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Using a probabilistic approach in an ecological risk assessment simulation tool: test case for depleted uranium (DU)

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Abstract

A probabilistic approach was applied in an ecological risk assessment (ERA) to characterize risk and address uncertainty employing Monte Carlo simulations for assessing parameter and risk probabilistic distributions. This simulation tool (ERA) includes a Window's based interface, an interactive and modifiable database management system (DBMS) that addresses a food web at trophic levels, and a comprehensive evaluation of exposure pathways. To illustrate this model, ecological risks from depleted uranium (DU) exposure at the US Army Yuma Proving Ground (YPG) and Aberdeen Proving Ground (APG) were assessed and characteri2zed. Probabilistic distributions showed that at YPG, a reduction in plant root weight is considered likely to occur (98% likelihood) from exposure to DU; for most terrestrial animals, likelihood for adverse reproduction effects ranges from 0.1% to 44%. However, for the lesser long-nosed bat, the effects are expected to occur (>99% likelihood) through the reduction in size and weight of offspring. Based on available DU data for the firing range at APG, DU uptake will not likely affect survival of aquatic plants and animals (<0.1% likelihood). Based on field and laboratory studies conducted at APG and YPG on pocket mice, kangaroo rat, white-throated woodrat, deer, and milfoil, body burden concentrations observed fall into the distributions simulated at both sites.

Keywords: Ecological risk assessment; Depleted uranium; Monte Carlo simulation; Bioavailability

1. Introduction

An ecological risk assessment is a process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors (US EPA, 1992a, 1998). The process is used to systematically evaluate and organize data, information, assumptions, and uncertainties to help understand and predict the relationships between stressors and ecological effects in a way that is useful for environmental decision-making. Ecological risks can be assessed through field studies; however, performing a large number of these studies may be inappropriate because of the expense in sacrificing receptors and overall cost in obtaining field data. Because of the variety of

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habitats and species in an ecosystem and the associated interactions between biota and physical–chemical conditions, risk assessment is a complex process. Therefore, computer simulation tools are needed for risk assessment and they have become a powerful, cost-effective tool for understanding and managing ecological risks (Carbonell et al., 2000; Sydelko et al., 2001; Naito et al., 2002; Lu et al., 2003).

A computer simulation tool, the ERA model, has been developed for conducting ecological risk assessments (Lu et al., 2003). This tool is based on a preliminary evaluation of existing eco-risk models and includes a Window's based interface, an interactive database management system (DBMS), and a comprehensive evaluation of exposure pathways addressing site- and species- specific estimation of chemical uptake from abiotic and biotic media. Monte Carlo simulations are used for characterizing parameter and risk uncertainty as probabilistic distributions. In the past, risk assessment methods have focused on a single indicator for risk. While this approach has found its usefulness as a screening tool, it does not consider the full range of available information, nor does it explicitly account for important sources of uncertainty in estimating risks (Lahkim et al., 1999; Yegnan et al., 2002). In addition, point estimates of risk may convey an incorrect sense of accuracy and can lead to inconsistencies in making comparisons among risks (Thompson and Graham, 1996). Furthermore, relying on a single value estimate of risk for remedial activity typically results in an over estimation of costs (US EPA, 1992a; Lahkim et al., 1999).

Probabilistic risk assessment differs from the deterministic approach by allowing a value to be chosen from a distribution of plausible values for an exposure variable. Variables that can assume different values for different receptors are referred to as random variables. In a probabilistic risk assessment, one or more (random) variables in the risk equation are defined mathematically by probability distributions. Similarly, the output of a probabilistic risk assessment is a range or distribution of risks experienced by the various members of the population of concern (Warren-Hicks and Moore, 1998). Probabilistic distribution methods have been employed in human (Vermeire et al., 2001), ecological (Jager et al., 2001), and technological risk assessments (Schumacher et al., 2001) to quantify uncertainties in predictions of risks.

When performing an uncertainty analysis with probabilistic distributions generated with Monte Carlo simulations in ecological or human risk assessment, several commercial software packages have been employed (Morgan and Henrion, 1998; Lohman et al., 2000; Moschandreas and Karuchit, 2002). For example, Crystal Ball® was invoked in a probabilistic analysis of regional mercury impacts on wildlife. Another software, @Risk® was used in a probabilistic assessment of screening mercury risks in the Florida Everglades food web (Lohman et al., 2000). These tools are not specific for ecological risk assessment and have been widely applied to assess risks in other fields, such as financial consulting, cost estimate consulting, market research, engineering cost analysis, and insurance. Therefore, the user needs to be aware of how to apply software functions and recreate model equations, input parameters, and the food web for a given application. These procedures are relatively time consuming. In a comprehensive risk assessment, where exposure is addressed via trophic levels of the food web, a spreadsheet approach for performing an uncertainty analysis is not practical as the result is only useful for the one condition studied. Our simulation tool compiles risk assessment algorithms with probabilistic distributions generated through Monte Carlo simulations; parameters and data including the food web are stored in the modifiable DBMS. In this study, we employ the ERA simulation tool to assess risks associated with exposure to depleted uranium (DU) at two US Army sites, Aberdeen Proving Ground (APG) and Yuma Proving Ground (YPG). Concerns have been raised about potential exposure to the associated ecosystems and adverse health effects of DU.

Depleted uranium is a by-product from processing natural uranium to produce the enriched form used as fuel for nuclear reactors or in military applications (Hartmann et al., 2000). Health risks from exposure to DU are a complex issue. Because of the low specific radioactivity and the dominance of α-radiation, no acute risk is likely from external exposure (Bleise et al., 2003). However, internalized DU has a greater potential for adverse impacts than that externalized, such as mutagenic radiological effects where risks are a function of the particle characteristics. Renal, reproductive, and developmental effects from chemical impacts are a function of the route of exposure, duration of exposure, and speciation (Fulco et al., 2000). McClain et al. (2001) studied the primary transport route of DU through wounds and confirmed mutagenic behavior of DU, which transformed human osteoblast cells to a tumorigenic phenotype. The non-radioactive or chemical effect associated with exposure to uranium and its compounds involves renal toxicity, detected by the presence of protein and cell casts in the urine. Additionally, the chemical and radiological impacts of uranium can act synergistically to cause tissue damage. Therefore, it cannot be assumed that cancer is due solely to the radiological effects of uranium or that organ damage is exclusively due to its heavy-metal properties (Fulco et al., 2000).

Since the 1950s, DU has been used as a penetrator in munitions and testing programs at APG, which is located in the western shore of the Chesapeake Bay, a productive and complex ecosystem. The facility provides design and testing of ordnance material in close proxi-

mity to the nation's industrial and shipping centers. As a result of the program, DU has been deposited on over 1500 acres. Most penetrator impacts occurred within about 500 m of the firing axis after the DU munitions passed through soft targets used to check accuracy and performance. Penetrators strike the ground, trees, and wetlands after hitting soft targets and eventually come to rest in the impact area (Ebinger et al., 1996). A second-highly used test area is located at YPG near the Arizona–California border and in the vicinity of the Colorado River, Squaw Lake, and Mittry Lake. YPG began testing DU munitions against soft targets in the 1980s; this firing range comprises 12000 acres (Oxenberg, 1997). Ebinger et al. (1996) reported that redistribution in the arid environment at YPG was mainly due to erosion of DU fragments and redeposition in washes that drain the area. Ingestion of DU by wildlife is likely from consuming DU-contaminated soil accumulated on vegetation or pelts.

In this paper, the components of the ERA model include with exposure pathways, the relation-based food web, ecosystem receptors, risk characterization, and uncertainty analysis. The process for conducting the DU risk assessment is presented, which includes selecting reference values, obtaining concentrations in media, and identifying exposure parameters. The risk assessment is then presented and the model is validated.

2. ERA model

Based on a review by Weiss (1999) and Lu et al. (2003), existing ecological risk assessment models are often site-specific. These models are therefore useful in addressing site-specific issues. However, when databases exist, they are often limited to site-specific conditions and not modifiable, resulting in applications with limited use. General models, which can be easily adapted to other sites, remain few, and are often simple and associated with significant uncertainties. Our ERA model (Lu et al., 2003) is a generic screening tool for ecological risk assessment that can be modified for varying site conditions and ecosystems through a Windows-based interface and interactive modifiable DBMS. Based on trophic sources, a food web has been integrated into the framework of the DBMS.

2.1. Exposure pathways and the food web

Following US EPA and other guidelines (US EPA, 1992b, 1993a; Thomann et al., 1992; Hope, 1995; Cheng, 1998; PNNL, 1998), the ERA model addresses potential exposure pathways of ingestion, inhalation, and dermal absorption for terrestrial animals; root and foliar uptake for plants; and direct absorption for aquatic species. Each mathematical equation for exposure incorporates species-specific information on diet composition, body weight, home range, food and water ingestion rates, and incidental ingestion rates of environmental media. Given a specified set of possible exposure pathways and routes, these equations can be combined to produce site- and species- specific estimates of chemical uptake from abiotic and biotic media. The exposure algorithms applied to the ERA model are based on a compilation of studies (Maughan, 1993; US EPA, 1993a,b; Farago, 1994; Hope, 1995; Cheng, 1998; PNNL, 1998). These exposure models for terrestrial and aquatic plants and animals are used in software developed with Visual Basic 6.0. The DBMS provides robust storage and retrieval capabilities and can solve problems, such as data redundancy and inconsistency, data relationship definition, and security problems. Based on these advantages, the Microsoft Access DBMS was selected to handle data in this model (Lu et al., 2003). The parameters associated with exposure models including benchmarks, site characteristics, chemical properties, and exposure parameters are stored in the database. Furthermore, the DBMS is linked to external databases such as the US EPA ECOTOX to address site-specific applications.

2.2. Animal and plant receptors at APG and YPG

Generally, assessment endpoints are explicit expressions of the environmental value that is to be protected, operationally defined by an ecological entity and its attributes (US EPA, 1998). Various endpoints may be used for predictive assessments, but the final selection is often affected by the availability of toxicity data in the literature and the quality of the data. Criteria were identified to provide guidance for defining the endpoint receptors (US EPA, 1992a; PNNL, 1998): (1) Commercial or recreational importance; (2) Protection status under the Endangered Species Act or similar state legislation; (3) Critical component of either the terrestrial or aquatic, ecosystem: key predator or prey; (4) High potential exposure to contaminants; (5) Availability of toxicological information for the species; and, (6) Representative of a foraging guild. In addition, the species listed as "threatened, endangered, and sensitive species on DOD lands" by the US Army (Martin and Fischer, 2000) have also been included (Lu et al., 2003).

APG and YPG were identified as baseline ecosystems for the ERA model, which represent coastal and desert ecosystems, respectively. Considering the diversity of the APG ecosystem and the large area of YPG, a significant number of wildlife species live within the two sites. Following US EPA (1998) guidance and criteria above and considering databases and records maintained by the federal and state agencies including those associated with the two proving grounds (Lu et al., 2003), the list of receptors are shown in Table 1.

Table 1
Selected receptors of APG and YPG (adapted from Lu et al. (2003))

Species category	Aberdeen proving ground	Yuma proving ground
Birds	Mallard, American kestrel, barred owl, bald eagle	Mexican spotted owl, loggerhead shrike, gamble's quail
Mammals	White-tailed deer, beaver, white-footed mouse, cottontail rabbit, Indiana bat	Kit fox, cactus mouse, black-tailed jackrabbit, mule deer, lesser long-nosed bat
Reptiles and amphibians	Eastern garter snake, lizards, woodhouse's toad	Desert tortoises, sonoran whipsnake, desert spiny lizard
Aquatic animals	Whitefish, pacific lamprey, white sturgeon, rainbow trout	NA[a]
Aquatic plants	Water millfoil, phytoplankton, periphyton	NA
Terrestrial plants	Fern, rushes, slender blue flag	Creosote bush, foothill paloverde trees, saguaro cactus

[a] NA: not applicable.

2.3. Risk characterization and uncertainty analysis

Once the ecosystem and site characteristics are fully understood, the applied daily dose (ADD) or body burden can be estimated for an individual receptor. An ecological hazard quotient (EHQ) is then calculated by dividing the $ADD_{pathway}$ (or body burden) by the reference value:

$$EHQ = ADD_{pathway} \div \text{reference value} \quad (1)$$

The reference value recommended in this model is the no observed adverse effect level (NOAEL) or no observed adverse effect concentration (NOAEC) for terrestrial and aquatic species, respectively. The NOAEL and NOAEC are derived from experiments conducted on laboratory species, and represent the highest dose or contaminant concentration applied that did not result in a measurable adverse effect in the 95% of potential population (Cockerham and Shane, 1994; Sample et al., 1998; Weiss, 1999). For example, uranium reference values for terrestrial animals represent doses that did not adversely affect the receptor's reproductive system; for terrestrial plants the exceedance of the benchmark represents potential reduction in the plants root weight at a 20% level of effects. The reference values for aquatic species are the highest doses that did not increase mortality at a 20% level of effects (Sample et al., 1998).

Based on the selected reference values, the EHQ represents varying levels of risk or measures of levels of concern (Tannenbaum et al., 2003). Although risk categories are outlined here, receptor risk should be evaluated individually based on the endpoint. An EHQ less than 1 indicates the toxicological effects are unlikely to occur and hence the potential for unacceptable risk is minimal (Tannenbaum et al., 2003). A NOAEL-based EHQ greater than 1 but less than the LOAEL (lowest observed adverse effect level) indicates that effects are possible but uncertain. Finally a LOAEL-based EHQ > 1 indicates that effects are probable and exposure exceeded the lowest dose associated with effects. The EHQ value provides an indication of level of risk to a receptor.

In the risk assessment, as discussed previously, uncertainties are inherent because the data and understanding of the ecosystem may be limited. Therefore, probability density functions were sampled using Monte Carlo simulations. By applying the simulation, distribution characteristics were studied and convergence revealed a minimum iteration of 500 based on the 95th confidence level, which is in agreement with Tellinghuisen (2000). However, in this study, the selected iteration is based on a 99th confidence level, as we are interested in the lower probability outcomes at the tails of the distributions. In this case, 1000 iterations were selected (Frey and Rhodes, 1998).

Probabilistic distributions have been used as a tool to qualify uncertainty in predicting risks to humans and ecological receptors (Frey and Rhodes, 1998). Risk is defined as an adverse change or condition resulting from a stressor (Bartell et al., 1992; Lackey, 1997). The distributions characterize the degree of belief that the true but unknown value of a parameter lies within a specified range of values for that parameter (Warren-Hicks et al., 2002). Criteria for selecting a distribution are based on National Council on Radiation Protection and Measurements (NCRP, 1996) and US EPA (1998) guidelines. The distribution should represent site-specific uncertainty and variation in that parameter (Schumacher et al., 2001). Also, the distribution must represent the range of values for that parameter in a given system. The selected distribution should be consistent between sites for specific parameters (Warren-Hicks et al., 2002). Moreover, the form of the distribution should reflect the magnitude, range, and interpretation of the parameter (NCRP, 1996). For example, contaminant concentration cannot be negative; therefore, the sampling distribution should reflect the restricted range. The probabilistic distributions of the exposure parame-

Table 2
Input variables used in the Monte Carlo simulation for DU case study

Parameter	Definition	Unit	Distribution	References
EC	Contaminant concentration	mg/l (water), mg/kg (soil), mg/m^3 (air)	Log normal	Hattis et al. (2001), Hertwich et al. (1999), McKone (1993), Ott (1990), Polder et al. (1998), Smith (1994), Stow and Qian (1998), Travis and Arms (1988), Veith et al. (1980)
BW	Body weight	kg	Normal	Hertwich et al. (1999), McKone (1993), Briggs et al. (1983), Kenaga and Goring (1980), MacIntosh et al. (1994), US EPA (1997a), Wiwatanadate and Claycamp (2000)
IR_f	Food ingestion rate	kg/day	Normal	McKone (1993), Briggs et al. (1983), MacIntosh et al. (1994), US EPA (1997a)
IR_{dw}	Ingestion rate of drinking water	l/day	Normal	McKone (1993), MacIntosh et al. (1994), US EPA (1997a)
IR_i	Inhalation rate	M^3/day	Normal	Hertwich et al. (1999), US EPA (1997a)
SA	Surface area	cm^2	Normal	McKone (1993), US EPA (1997a)
AF	Soil-to-skin adherence factor	mg/cm^2	Default value: 1	Hope (1999), US EPA (1993a)
α_d	Contaminant-specific dermal absorption factor	mg/kg (body burden)/ mg/kg (daily dose)	Default value: 0.01	Hope (1999), US EPA (1993a)
P_{cs}	Fraction of receptor surface area in contact with soil per day	d^{-1}	Default value: 0.22	Kenaga and Goring (1980), Hope (1999), US EPA (1993a)
θ	Site use factor	Ratio of contaminant area to home range	Default value: 1	Kenaga and Goring (1980), Hope (1999)
ψ	Seasonal factor	Fraction of time per year receptor occurs at site	Default value: 1	Kenaga and Goring (1980), Hope (1999)
BCF	Bio-concentration factor	l/kg	Log normal	McKone (1993), Kenaga and Goring (1980), Hope (1995), Lahkim et al. (1999), McKone (1994), Nayak and Kundu (2001), West and Kodell (1999)
B_v	Bio-concentration factor for vegetative plant parts	mg/kg (soil)/mg/kg (vegetative plant)	Log normal	Hope (1999), US EPA (1993a), West and Kodell (1999), Absallom et al. (1999), Finley et al. (1994)
B_r	Bio-concentration factor for non-vegetative plant parts	mg/kg (soil)/mg/kg (vegetative plant)	Log normal	Hope (1999), US EPA (1993a), West and Kodell (1999), Absallom et al. (1999), Finley et al. (1994)

ters were gathered from a number of studies and are summarized in Table 2. As the log normal distribution has a longer tail than other distributions, it is widely used in environmental analysis to represent positively valued data exhibiting positive skewness (NCRP, 1999; Cullen and Frey, 1999). Pollutant concentration tends to be log normally distributed, which has been explained by the theory of successive random dilutions (Ott, 1990). After the pollutants are emitted from a source, they undergo successive mixing and dilution, resulting in a log normal frequency distribution. Furthermore, a goodness of fit test was conducted to assess the appropriateness of the log normal distribution for sampling data at both APG and YPG sites. By using the Anderson-Darling (A) test, the log normal distribution was found to be the most appropriate for the DU data. Therefore, in this study, the log normal distribution is selected to represent DU concentrations in the media. Both aquatic species bio-concentration factors and soil to plant uptake factors are defined as the equilibrium concentration in tissues to that in water or soil based field and/or laboratory data (Jorgensen et al., 1991; PNNL, 1998; Sample et al., 1998). The associated distributions have been observed as skewed, which has led to the use of the logarithmic transformation of the parameter to obtain the log normal distribution (Traas et al., 1996; Verhaar et al., 1999; Samsoe-Petersen et al., 2002; Liao et al., 2003).

Physiological parameters such as body weight, surface area, and ingestion and inhalation rates in terrestrial animals may vary seasonally, geographically, and by age. These parameters typically follow a Gaussian distribution (US EPA, 1993a, 1997b). The normal distribution is commonly used to represent uncertainty resulting from unbiased measurement errors (Morgan and Henrion, 1998). Because the normally distributed random variable takes on values over the entire range of real data, the standard deviation is a measure of the

population variance. Surface area, ingestion, and inhalation rates are a function of the body weight and are often estimated using allometric equations (US EPA, 1993a).

With limited field or laboratory data, conservative values are recommended (Hope, 1995, 1999). The US EPA applied such an approach for soil to skin adherence factors and the contaminant specific dermal absorption factor (US EPA, 1989, 1993a, 2001). Moreover, because of limited data, these values were based on exposure for humans not terrestrial animals to which they were applied (US EPA, 1989; Hope, 1995). Therefore, in this study, a similar approach was used for parameters related to dermal contact (Table 2): soil to skin adherence factor, contaminant specific dermal absorption factor, soil contact fraction factor, and site use factor.

3. Risk assessment

The US EPA framework for ERAs consists of problem formulation, analysis, risk characterization, and risk management and communication (US EPA, 1998). Problem formulation involves data collection, hazard identification, assessment of endpoints, and development of a plan for exposure assessment. Detailed exposure and ecological effects assessments are the two basic components covered under the analysis phase. Risk then can be characterized based on exposure and toxicity assessment. Mitigation measures and communicating risk to interested groups is the last step in the risk assessment process (US EPA, 1998; Sadiq et al., 2003). The case study for the DU assessment is conducted following this framework.

Once the ecosystem is defined along with the food web, the process for implementing the DU risk assessment begins with selecting reference values, obtaining concentrations in media, and identifying exposure parameters. In the following sections, we present reference value selection, DU concentrations in media, risk characterization with a detailed exposure and toxicity assessment, and lastly model validation.

3.1. Reference value

The relevant NOAEL and NOAEC data were identified from multiple sources for the terrestrial and aquatic receptors in this case study (Sample et al., 1996; Efroymson et al., 1997; US EPA, 2003). In instances where toxicological data for receptors were unavailable, surrogate species were selected based on taxonomy, life style, and/or toxicological response similarity. Surrogate application requires applying a conversion method based on test species and the receptor's body weights. Wildlife

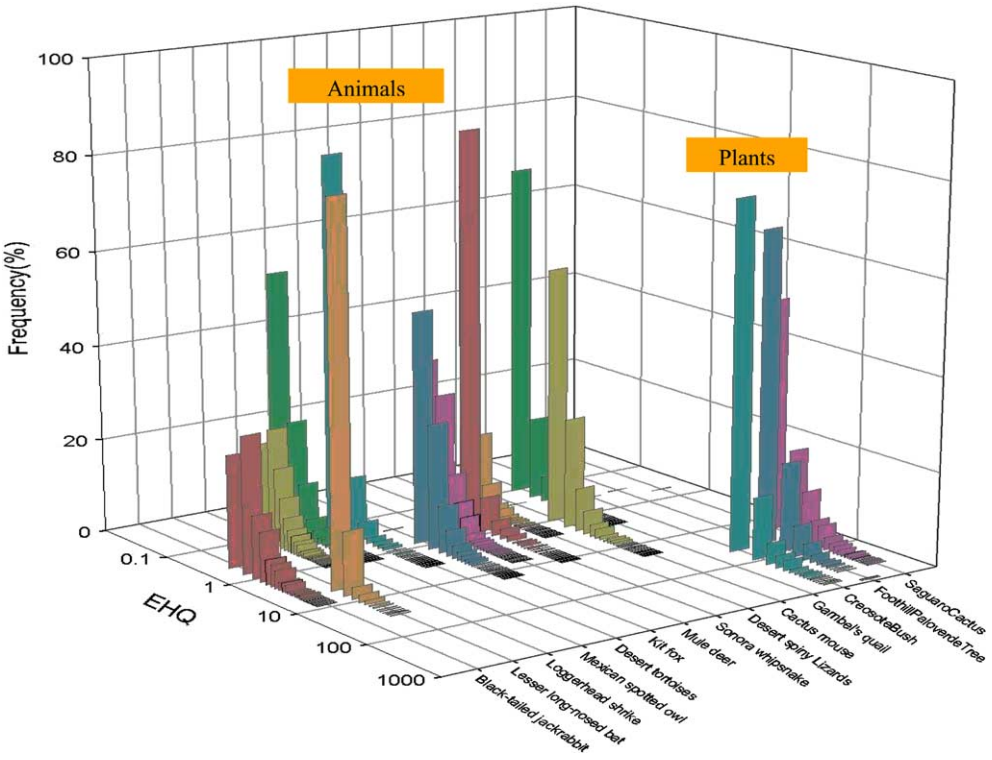
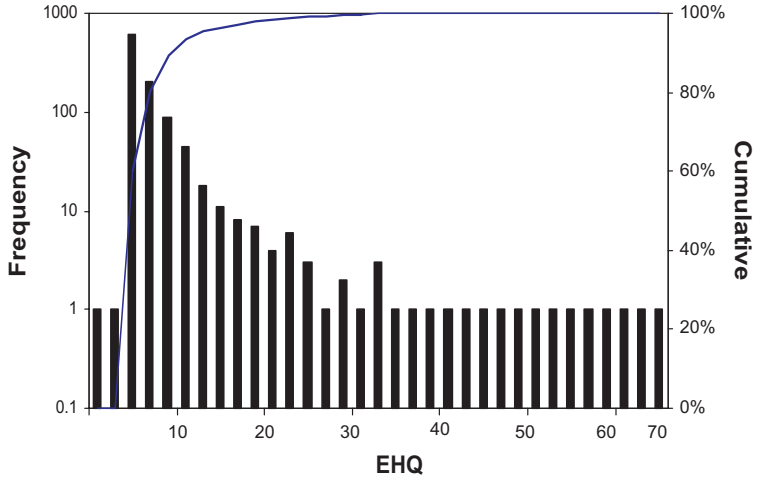


Fig. 1. EHQ distributions for YPG terrestrial receptors.



Statistical data	
Mean	3.86E+00
Standard Error	1.08E-01
Median	3.05E+00
Standard Deviation	3.41E+00
Sample Variance	1.16E+01
Kurtosis	1.68E+02
Skewness	1.07E+01
Range	6.77E+01
Minimum	2.18E+00
Maximum	6.99E+01
Sum	3.86E+03
Count	1.00E+03
Confidence Level(95.0%)	2.11E-01

Fig. 2. Statistical data for EHQ (lesser long-nosed bat).

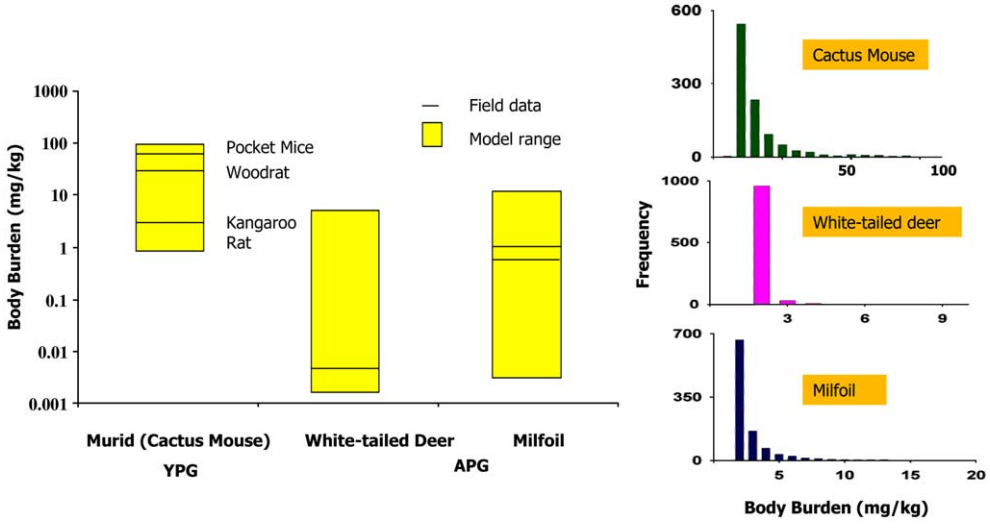


Fig. 3. ERA modeling validation on DU.

NOAELs can be estimated for an untested species by the following equation (Sample and Arenal, 1999):

$$\mathrm{NOAEL}_{\mathrm{wildlife}} = \mathrm{NOAEL}_{\mathrm{test}} \left(\frac{\mathrm{bw}_{\mathrm{test}}}{\mathrm{bw}_{\mathrm{wildlife}}} \right)^{1-b} \quad (2)$$

where the $\mathrm{NOAEL}_{\mathrm{wildlife}}$ represents the ecosystem receptor of concern, the $\mathrm{NOAEL}_{\mathrm{test}}$ is the surrogate test species for which the NOAEL is available, 'bw' represents their respective body weights, and b is an allometric scaling factor. From Sample and Arenal (1999), scaling factors of 1.2 and 0.94 are recommended for birds and mammals, respectively. NOAEL data on test species, mouse and black duck, were used to calculate other untested species NOAEL values based on Eq. (2). (Toxicological data are available in Appendix A.)

3.2. DU concentrations in media

As discussed previously, the log normal distribution was applied to describe DU concentrations in both water and soil for APG and YPG. Sampling data on uranium concentrations in surface water, groundwater, and soils from APG and YPG were collected by Ebinger et al. (1996) and stored in a database developed and maintained by Los Alamos National Laboratory (Ebinger, 2002). At APG, uranium concentrations in the surface- and ground-water samples were analyzed based on nine samples collected near the western shore of the Chesapeake Bay. Potentially impacted soils over 1500 acres were sampled mainly in conjunction with well water; a total of 35 samples were collected representing an extremely limited data set. (See Appendix B for sampling area and associated data.)

YPG is characterized as a typical desert ecosystem; therefore field studies were conducted, for the most part, on soil samples. Ebinger et al. (1996) established sample plots on two firing ranges at YPG. Plots were distributed non-randomly along the area of 12000 acres, where first penetrator impacts were closely clustered and had been identified as exhibiting elevated levels of DU contamination (Price, 1991; Ebinger et al., 1996; Oxenberg, 1997). These areas were situated along the axis of the firing line and could be identified by impact craters, recently displaced soils, and DU fragments. Locations for sample plots varied along the firing line and from observable impact craters and according to Ebinger et al. (1996) were assumed to cover a range of contaminant levels for each firing line. According to US EPA's soil sam-

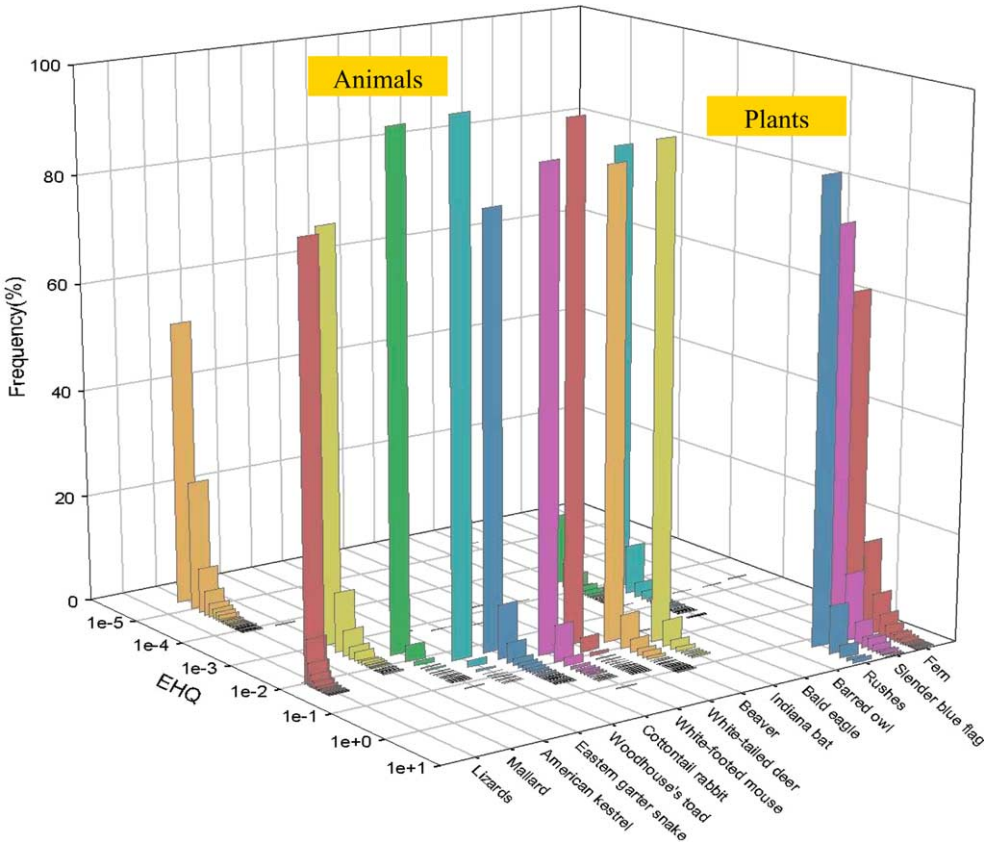


Fig. 4. EHQ distributions for APG terrestrial receptors.

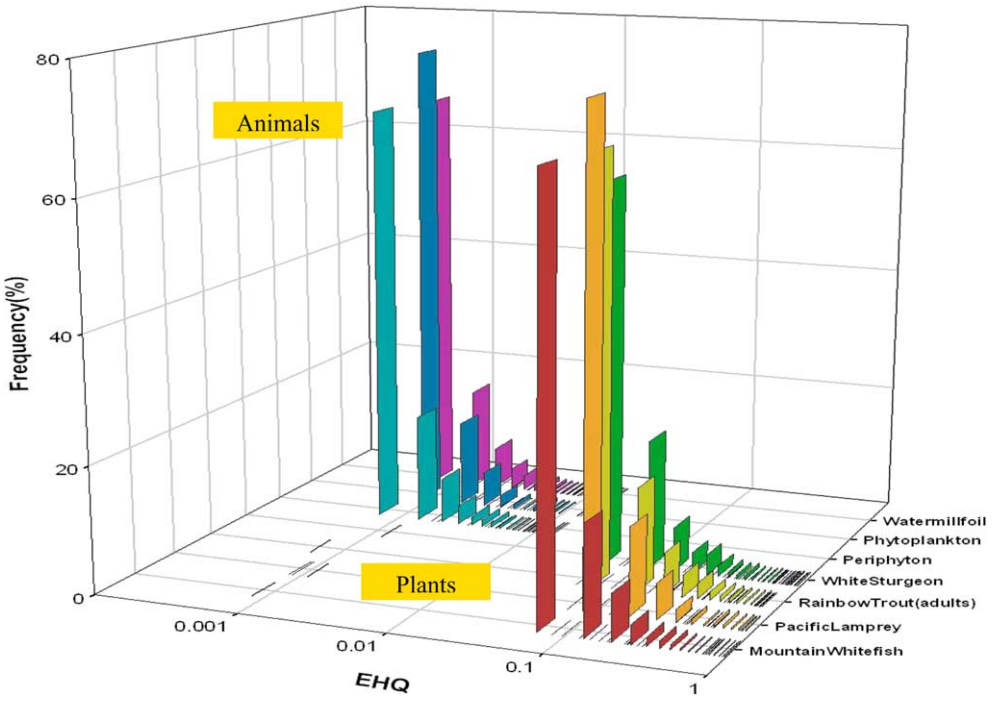


Fig. 5. EHQ distributions for APG aquatic receptors.

Table A.1
Uranium toxicological data for terrestrial wildlife

Analyte	Form[a]	Test species	Test NOAEL[b] (mg/kg/d)	Endpoint	Estimated NOAEL[c,d] (mg/kg/d)
$UO_2(CH_2COOH)_2$	$UO_2CO_{3(AQ)}$ $UO_2(OH)^+$	Mouse	3.07	Little brown bat	3.322
$UO_2(CH_2COOH)_2$	$UO_2CO_{3(AQ)}$ $UO_2(OH)^+$	Mouse	3.07	Short-tailed shrew	3.187
$UO_2(CH_2COOH)_2$	$UO_2CO_{3(AQ)}$ $UO_2(OH)^+$	Mouse	3.07	White-footed mouse	3.115
$UO_2(CH_2COOH)_2$	$UO_2CO_{3(AQ)}$ $UO_2(OH)^+$	Mouse	3.07	Meadow vole	2.988
$UO_2(CH_2COOH)_2$	$UO_2CO_{3(AQ)}$ $UO_2(OH)^+$	Mouse	3.07	Mink	2.477
$UO_2(CH_2COOH)_2$	$UO_2CO_{3(AQ)}$ $UO_2(OH)^+$	Mouse	3.07	Cottontail rabbit	2.45
$UO_2(CH_2COOH)_2$	$UO_2CO_{3(AQ)}$ $UO_2(OH)^+$	Mouse	3.07	Red fox	2.263
$UO_2(CH_2COOH)_2$	$UO_2CO_{3(AQ)}$ $UO_2(OH)^+$	Mouse	3.07	River otter	2.187
$UO_2(CH_2COOH)_2$	$UO_2CO_{3(AQ)}$ $UO_2(OH)^+$	Mouse	3.07	White-tail deer	1.945
$U_{(s)}$	$DU_{(s)}$	Black duck	16	Rough-winged swallow	6.684
$U_{(s)}$	$DU_{(s)}$	Black duck	16	American robin	9.163
$U_{(s)}$	$DU_{(s)}$	Black duck	16	Belted kingfisher	10.442
$U_{(s)}$	$DU_{(s)}$	Black duck	16	American woodcock	11.068
$U_{(s)}$	$DU_{(s)}$	Black duck	16	Cooper's hawk	12.979
$U_{(s)}$	$DU_{(s)}$	Black duck	16	Barn owl	13.135
$U_{(s)}$	$DU_{(s)}$	Black duck	16	Barred owl	14.317
$U_{(s)}$	$DU_{(s)}$	Black duck	16	Red-tailed hawk	15.669
$U_{(s)}$	$DU_{(s)}$	Black duck	16	Osprey	16.594
$U_{(s)}$	$DU_{(s)}$	Black duck	16	Great blue heron	18.215

[a] pH: 6–7.
[b] Sample et al. (1996).
[c] $b = 0.94$ mammals and 1.2 birds (Sample and Arenal, 1999).
[d] NOAEL: 0.9 (mg/kg/d) (for Lizards (side-blotched), Western aquatic garter snake, Woodhouse's toad (adult)) (PNNL, 1998).

pling protocol (US EPA, 1992c), when a plume is suspected and the orientation of the plume can be estimated, the sampling grid should be oriented in such a manner that the extending axis of the grid is parallel

Table A.2
Uranium toxicological data for terrestrial plants

Analyte	Form[a]	Test species[b]	Test LOEC (mg/kg)	Endpoint
$UO_2(NO_3)_2$	$UO_2CO_{3(AQ)}$ $UO_2(OH)^+$	Swiss chard	5	Fern
$UO_2(NO_3)_2$	$UO_2CO_{3(AQ)}$ $UO_2(OH)^+$	Swiss chard	5	Rushes
$UO_2(NO_3)_2$	$UO_2CO_{3(AQ)}$ $UO_2(OH)^+$	Swiss chard	5	Slender blue flag
$UO_2(NO_3)_2$	$UO_2CO_{3(AQ)}$ $UO_2(OH)^+$	Swiss chard	5	Creosote bush,
$UO_2(NO_3)_2$	$UO_2CO_{3(AQ)}$ $UO_2(OH)^+$	Swiss chard	5	Foothill paloverde trees
$UO_2(NO_3)_2$	$UO_2CO_{3(AQ)}$ $UO_2(OH)^+$	Swiss chard	5	Saguaro cactus

[a] pH: 6–7.
[b] Efroymson et al. (1997).

Table A.3
Uranium toxicological data for aquatic species

Analyte	Form[a]	Species	Test NOAEC[b] (mg/l)	Aquatic species
$UO_2(NO_3)_2$	$UO_2CO_{3(AQ)}$ $UO_2(OH)^+$	Periphyton	2	Aquatic plants[c]
$UO_2(NO_3)_2$	$UO_2CO_{3(AQ)}$ $UO_2(OH)^+$	Phytoplankton	2	
$UO_2(NO_3)_2$	$UO_2CO_{3(AQ)}$ $UO_2(OH)^+$	Water milfoil	2	
$UO_2(NO_3)_2$	$UO_2CO_{3(AQ)}$ $UO_2(OH)^+$	Mountain whitefish	0.021	Aquatic animals[d]
$UO_2(NO_3)_2$	$UO_2CO_{3(AQ)}$ $UO_2(OH)^+$	Pacific lamprey	0.021	
$UO_2(NO_3)_2$	$UO_2CO_{3(AQ)}$ $UO_2(OH)^+$	Rainbow trout (adults)	0.021	
		Rainbow trout (edds)		
		Rainbow trout (larvae)		
$UO_2(NO_3)_2$	$UO_2CO_{3(AQ)}$ $UO_2(OH)^+$	White sturgeon	0.021	

[a] pH 6–7.
[b] Ecological Toxicity Database (US EPA, 2003).
[c] Surrogate aquatic plants are *Chlorella vulgaris* and Green algae.
[d] Surrogate aquatic animals are Fathead minnow.

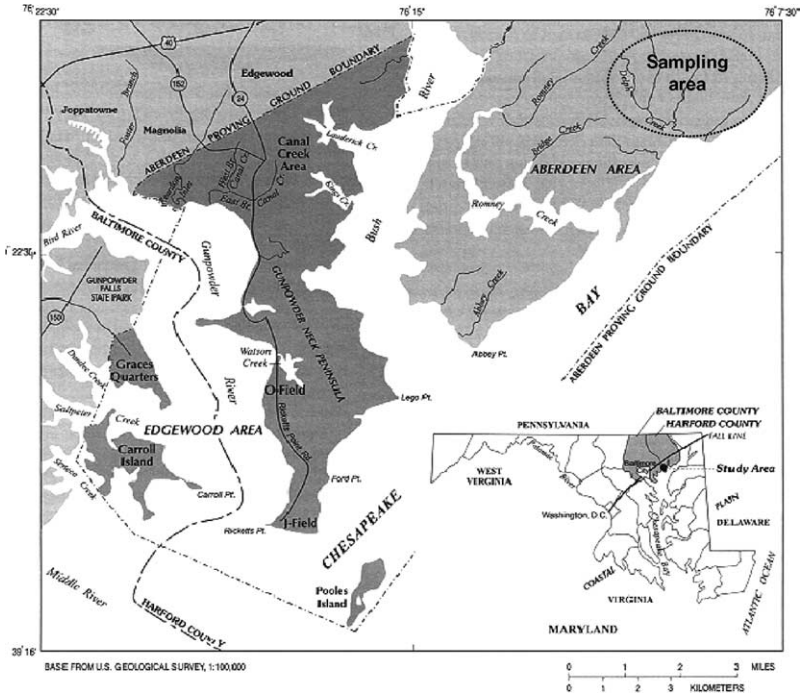


Fig. B.1. APG area, Maryland (adapted from Donnelly and Tenbus (1998)).

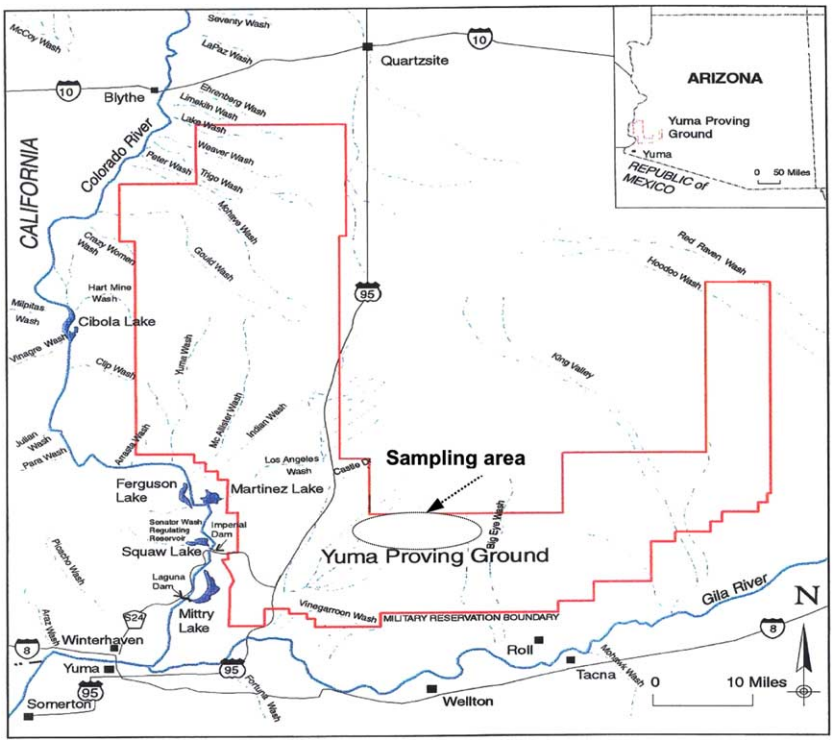


Fig. B.2. YPG area, Arizona (adapted from Entech Engineers, Inc. (1988)).

to the suspected plume; however, this is not necessary and a square or rectangular grid is one of the most useful for reconnaissance. DU concentrations in soil were based on 22 samples, again a very limited data set for the impacted area. (See Appendix B for sampling area and associated data.)

3.3. Risks results

Based on speciation, $UO_2CO^0_{3(AQ)}$ and $UO_2(OH)^+$ are the two dominant and mobile species at pH 6–7 and pE 5–15 that may adversely affect receptors from exposure. For YPG terrestrial plants (Fig. 1), because of high DU concentrations in soil, the resulting distributions suggest a 98% likelihood of a reduction in root weight. For terrestrial animals at YPG such as Mexican spotted owl, loggerhead shrike, gamble's quail, desert spiny lizard, and desert tortoises, given soil concentrations, the dose is less than that resulting in a decrease in offspring and probability analysis showed the likelihood for adverse reproduction effects ranges from 0.1% to 0.6%. For the kit fox, cactus mouse, black-tailed jackrabbit, mule deer, and sonora whipsnake, the likelihood ranges from 9% to 44%. However, for the lesser long-nosed bat, the effects are expected to occur (>99% likelihood) through the reduction in size and weight of offspring. Among the different exposure pathways for the bat, including ingestion, inhalation, and dermal absorption, the dominant pathway is through insect ingestion, which accounts for 97% of its diet. Furthermore, insect exposure includes ingestion pathways—soil, water, and food (plants), as well as dermal and inhalation routes. Based on characteristics of terrestrial animals and their responses to DU exposure, the bat is more vulnerable than other terrestrial species; the positive skewness of the risk distribution exemplifies it's sensitivity (Fig. 2).

From the field studies (Ebinger et al., 1996), pocket mice, kangaroo rat, and white-throated woodrat samples were analyzed for uranium concentrations to estimate risk levels at YPG. Samples of carcasses, kidneys, and livers from these animals were collected for identifying uranium concentrations. For pocket mice, the greatest uranium concentration was found in carcass samples, 115.4 $mg\,kg^{-1}$; for kangaroo rat, the worst case was observed in kidney samples 4.3 $mg\,kg^{-1}$; and for white-throated woodrat, the greatest concentration of uranium was 76.7 $mg\,kg^{-1}$ in carcass samples. Based on our risk assessment, a receptor from the same family Murid, cactus mouse, exhibited a uranium concentration of 2.46 to 224.6 $mg\,kg^{-1}$. Sampling data from Murid receptors, pocket mice, kangaroo rat, and white-throated woodrat, fall into associated distributions predicted in the ERA tool (Fig. 3).

At APG, based on limited DU data, exposure potentially poses little risk (<0.3% likelihood) for terrestrial animals (Fig. 4), suggesting no observable impact on receptor's reproduction or development. Ebinger et al. (1996) collected deer samples to evaluate potential DU uptake and transfer to humans who consume deer. They analyzed kidney, livers, muscle, and bone samples, and found that the greatest uranium concentration among those samples was 0.0051 $mg\,kg^{-1}$, which falls in the distribution observed here of 0.0042 to 7.3 $mg\,kg^{-1}$ for the receptor, white-tailed deer (Fig. 3). For APG terrestrial plants (Fig. 4), modeling results showed 51%, 24%, and 27% likelihood of a reduction in root weight for rushes, slender blue flag, and fern, respectively.

Compared with terrestrial plants at APG and again this is based on a very limited set of data, uranium uptake potentially does not pose a risk to aquatic plants (Fig. 5). Considering DU exposure to aquatic animals at APG, its uptake is potentially not expected to increase mortality (<0.1% likelihood). For the aquatic plant, milfoil, two samples were collected (Ebinger et al., 1996) from field studies, where 2.1 and 0.8 $mg\,kg^{-1}$ of uranium were observed. Our modeling results showed that the uranium concentration in milfoil ranged from 6.4×10^{-3} to 18.6 $mg\,kg^{-1}$, and are consistent with field data (Fig. 3). In addition, their results (Ebinger et al., 1996) indicated that the presence of DU was confirmed by isotopic ratios observed in the cattail and pickerel weed, representing uptake, attachment, or adsorption of DU from water or sediments where these aquatic organisms grow.

4. Conclusions

Risks from exposure to DU at two US Army sites, APG and YPG, were characterized based on available data. Exposure pathways for terrestrial and aquatic plants and animals were applied in software developed using Visual Basic 6.0 with associated parameters stored in the Microsoft Access DBMS. To characterize risk and address uncertainty, the model employs Monte Carlo simulations for assessing parameter and risks as probabilistic distributions. Results from the ERA model suggest that at YPG, a reduction in plant root weight is considered likely to occur (98% likelihood) from exposure to uranium. For terrestrial animals at YPG such as Mexican spotted owl, loggerhead shrike, gamble's quail, desert spiny lizard, and desert tortoises, probability analysis showed the likelihood for adverse reproduction effects ranges from 0.1% to 0.6%. For kit fox, cactus mouse, black-tailed jackrabbit, mule deer, and sonora whipsnake, likelihood ranges from 9% to 44%. However, for the lesser long-nosed bat, the effects are expected to occur (>99% likelihood) through the reduction in size and weight of offspring. At APG, uranium uptake will not likely affect survival of aquatic plants and animals (<0.1% likelihood). However, data were limited reflecting the risk observed and further field investigations at both sites are recommended. Through model validation, the results from the ERA model are consistent with sampling data from field studies of Ebinger et al. (1996).

To accurately address contaminant mobility and bioavailability, the ERA is currently being linked with speciation and transport models to account for spatial and temporal aspects. The resulting simulation tool will assist in better quantifying receptor exposure and support advancing the ability to apply mobile and

Table B.1
Uranium concentrations in media at APG and YPG (adapted from Ebinger et al. (1996))

Sample no.	YPG in soil (mg/kg)	APG	
		In soil (mg/kg)	In water (mg/l)
1	220.6	17.28	1.71×10^{-4}
2	43.22	2.7	9.90×10^{-4}
3	110.42	5.94	5.10×10^{-4}
4	140.6	86.4	3.30×10^{-5}
5	21.05	9.18	1.86×10^{-4}
6	43.22	7.29	9.90×10^{-4}
7	602.6	5.13	9.60×10^{-4}
8	822.8	11.07	1.03×10^{-2}
9	55.26	1.19	1.01×10^{-3}
10	21.15	0.95	
11	1205.6	4.05	
12	1404.2	0.84	
13	24.12	0.81	
14	41.27	0.54	
15	2.7	0.27	
16	0.21	7.56	
17	25.04	5.4	
18	13.47	0.27	
19	26.94	1.81	
20	38.11	0.27	
21	0.0025	1.0	
22	100.44	0.19	
23		1.11	
24		0.3	
25		2.19	
26		0.49	
27		0.54	
28		0.27	
29		2.7	
30		0.38	
31		1.4	
32		0.65	
33		0.43	
34		0.35	
35		2.19	

available concentrations found in subsurface environments.

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Appendix A

See Tables A.1–A.3.

Appendix B

See Figs. B.1, B.2 and Table B.1.

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