertebrates. With minor modification, it can be extended to screen risks at sites other than the INEEL. It can also be extended to other functional groups (e.g., birds) with slightly more involved, but not difficult modification. This would involve changes to parameters that modify external and internal dose. For birds, because the external dose calculation assumes an infinite sphere of contaminated material, it will likely require modification to an infinite plane source. Assessment of risk to animals that spend a significant fraction of their time in contaminated water (e.g., waterfowl or aquatic mammals) will also require more extensive

modification of this method. Other model changes associated with less conservative assumptions might involve the inclusion of an assimilation fraction in Eq. 1, a relatively minor change, or the consideration of inhalation dose, a major change in the model structure.

Table 3. Data and conclusions from a case study using a proposed methodology for screening risks from radionuclides to terrestrial vertebrates. Because $TQ > 1$, we would conclude the functional group under examination (see text) is potentially at risk

.Parameter	Units	Value						
Organism Data								
M	g	22						
F _P		0.49						
F _S		0.028						
F _V		0.482						
F _B		1						
R _{PSV}	g d ⁻¹	2.98						
R _W	l d ⁻¹	0.0032						
F _d ^a		1						
F _w ^b		0						
Home Range Size	Ha	0.0093						
Fraction of year in area		1						
Site Data								
Size of Site	Ha	1009.6						
ρ _s	g cm ⁻³	1.6						
Radionuclide data								
	Parameter							
Radionuclide	C _{Si} (Bq g ⁻¹)	C _{Wi} ^b (Bq l ⁻¹)	L _{Pi} (d ⁻¹)	L _{Bi} (d ⁻¹)	F _{Bi}	F _{Ui}	J _{Ii} ^b (Gy g Bq ⁻¹ d ⁻¹)	D _{Ti} (Gy d ⁻¹)
⁶⁰ Co	2.4	0	0.00036	0.0693	1	11	1.50E-04	3.60E-04
⁹⁰ Sr	290	0	6.66E-05	0.00924	1.5	19	2.71E-03	7.87E-01
¹³⁴ Cs	63	0	9.20E-04	0.693	1	0.53	1.05E-05	6.63E-04
¹³⁷ Cs	36	0	6.29E-05	0.693	1	0.53	4.28E-06	1.54E-04
¹⁵² Eu	36	0	1.40E-04	0.0365	1	0.01	1.83E-05	6.58E-04
¹⁵⁴ Eu	5.6	0	2.21E-04	0.0365	1	0.01	2.38E-05	1.33E-04
¹⁵⁵ Eu	0.7	0	3.82E-04	0.0365	1	0.01	1.91E-06	1.33E-06
²²⁸ Th	0.063	0	9.92E-04	0.0365	1	0.039	2.87E-03	1.81E-04
²³⁰ Th	0.01	0	2.46E-08	0.0365	1	0.039	2.56E-03	2.56E-05
²³² Th	0.059	0	1.35E-13	0.0365	1	0.039	2.20E-03	1.30E-04
²³⁴ U	0.21	0	7.76E-09	0.0365	1	0.014	2.55E-03	5.36E-04
²³⁸ U	0.044	0	4.25E-13	0.0365	1	0.014	2.25E-03	9.91E-05
²³⁸ Pu	0.021	0	2.16E-05	3.47E-05	1	0.0044	1.89E+00	3.98E-02
²³⁹ Pu	0.011	0	7.78E-08	3.47E-05	1	0.0044	2.88E+00	3.16E-02
²⁴¹ Am	4.1	0	4.38E-06	3.65E-02	1	0.017	2.94E-03	1.20E-02
							D _T	0.87
							TQ	87.35

^a If home range size < size of site = 1; otherwise = home range size/size of site.

^b The site contained no contaminated surface water. For this reason, J_{2i} = 0 and is not shown.

This methodology is a screening level methodology and the results are conservative. If a particular site is shown by this methodology to present a potentially unacceptable risk, that result should only be an indication that further work is necessary. However, if the methodology shows that risks are acceptable, it is reasonable to conclude the site may be excluded from further analysis for the functional groups in question.

Unlike the methodology previously used on the INEEL, this methodology provides an assessment of the risks from food, water, and external sources using a consistent and combined approach. Thus, it is more in line with currently accepted methods in radioecology and, if appropriately used, should provide more consistent, defensible results.

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