Biological Pathways of Exposure and Ecotoxicity Values for Uranium and Associated Radionuclides

By Jo Ellen Hinck, Greg Linder, Susan Finger, Edward Little, Donald Tillitt, and Wendy Kuhne

Chapter D of

Hydrological, Geological, and Biological Site Characterization of Breccia Pipe Uranium Deposits in Northern Arizona

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Conversion Factors

Inch/Pound to SI

Multiply	By	To obtain
	Mass	
ton, short (2,000 lb)	0.9072	megagram (Mg)

SI to Inch/Pound

Multiply	By	To obtain
	Length	
millimeter (mm)	0.03937	inch (in.)
	Specific volume	
liter per kilogram (L/kg)	27.68	cubic inch per pound (in^3/lb)
	Density	
milligram per cubic meter (mg/m^3)	0.00000006242	pound per cubic foot (lb/ft^3)
microgram per cubic centimeter ($\mu g/cm^3$)	0.0000624220	pound per cubic foot (lb/ft^3)
	Surface density	
microgram per square centimeter ($\mu g/cm^2$)	0.0000002276	ounce per square inch (oz/in^2)
	Concentration	
milligram per liter (mg/L)	1.0	parts per million (ppm)
microgram per liter (µg/L)	1.0	parts per billion (ppb)
milligram per kilogram (mg/kg)	0.000016	ounce per pound (oz/lb)
nanomole per cubic centimeter ($nmol/cm^3$)	1,000	nanomole per liter (nmol/L)
	Radioactivity	
becquerel (Bq)	3.7×10^{10}	curie (Ci)
becquerel per kilogram (Bq/kg)	37	picocurie per gram (pCi/g)
gray (Gy)	0.01	rad (rad)
sievert (Si)	0.01	rem (rem)
coulomb/kg in air	2.58×10^{-4}	roentgen (R)

Chapter D

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Abstract

This chapter compiles available chemical and radiation toxicity information for plants and animals from the scientific literature on naturally occurring uranium and associated radionuclides. Specifically, chemical and radiation hazards associated with radionuclides in the uranium decay series including uranium, thallium, thorium, bismuth, radium, radon, protactinium, polonium, actinium, and francium were the focus of the literature compilation. In addition, exposure pathways and a food web specific to the segregation areas were developed. Major biological exposure pathways considered were ingestion, inhalation, absorption, and bioaccumulation, and biota categories included microbes, invertebrates, plants, fishes, amphibians, reptiles, birds, and mammals. These data were developed for incorporation into a risk assessment to be conducted as part of an environmental impact statement for the Bureau of Land Management, which would identify representative plants and animals and their relative sensitivities to exposure of uranium and associated radionuclides. This chapter provides pertinent information to aid in the development of such an ecological risk assessment but does not estimate or derive guidance thresholds for radionuclides associated with uranium.

Previous studies have not attempted to quantify the risks to biota caused directly by the chemical or radiation releases at uranium mining sites, although some information is available for uranium mill tailings and uranium mine closure activities. Research into the biological impacts of uranium exposure is strongly biased towards human health and exposure related to enriched or depleted uranium associated with the nuclear energy industry rather than naturally occurring uranium associated with uranium mining. Nevertheless, studies have reported that uranium and other radionuclides can affect the survival, growth, and reproduction of plants and animals.

Exposure to chemical and radiation hazards is influenced by a plant's or an animal's life history and surrounding environment. Various species of plants, invertebrates, fishes, amphibians, reptiles, birds, and mammals found in the segregation areas that are considered species of concern by State and Federal agencies were included in the development of the site-specific food web. The utilization of subterranean habitats (burrows in uranium-rich areas, burrows in waste rock piles or reclaimed mining areas, mine tunnels) in the seasonally variable but consistently hot, arid environment is of particular concern in the segregation areas. Certain species of reptiles, amphibians, birds, and mammals in the segregation areas spend significant amounts of time in burrows where they can inhale or ingest uranium and other radionuclides through digging, eating, preening, and hibernating. Herbivores may also be exposed though the ingestion of radionuclides that have been aerially deposited on vegetation. Measured tissues concentrations of uranium and other radionuclides are not available for any species of concern in the segregation areas. The sensitivity of these animals to uranium exposure is unknown based on the existing scientific literature, and species-specific uranium presumptive effects levels were only available for two endangered fish species known to inhabit the segregation areas.

Overall, the chemical toxicity data available for biological receptors of concern were limited, although chemical and radiation toxicity guidance values are available from several sources. However, caution should be used when directly applying these values to northern Arizona given the unique habitat and life history strategies of biological receptors in the segregation areas and the fact that some guidance values are based on models rather than empirical (laboratory or field) data. No chemical toxicity information based on empirical data is available for reptiles, birds, or wild mammals; therefore, the risks associated with uranium and other radionuclides are unknown for these biota.

Introduction

Proposed uranium mining in areas adjacent to the Grand Canyon National Park, Ariz., has prompted the U.S. Department of the Interior (DOI) to investigate physical, chemical, and biological issues potentially affected by mining. On July 21, 2009, U.S. Secretary of the Interior Ken Salazar proposed that about 1 million acres of Federal lands near the Grand Canyon be withdrawn from consideration for future mining activity. The land under consideration is contained in three parcels: two Bureau of Land Management (BLM) parcels on the North Rim of the Grand Canyon and one U.S. Forest Service (USFS) parcel on the South Rim. The Secretary's action prompted the U.S. Geological Survey (USGS) to design a series of studies to evaluate the

environmental impacts of uranium mining in this area. This investigation is in response to that action. Habitats in the Grand Canyon and its environs support diverse flora and fauna that include culturally significant, threatened, and endangered species. Mining activity can result in changes to this habitat that may increase exposure of the biological resources to chemical elements including uranium, thallium, radium, and other radioactive decay products. This chapter will identify biological pathways of exposure for these radionuclides and consolidate information from the scientific literature on concentrations of these chemical elements known to adversely affect biological resources.

The goal of this chapter is to compile available toxicological and radiological information necessary to evaluate the potential effects of uranium and associated compounds on biological resources and affected habitat in the three segregation areas in northern Arizona. This chapter addresses chemical and radiation effects associated with radionuclides in the uranium-238 (^{238}U) decay series including uranium, thallium, thorium, bismuth, radium, radon, protactinium, polonium, actinium, and francium. Specific objectives are:

- to identify possible routes of exposure linked to atmospheric dispersion (including wind-borne dusts) and aqueous (surface water and groundwater), soil, sediment, and food-chain pathways;
- to identify species and habitats vulnerable to chemical and radiation effects potentially associated with uranium decay series products that are linked to exposures that result from uranium mining activities; and
- to compile relevant scientific information on toxicity threshold effects levels for uranium and associated radionuclides for aquatic and terrestrial flora and fauna.

Our approach included three interrelated steps. First, a literature search and compilation was conducted to provide a foundation for identifying the underlying biological and ecological issues that should be considered related to mining activities near the segregation areas. The second step was to characterize natural sources of uranium, other elements in the uranium decay series, and radiation (alpha, beta, or gamma) released during the uranium decay process. Because biological resources may be exposed to chemical and physical hazards associated with mining activities, the third step was the identification of ecological receptors in the area and the associated pathways of chemical and radiation exposure. Available data on culturally significant, threatened, and endangered species and associated habitats were compiled. These data were then used to develop a preliminary conceptual model.

Literature Compilation

The literature compilation was directed to collect existing chemical toxicity and radiation effects data associated with chemicals of potential concern, primarily uranium, thallium, radium, and other elements in the uranium decay series, commonly characterized in uranium ores of breccia pipe formations typical of the segregation areas. Nested literature searches and acquisitions of existing peer-reviewed data and literature were completed through:

- Cambridge Scientific Abstracts (Aquatic Sciences and Fisheries Abstracts, Biological Sciences, Environmental Sciences and Pollution Management databases; to a lesser extent, Aqualine, Water Resources Abstracts, GeoRef, Biology Digest, Conference Papers Index, Medline, and Toxline databases);
- OCLC FirstSearch (Agricola, ArticleFirst, BasicBiosis, Dissertations, GeoBase, WorldCat, BioAgIndex, Electronic Collections Online, PapersFirst, and Proceedings databases);
- ECOTOX database (in particular, combined AQUIRE and TERRATOX databases), for chemical toxicity values to characterize acute and chronic threshold effects;
- Frederica Radiation Effects Database (FREDERICA, previously known as FRED; *http://87.84.223.229/fred/mainpage.asp*, accessed October 30, 2009) to confirm published effects-based radiation exposure values;
- PubMed-NLM (U.S. National Library of Medicine), Synergy, and ScienceDirect databases; and
- Focused manual searches reliant on networked citations derived from data and literature sources encountered in directed computer-aided data mining searches.

Searches were primarily conducted for chemical name, but additional searches for species of concern used scientific name (at genus or species level) and common names. Additional search terms were added depending on the number of citations found. Terms also included keywords related to the distribution of the species, its life history and habitat, and its interaction with other species.

Sources of Uranium in Northern Arizona

The uranium province situated in the Colorado Plateau is typically characterized as being semiarid and sparsely vegetated and having a terrain of broad plateaus, ancient volcanic mountains, and deeply dissected canyons. The region contains substantial amounts of oil, gas, coal, oil shale, and uranium resources. In general, the uranium provinces throughout the western United States occur across a wide range of physiographic and ecological regions (fig. 1). Within these provinces, uranium ores occur in various geologic deposits. The spatial distribution of uranium reserves serves as a frame of reference for identifying sources of chemical and radiation hazards potentially released to the environment as a result of mining activities. Uranium in the segregation areas (fig. 2) occurs in collapse-breccia pipe deposits as an Orphan Lode–type deposit, with clusters of pipes found in proximity to one another (Weinrich, 1985; Finch, 1992). The deposits occur

as uraninite (uranium oxide) and associated sulfide, arsenide, sulfate, and arsenic-sulfosalt minerals as disseminated replacements and minor fracture fillings in near-vertical cylindrical solution-collapse breccia pipes (fig. 3). Economically recoverable quantities of copper, gold, molybdenum, nickel, silver, thorium, and vanadium can also occur with the uranium deposits. As a naturally occurring source of uranium, the pipe materials leach uranium with subsequent enrichment of copper and vanadium, among other metals, particularly in those pipes that have been deeply weathered. A massive sulfide cap prevents oxidation if there is no erosion or mining; therefore, deposits of antimony, arsenic, barium, cadmium, cobalt, chromium, cesium, copper, mercury, molybdenum, nickel, lead, selenium, silver, strontium, uranium, vanadium, yttrium, zinc, zirconium, and rare earth elements are present in the pipes (Weinrich, 1985).

Uranium can be mined by open pit, underground tunnels, and solution. Methods used depend on the type of deposit being mined, environmental conditions associated with the mining site (for example, depth to groundwater), and the economics of the mining operations (Hartman and Mutmansky, 2002; Moon and others, 2006). Heap leach mining has been phased out in the United States in favor of alternatives that reduce environmental impacts (*http://www.epa.gov/radtown/uranium-mines.html*, accessed October, 30 2009), such as in situ leaching (ISL) and in situ recovery (ISR), which have been used increasingly in mining operations throughout the world (Bartlett, 1998). However, ISL is not used for mining operations associated with breccia pipes because of their porous nature; fluids from this method have the potential to leak out of the pipe and contaminate nearby water sources (for example, aquifers and springs). Accordingly, current proposals do not include ISL or ISR in the segregation areas in northern Arizona. In addition, onsite milling of the uranium in the segregation areas is not anticipated as established uranium mills located nearby in Utah have historically been used. Nevertheless, technical literature focused on geological and mining engineering should be reviewed for a detailed analysis of extraction and recovery of uranium ores from the segregation areas. The chemicals released to the surrounding environment during uranium mining operations can only be evaluated based on an understanding of which of the mining techniques are to be used.

Conventional mining methods employed in the uranium industry and used previously in the segregation areas are associated with hazardous chemical and physical effects on the surrounding ecosystem. Deposits occurring at greater depths require underground extraction—rock is crushed to fist-sized pieces underground, which creates substantial amounts of particulates (friable sand and silt sized materials), and is brought to the surface by vertical shaft mines. Mined materials are then transported to milling operations for extraction and recovery. Uranium ore grades range from tenths of a percent to single-digit percentages, indicating that the ratio of usable uranium to mined rock is low. Therefore, conventional mining creates relatively large quantities of waste materials characterized by low-level radiation, heavy metals, and other inorganic and organic materials, which are potential sources of chemical and radiation exposure to biota.

Chemical and radiation hazards caused by source materials and waste products of uranium mining must be considered when conducting biological assessments. The number of chemical and radiation hazards potentially released to the environment can be identified and characterized with respect to their potential links to adverse effects to biota. For example, deposition of particulates above ground on soils, plants, and surface water, as well as runoff or erosion to surface water, create pathways to expose a variety of biota to uranium and its decay products. In addition, the physical habitat of biological receptors may be affected by mining activities—for example, mine shafts can become habitats for bats and birds, plant communities can be destroyed by road and building construction, and water sources for amphibians, birds, and mammals can be contaminated.

Chemical Speciation of Uranium as It Relates to Biota

Uranium naturally occurs as a major constituent in more than 150 different minerals and is a minor constituent in at least 50 other minerals (Burns and Finch, 1999). Naturally occurring uranium consists of three isotopes—^{238}U, ^{235}U, and ^{234}U—with each isotope characteristically contributing to total uranium within an isotope-specific range: ^{238}U dominates naturally occurring sources of uranium (99.28 percent expressed with regard to the mass versus 49 percent by radioactivity for natural uranium), followed by ^{235}U (0.72 percent by mass versus 2 percent by radioactivity for natural uranium) and ^{234}U (0.0050 percent by mass versus 49 percent by radioactivity for natural uranium). All three uranium isotopes yield decay products that, along with parent uranium and radiation releases typical of the decay series, present chemical hazards, radioactive hazards, or both to biological receptors. Geochemical properties of these isotopes vary among source materials. The ^{238}U decay series will be considered the primary source of chemical and radiological hazards, considering the isotope composition of naturally occurring uranium (>99 percent by mass). Constituents in the ^{238}U decay series include very short-lived elements, with half-lives on the order of minutes to days, and long-lived elements, with half-lives up to 4×10^9 years. The short-lived radionuclides are considered to have a limited potential for mediating adverse biological effects linked to chemical exposure because of reduced duration of exposure (table 1, appendix 1). Literature evaluating the chemical toxicity of these elements is limited but does indicate that exposure to these radionuclides can affect the survival, growth, reproduction, and renal function of biota. Radiation emitted during uranium decay contributes to radioactivity encountered in the environment. Therefore, radiation hazards of all daughter products are considered in evaluating radiation exposure.

Figure 1. Uranium provinces in the western United States (Finch, 1996).

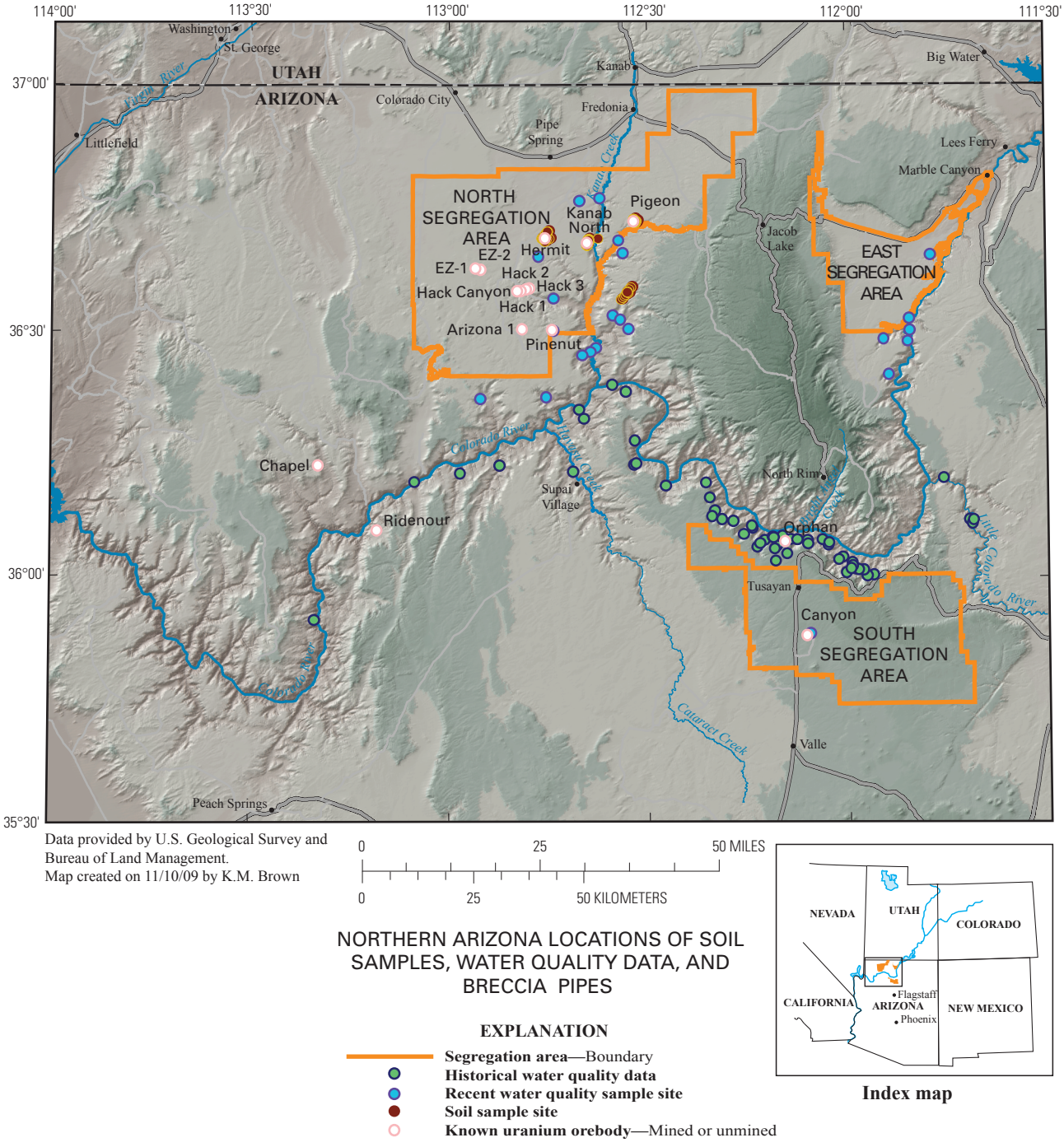

Figure 2. Segregation areas, mine sites, and sample collection locations.

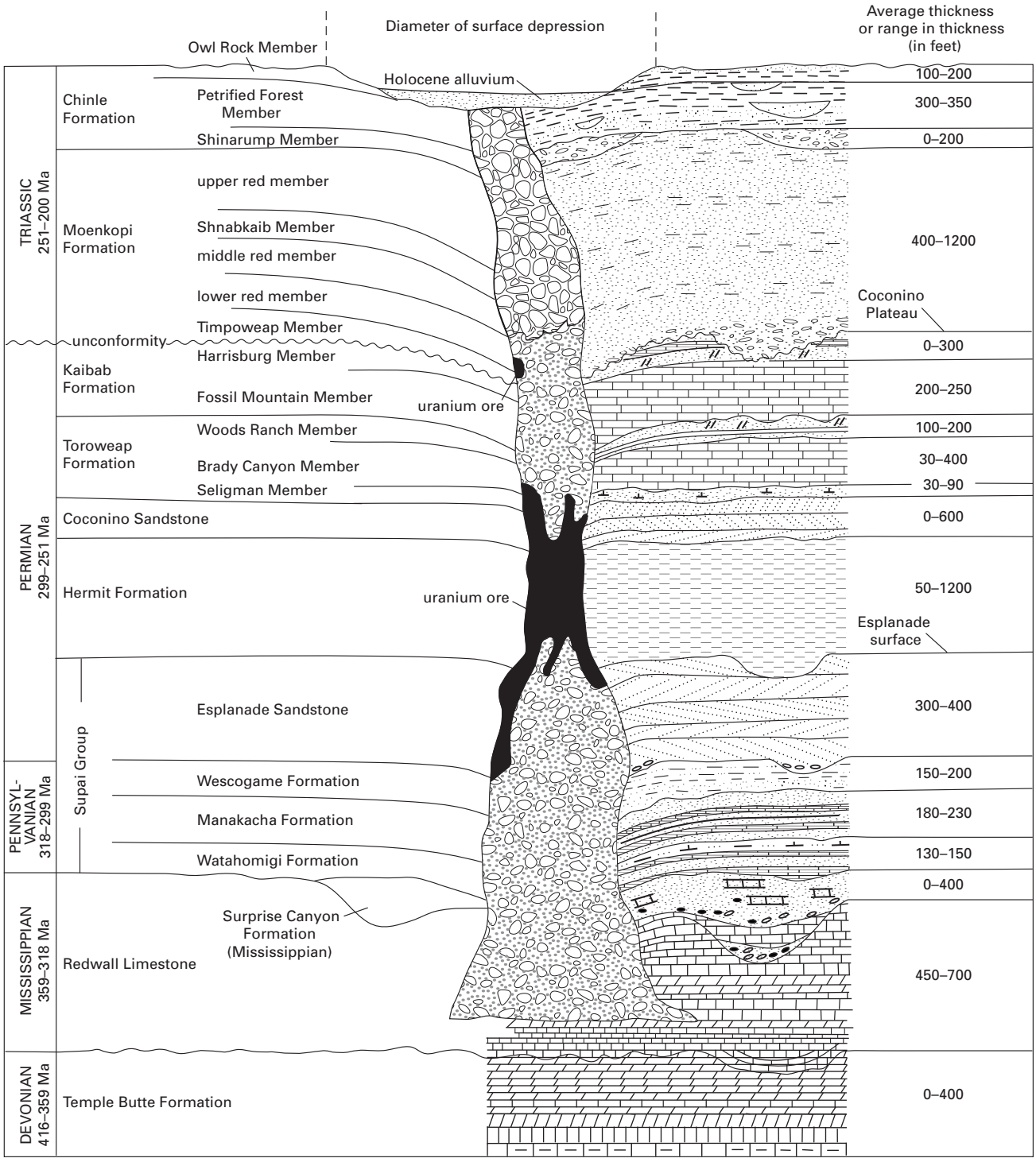

Figure 3. Schematic cross section of a solution-collapse breccia pipe in northern Arizona showing the general distribution of uranium ore within the pipe.

Table 1. Radiation and half-life for constituent elements of ^{238}U decay series.

Element	Type of radiation released in daughter formation	Half-life	Toxicity hazard
Uranium-238	Alpha	4.5×10^{9} years	Chemical, radiation
Thorium-234	Beta	24.5 days	Chemical, radiation
Protactinium-234	Beta	1.14 minutes	Radiation
Uranium-234	Alpha	2.33×10^{5} years	Chemical, radiation
Thorium-230	Alpha	8.3×10^{4} years	Chemical, radiation
Radium-226	Alpha	1,590 years	Chemical, radiation
Radon-222	Alpha	3.825 days	Chemical, radiation
Polonium-218	Alpha	3.05 minutes	Radiation
Lead-214	Beta	26.8 minutes	Radiation
Bismuth-214	Beta	19.7 minutes	Radiation
Polonium-214	Alpha	1.5×10^{-4} seconds	Radiation
Lead-210	Beta	22 years	Chemical, radiation
Bismuth-210	Beta	5 days	Chemical, radiation
Polonium-210	Alpha	140 days	Chemical, radiation
Lead-206	Stable	Stable	Chemical

Within the context of radiation exposures to ^{238}U decay products (including ^{234}U) in the field, an initial data filter was applied to distinguish between constituents that present significant chemical hazards relative to radiation hazards and those that do not. Radionuclides with half-lives of less than one day were removed from consideration as a chemical hazard; their short lives prevent them from participating in pathways or in reaching significant concentrations to pose a chemical risk. However, they can significantly contribute to radiological doses and are considered from that perspective. These radionuclides, or chemicals of potential ecological concern (COPECs), were sorted to expedite compilation of ecotoxicity data for uranium exposures in the field. Those radionuclides with a half-life of less than one day include ^{214}Bi (bismuth), ^{214}Pb (lead), ^{214}Po (polonium), ^{218}Po, and ^{234}Pa (protactinium). Their radiation hazard is largely a function of their alpha particle or beta particle emissions in the ^{238}U decay series. In contrast, those longer-lived radionuclides (half-life greater than or equal to one day) were considered for both radiation and chemical hazards. These COPECs include ^{230}Th (thorium), ^{234}Th, ^{226}Ra (radium),^{222}Rn (radon), ^{210}Bi, ^{210}Po, ^{210}Pb, ^{234}U, and ^{238}U. A stable isotope of lead (^{206}Pb) is the end-state of the ^{238}U decay series.

The bioavailability of metals (radionuclides) in water, sediment, and soil must be characterized to estimate effects on biota associated with exposure to uranium concentrations that result from natural weathering of mineral deposits or from mining activities. Bioavailability is the fractional uptake of metals (radionuclides) within the tissues of biota and is considered when examining the incorporation of metals (radionuclides) into biotic tissues. It is very context-specific to the metal (radionuclide), environmental conditions, and the biota of interest. Bioavailability alters the assimilation efficiencies of metals (radionuclides) by biota and therefore can reduce the effective exposure and resultant dose. Uptake of metals (radionuclides) may occur directly through water exposure (aquatic organisms), soil solution exposure (terrestrial plants or soil fauna), dermal or foliar contamination, inhalation (terrestrial biota), or through intentional or coincidental dietary ingestion of water, food, soils, or sediments.

Chemical speciation of uranium has been summarized previously (Gunther and others, 2002; Markich, 2002; Salbu and Skipperud, 2009). In general, speciation of uranium influences the chemical's transport (or mobility) within specific environments, as well as the bioavailability in aquatic and terrestrial systems. The metal's chemical toxicity will vary depending on the matrix in which it occurs. The speciation of uranium in aquatic systems (fresh waters and sediments) can be characterized by potential oxidation states. Uranium(VI) is the major form of uranium in oxic surface waters, whereas U(IV) is the major form in anoxic waters. The relationship between uranium speciation and bioavailability is complex and incompletely understood (Gunther and others, 2002; Markich, 2002). Uranium in surface waters occurs as various physicochemical forms depending on the environmental conditions. Uranium complexes with organic ligands, inorganic complexes of U(VI) as phosphate or humic substances, or metal-bound particulates or colloidals yield lower bioavailability by reducing activities of $UO_2^{(2+)}$ and UO_2OH^+. The uranyl ion ($UO_2^{(2+)}$) and the uranyl hydroxyl complex UO_2OH^+ are the major forms of U(VI) available to organisms. In contrast to other metals, characterization of uranium bioavailability is highly dependent on geochemical speciation models, and data from empirical studies using natural waters and aquatic biota are limited. Data are not available to characterize uranium speciation and its bioavailability in sediments.

Uranium in rocks and minerals generally occurs in low concentrations or grade in terrestrial systems, although uranium distribution in sedimentary rocks varies widely with high uranium contents in black shales, phosphate rocks, and coal. The natural background uranium concentration is 2.3 mg/kg for soils in the segregation areas (Smith, 1997). Uranium may be a

major element in a mineral (for example, uraninite, UO_{2+}) or an accessory mineral (for example, uranothorite, $(Th,U)SiO_4$). The redox state of uranium, in particular the ratio of U(VI) to U(IV), governs its solubility and subsequent movements in undisturbed materials and its release to the environment through various anthropogenic activities (Hem, 1992; Roh and others, 2000; Zhang and Brady, 2002; Stewart, 2008). The biogeochemistry of uranium under field conditions is incompletely characterized with respect to its physicochemical behavior in complex chemical mixtures with dissolved metals, organic ligands, and mineralogical matrices (Alloway, 1990; Hem, 1992; Zhang and Brady, 2002; Cooper and others, 2003; Stewart, 2008). The environmental fate and movement of uranium in near-surface and subsurface environments is strongly influenced by oxygen. For example, UO_2 displays decreased solubility and movement in soil under anaerobic conditions and increased solubility and movement in aerobic soils. The physicochemical behavior of uranium affects adsorption and desorption processes that occur over a wide range of mineral substrates and soil types and influence concentrations in soil solution. These surface-mediated processes are strongly influenced by the geochemical composition of soils. For example, soil calcium promotes the formation of ternary uranyl-calcium-carbonato complexes, which decreases the extent and rate of U(VI) reduction and therefore reduces the amount of U(VI) adsorption to mineral surfaces (Stewart, 2008).

The bioavailability of uranium depends on its speciation in the environment. Metals including uranium partition between solid and liquid phases and may occur as dissolved, exchangeable, carbonate, iron-manganese oxide, organic, or crystalline species. Partitioning or speciation is influenced to varying degrees by pH, redox state, organic content, and other environmental factors such as temperature, flow rates, and periodic events (such as storms). Hydrogen ion activity (pH) is likely one of the more critical factors governing metal speciation, solubility from mineral surfaces, transport, and eventually bioavailability (Zhang and Brady, 2002). Particulate size and total surface area available for adsorption affect metal speciation and metal bioavailability. For example, finely milled ore may release smaller particles that are likely more widely dispersed by water and wind and enhance metal adsorption (Jones and others, 1990; Hem, 1992; Zhang and Brady, 2002).

Radioisotopes of interest in uranium mill tailings include ^{230}Th, ^{226}Ra, ^{222}Rn, and its daughter products. With its long half-life, ^{230}Th is the parent and a constant source of ^{226}Ra (see Esienbud and Gesell, 1997; Burns and Finch, 1999). Thorium and radium migrate slowly in the soil but can move via groundwater into sediments and surface waters. Radium isotopes ^{226}Ra (from the ^{238}U decay chain) and ^{224}Ra (from the ^{232}Th decay chain) are chemically similar to calcium. Radium is assimilated from the soil by plants and passed up the food chain to terrestrial biota including humans. In vertebrates, radium can enter the body through ingestion or inhalation pathways. The radium isotopes are alpha emitters and the parents of radon gas (^{226}Ra decays to ^{222}Rn; ^{224}Ra decays to ^{220}Rn). All radon isotopes are noble gases and inert with relatively short half-lives. Radon progeny are electrically charged when formed and attach to naturally occurring dust particles within the atmosphere. The inhalation of minute dust particles laden with radon progeny is a major contributor to the annual dose of natural radioactivity received by humans (National Council on Radiation Protection and Measurements, 1987). Radiation dose associated with inhalation of larger sized particulates carrying radon particles is less important because larger particulates are less likely to enter vertebrate lungs. Inhalation of aerosols and particulates by miners working in the confined spaces of the uranium mines was critical to occupational exposures; however, in natural settings the release of the radionuclides is often dispersed into the atmosphere and diluted, thus decreasing the radiogenic effects on surrounding biota (International Atomic Energy Agency, 1997).

Biological Receptors for Exposure Effects Associated with Uranium Mining

Habitats in northern Arizona support diverse flora and fauna that include culturally significant, threatened, and endangered species. The wide range of elevation and slope aspect creates a variety of habitats including desert scrub, ponderosa and pinyon pine forests (*Pinus* spp.), and seeps and springs in which species can thrive. A diagrammatic sketch of the system at risk was developed to identify potential linkages between chemical and radiation hazards associated with mining operations and biota present in the segregation areas (fig. 4). Biological receptors co-occur with environmental hazards that are associated with proposed uranium mining activities. Direct effects associated with radiochemicals and associated radiation released in the decay process are the primary focus of this chapter, but inorganic chemicals, physical and biological stressors, and indirect effects also warrant attention. These potentially confounding factors likely operate at landscape levels (such as ecoregion and watershed) and should be considered as contributing factors in an ecological risk assessment.

Biota of concern, based on the food web, were identified as soil microorganisms (including soil crust and microbial communities), aquatic microorganisms, terrestrial and aquatic vascular plants, terrestrial and aquatic invertebrates, fish, amphibians and reptiles, birds, and mammals. Threatened and endangered species including aquatic invertebrates, birds, fishes, and terrestrial plants occur within or near the segregation areas (table 2, figs. 5–7).[1] Distribution maps are

[1]The species distribution map for Kanab ambersnail (*Oxyloma haydeni kanabensia*) does not include a translocated population at Elves Chasm, at River Mile 117 (Sorenson and Nelson, 2000).

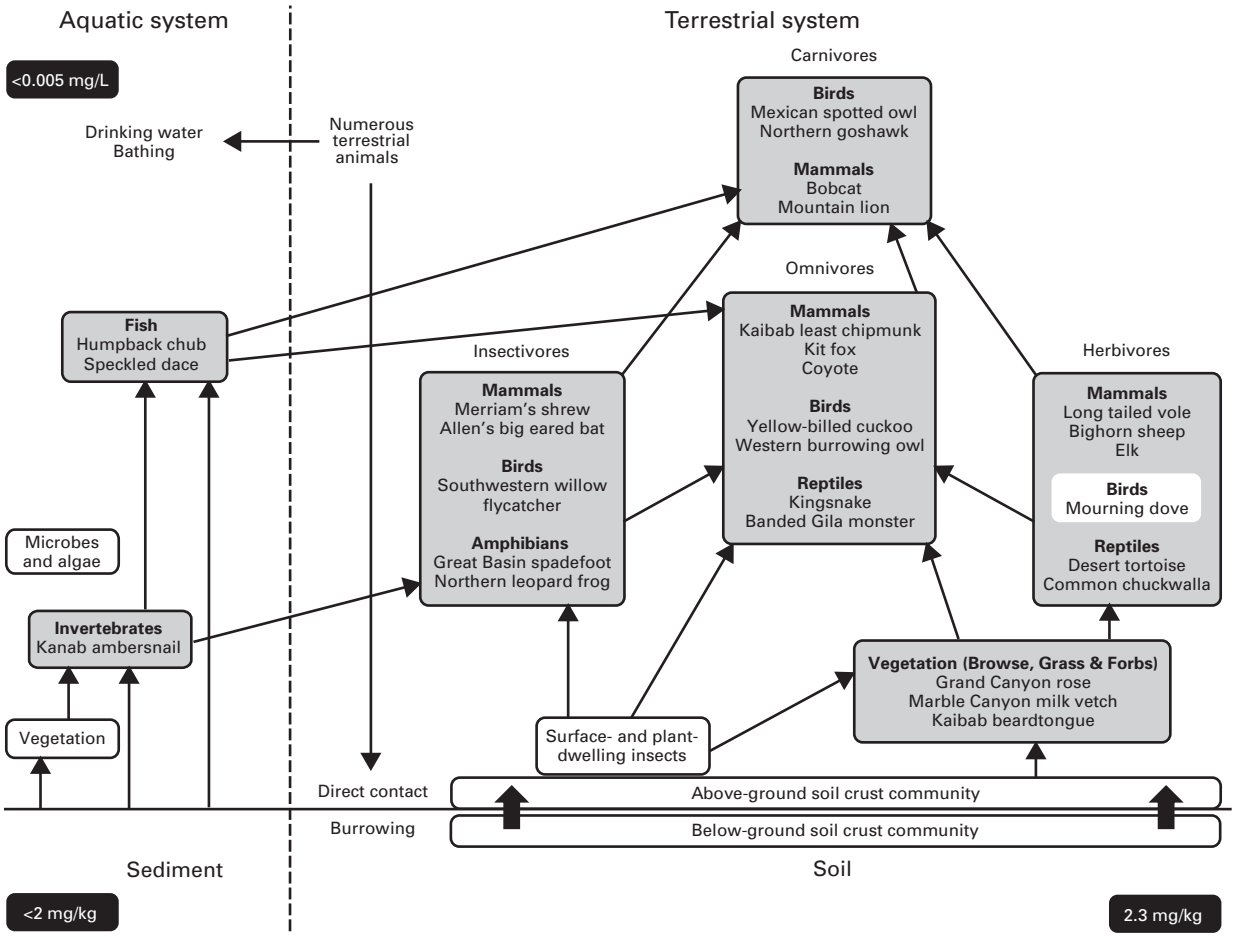

Figure 4. Diagrammatic representation of a generalized food web for the segregation areas. Gray shading indicates receptor groups for which sensitive species have been identified by State and Federal agencies (see tables 2 and 3 for species-specific information). Example species for each receptor group are provided. Black boxes are regional background concentrations of uranium in water, sediment, and soil.

not available for all species, but documentation from State and Federal agencies indicate that appropriate habitat for these species is available within or near the segregation areas (table 2). Species of concern (as identified by State and Federal agencies) occur in most compartments of the food web within the segregation areas (table 3, fig. 4).

Uranium and other radionuclides can be transported through the environment and contribute to exposure of biological receptors via atmospheric deposition, dust, runoff, erosion and deposition, groundwater and surface water, and the food chain. As a result, biological receptors can be exposed to radionuclides through various pathways including ingestion (soil, food, or water), inhalation, cell membrane–mediated uptake, cutaneous absorption, and biotic uptake/trophic transfer (table 4, fig. 8). Chemical and radiological exposure of burrowing or subterranean invertebrates such as tiger beetles and desert centipedes, amphibians such as the Great Basin spadefoot (*Spea intermontana*), reptiles such as the northern sagebrush lizard (*Sceloporus graciosus graciosus*) and common kingsnake (*Lampropeltis getulus*), mammals such as House Rock Valley chisel-toothed kangaroo rat (*Dipodomys microps leucotis*), Kaibab northern pocket gopher (*Thomomys talpoides kaibabensis*), and kit fox (*Vulpes macrotis*), and birds such as the western burrowing owl (*Athene cunicularia hypugea*) are of particular concern. In addition, a variety of species of scorpions, birds, and bats use cave-like mine shafts for habitat. For example, several bat species listed as species of concern—Allen's big-eared bat (*Idionycteris phyllotis*), Pale Townsend big-eared bat (*Corynorhinus townsendii*), fringed myotis (*Myotis thysanodes*), long-eared myotis (*M. evotis*), and western small-footed myotis (*M. ciliolabrum*)—hibernate in mine shafts where these animals are exposed to prolonged radiation and chemical hazards associated with uranium mining (table 3). Herbivores listed as species of concern, such as the desert tortoise (*Gopherus agassizii*), common chuckwalla (*Sauromalus obesus*), Kaibab northern pocket gopher, Navajo Mexican vole (*Microtus mexicanus navaho*), elk (*Cervus elaphus*), and bighorn sheep (*Ovis canadensis*), can be exposed to radionuclides through the aerial deposition of uranium or its decay products onto vegetation.

Table 2. Threatened and endangered species occurring within the segregation areas. Habitat data from the State of Arizona's Natural Heritage Program–Heritage Data Management System (*http://www.azgfd.gov/w_c/edits/species_concern.shtml*, accessed Octber 2009).

[NA, not applicable]

Name	Status	Habitat
	Amphibians	
None	NA	NA
	Aquatic vascular plants	
None	NA	NA
	Aquatic invertebrates	
Kanab ambersnail (*Oxyloma haydeni kanabensis*)	Endangered	Marshes watered by springs and seeps at base of sandstone cliffs or limestone in Vaseys Paradise of the Grand Canyon.
	Birds	
Mexican spotted owl (*Strix occidentalis lucida*)	Threatened	Patchy distribution along steep canyons of Grand Canyon in Coconino and Mohave Counties.
Southwestern willow flycatcher (*Empidonax traillii extimus*)	Endangered	Riparian obligate that avoids riparian areas in steep, closed canyons. Breeds locally along Colorado River in Grand Canyon near mouth of Little Colorado River.
California condor (*Gymnogyps californianus*)	Endangered	Wide distribution. Reintroduced to Vermilion Cliffs and Hurricane Cliffs in Arizona.
Yuma clapper rail (*Rallus longirostris yumanensis*)	Endangered	Marsh areas along the Colorado River.
American peregrine falcon (*Falco peregrinus anatum*)	Delisted	Wide distribution in Arizona. Most of Arizona's breeding peregrines are found near Mogollon Rim, Grand Canyon, and Colorado Plateau.
Bald eagle (*Haliaeetus leucocephalus*)	Delisted	Wintering populations found in areas of northern Arizona with open water.
	Fish	
Humpback chub (*Gila cypha*)	Endangered	Turbulent, high gradient canyon-bound reaches of Colorado and Little Colorado Rivers in the Grand Canyon.
Razorback sucker (*Xyrauchen texanus*)	Endangered	Variety of habitats in Colorado River.
	Mammals	
Black-footed ferret (*Mustela nigripes*)	Endangered	Arid prairies with prairie dogs. Reintroduced in Aubrey Valley of Coconino County.
	Reptiles	
None	NA	NA
	Terrestrial invertebrates	
None	NA	NA
	Terrestrial plants	
Sentry Canyon milk-vetch (*Astragalus cremnophlax* var. *cremnophlax*)	Endangered	Kaibab limestone on North and South Rims of the Grand Canyon in Coconino County.
Gierisch globemallow (*Sphaeralcea gierischii*)[1]	Candidate	Gypsum outcrops on Kaibab limestone in Pigeon Canyon, Black Knolls, and Black Rock Gulch in Mohave County.
Jones cycladenia (*Cycladenia humilis* var. *jonesii*)	Threatened	Gypsum soils on clay hills that form steep side slopes and bases of canyons in Vermilion Cliffs and Moccasin Mountains, Arizona.
Silver pincushion cactus (*Pediocactus sileri*)	Threatened	Red or gray gypsum badlands from Moenkopi Formation in Mohave County from Hurricane Cliffs to Pipe Spring, Coconino County.
Fickeisen plains cactus (*Pediocactus peeblesianus* var. *fickeiseniae*)	Candidate	Kaibab limestone in House Rock Valley and Gray Mountain in Coconino County and Hurricane and Main Street Valleys and near Clayhole and Sunshine Ridge in Mohave County.
Paradine (Kaibab) pincushion cactus (*Pediocactus paradinei*)[1]	Conservation agreement	Level sites on alluvial fans, valley bottoms, and ridge tops on eastern slopes of Kaibab Plateau and west side of House Rock Valley.
Brady pincushion cactus (*Pediocactus bradyi*)	Endangered	Kaibab limestone alluvium on gentle slopes of Marble Canyon, Coconino County.

[1]No distribution map available.

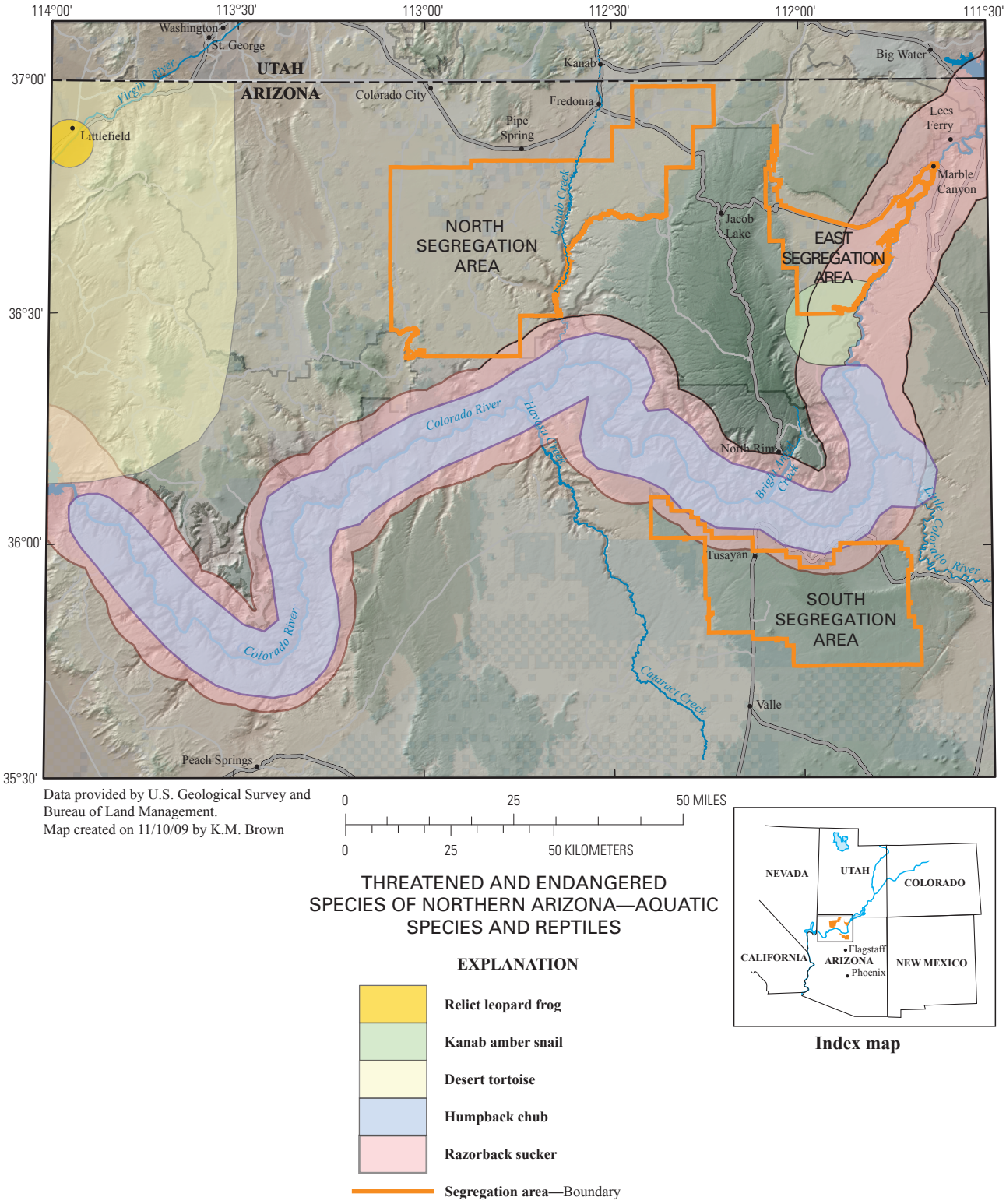

Figure 5. Range distributions of threatened and endangered fish, amphibians, and reptiles in relation to the segregation areas. The species distribution map for Kanab ambersnail does not include a translocated population at Elves Chasm, at River Mile 117 (Sorenson and Nelson, 2000). (Geospatial data provided by Mary Richardson, U.S. Fish and Wildlife Service, Phoenix, Ariz.)

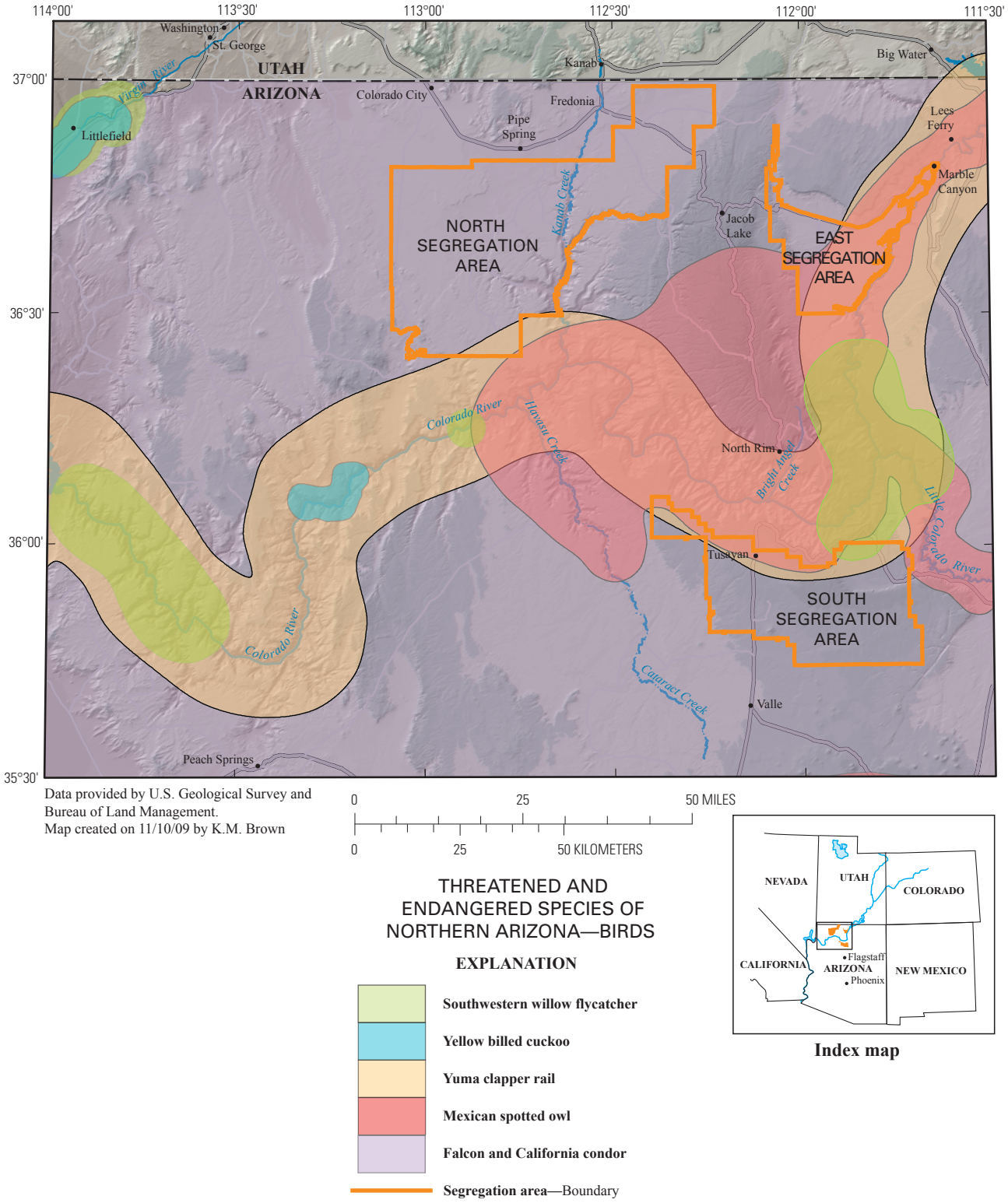

Figure 6. Range distributions of threatened and endangered birds in relation to the segregation areas. (Geospatial data provided by Mary Richardson, U.S. Fish and Wildlife Service, Phoenix, Ariz.)

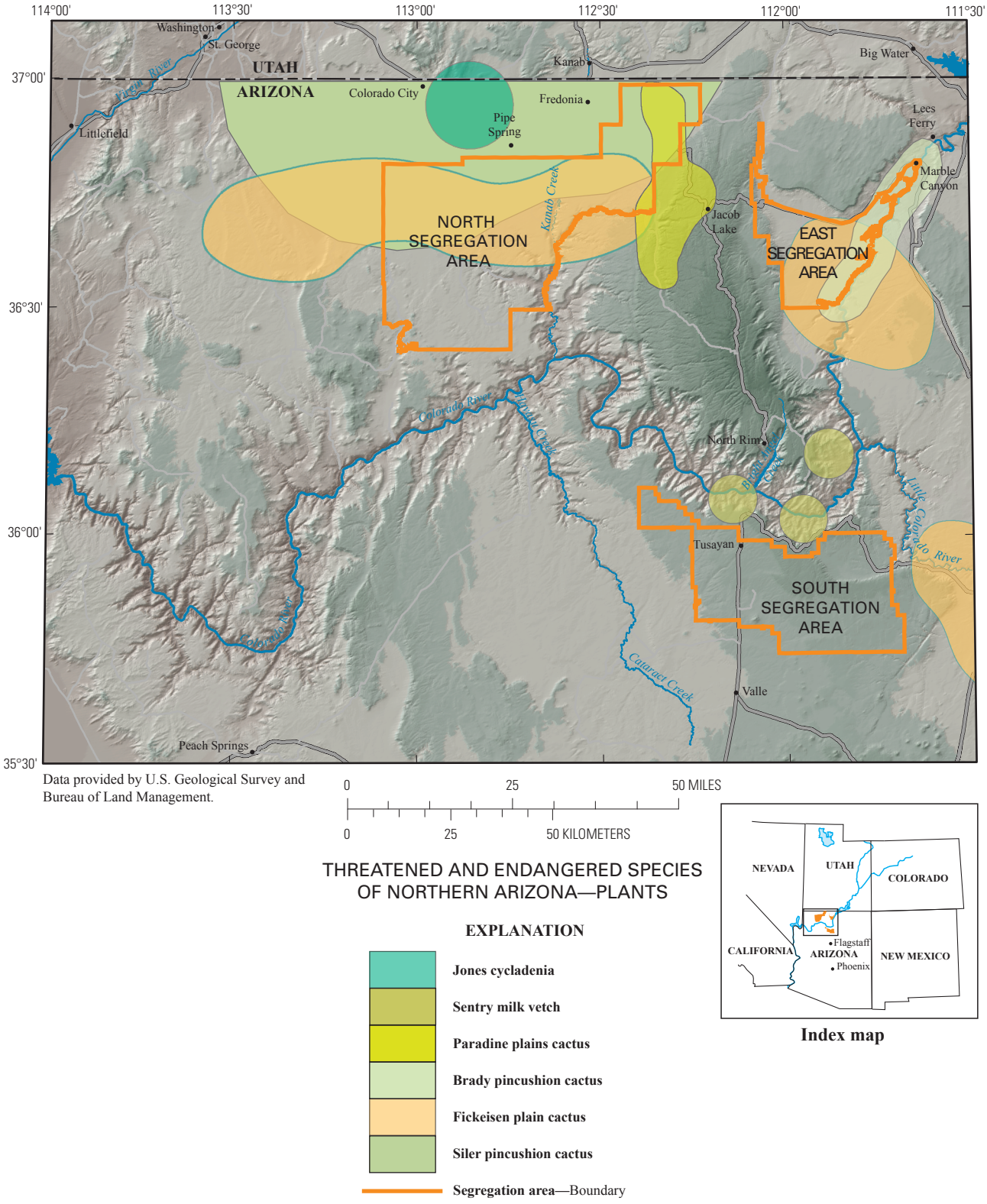

Figure 7. Range distributions of threatened and endangered plants in relation to the segregation areas. (Geospatial data provided by Mary Richardson, U.S. Fish and Wildlife Service, Phoenix, Ariz.)

Table 3. Examples of species of concern documented within the segregation areas based on data from the State of Arizona's Natural Heritage Program–Heritage Data Management System (*http://www.azgfd.gov/w_c/edits/species_concern.shtml*, accessed October 2009), the U.S. Bureau of Land Management, the U.S. Forest Service, and the National Park Service.

Name	Habitat	Diet
	Amphibians	
Great Basin spadefoot (*Spea intermontanus*)	Sagebrush flats, semi-desert shrublands, and pinyon-juniper woodland. Burrow in loose soil or use those of small mammals; emergence and surface activity associated with rainfall.	Adults are invertivores (insects, arachnids, snails). Larvae eat organic debris, plant tissue, invertebrates, and amphibian larvae.
Relict leopard frog (*Lithobates* [*Rana*] *onca*)	Springs and wetlands with open shorelines.	Adults are invertivores. Larvae eat algae, organic debris, plant tissue, and microorganisms.
Northern leopard frog (*Rana pipiens*)[2,3]	Grassland, brushland, woodland, and forest typically in permanent waters with rooted aquatic vegetation.	Adults are insectivores. Larvae eat algae, organic debris, plant tissue, and small invertebrates.
Red spotted toad (*Bufo punctatus*)	Riparian area of rocky streams and arroyos.	Insectivorous.
	Plants	
Roaring Springs prickly-poppy (*Argemone arizonica*)[3]	Steep, south-facing slopes; rockslides in pinyon juniper/desertscrub.	Not applicable.
Welsh's milkweed (*Asclepias welshii*)	Open, sparsely vegetated semi-stabilized sand dunes in desert scrub.	Not applicable.
Atwood's catseye (*Cryptantha atwoodii*)	Sandy to clayey soils with sagebrush and pinyon juniper. West rim of Marble Canyon on Kaibab limestone.	Not applicable. Note: Potentially at risk from grazing sheep, goats, and insects.
Bigelow onion (*Allium bigelovii*)	Dry rocky soil in grassland, open chaparral, and desertscrub communities.	Not applicable. Note: Species considered edible and bulbs sought after by Native Americans for food and seasoning.
Ditch evening-primrose (*Camissonia* ssp. *abyssa*)	Debris slides and crevices of broken Redwall Limestone.	Not applicable.
Grand Canyon rose (*Rosa stellata*)[1,2]	Known populations are on or near canyon rims or cliff tops at edges of mesas or plateaus, along low ledges at depressions caused by breccia pipes. Kanab Canyon: rim on low limestone breaks and in small, shallow drainages. Twin Point: on deeper soils along west edge, Kaibab limestone bedrock outcropping in places.	Not applicable. Note: Wildlife may browse on this plant, especially rabbits; grows in breccia pipes where uranium prospects have been concentrated.
Gumbo milk-vetch (*Astragalus ampullarius*	Gumbo clay knolls.	Not applicable. Note: Potentially threatened by mineral exploration and livestock grazing.
Cliff milk-vetch (*Astragalus cremnophlax* var. *myriorrhaphis*)[1,2]	Crevices and depressions with shallow or no soil on Kaibab limestone on rim-rock benches, cliff ledges, and pinnacles.	Not applicable.
Marble Canyon milk-vetch (*Astragalus cremnophlax* var. *hevronii*)[1,2]	Desertscrub on rim-rock benches on canyon edge in crevices and depressions with shallow soils on Kaibab limestone.	Not applicable.
North Rim vetch (*Astragalus septentriorema*)[3]	Not well defined.	Not applicable.
Sentry Canyon milk-vetch (*Astragalus cremnophylax*)	Kaibab limestone with mat rockspirea (*Petrophytum caespitosum*) in pinion-juniper-cliffrose plant community.	Not applicable.
Kaibab beardtongue (*Penstemon pseudoputus*)	Kaibab limestone and sandstone in grassland meadows; disturbed areas.	Not applicable. Note: Browsed upon by deer and rodents.
Flagstaff penstemon (*Penstemon nudiflorus*)[2]	Dry ponderosa pine forests.	Not applicable.
House Rock fishhook cactus (*Sclerocactus sileri*)[1]	Pinyon-juniper mesa tops in House Rock Valley and Paria Plateau.	Not applicable.
Kaibab bladderpod (*Lesquerella kaibabensis*)	Limestone-clay knolls with exposed rock; meadows of Kaibab Plateau.	Not applicable.
Kaibab paintbrush (*Castilleja kaibabensis*)	Fine silts and clays to rocky meadow soils from Kaibab limestone on low rounded ridge tops and small knolls.	Not applicable. Note: Grazing is the most significant risk.
Grand Canyon catchfly (*Silene rectiramea*)[3]	North-facing in gravel loam to clay soils in limestone and calcareous sandstone.	Not applicable.

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Name	Habitat	Diet
	Plants—Continued	
Grand Canyon suncup (*Camissionia confertiflora*)[3]	Volcanic substrates, desert shrub/scrub.	Not applicable.
Spiked ipomopsis (*Ipomopsis spicata* ssp. *tridactyla*)[3]	Not well described.	Not applicable.
Tusayan rabbitbrush (*Chrysothamnus molestus*)[2]	Pinyon-juniper grasslands on slopes and flats.	Not applicable.
Mt. Dellenbaugh sandwort (*Arenaria aberrans*)[2]	Open pine and pine-pinyon woodlands.	Not applicable.
Arizona leatherflower (*Clematis hirsutissima*)[2]	Grassland, sagebrush, ponderosa pine.	Not applicable.
Tusayan fameflower (*Phemeranthus validulus*)[3]	Bare slopes and summits in rock soil on chert, basalt, and cinder.	Not applicable.
Grand Canyon flaveria (*Flaveria mcdougallii*)[3]	Alkaline or saline seeps along ledges.	Not applicable.
Resin brittlebush (*Encelia resinifera* ssp. *tenuifolia*)[3]	Rocky hillsides, dry slopes, washes.	Not applicable.
Grand Canyon goldenweed (*Ericameria arizonica*)[3]	Rocky ledges and crack of Kaibab limestone.	Not applicable.
Mollogon columbine (*Aquilegia desertorum*)	Xerophyte on rocky slopes in the transition zone. Enormous root and long lived.	Not applicable.
Shiny-leaved sandpaper-plant (*Petalonyx nitidus*)	Open slopes and mesas; frequently on volcanic substrates including breccias.	Not applicable.
	Invertebrates	
Niobrara ambersnail (*Oxyloma haydeni haydeni*)	Seep or spring-fed wetlands.	Not defined. Note: Predators include insects, mammals, birds, and other snails.
Grand Canyon cave pseudoscorpion (*Archeolarca cavicola*)[3]	Subterranean cave habitat associated with bats and rodents.	Insectivorous—arthropods.
	Birds	
Mourning dove (*Zenaida macroura*)	Brushlands and woodlands.	Herbivorous—seeds and grains.
Western yellow-billed cuckoo (*Coccyzus americanus occidentalis*)[3]	Streamside cottonwood, willow groves, mesquite bosques.	Omnivorous—caterpillars, bird eggs, frogs, lizards, ants, beetles, berries.
Western burrowing owl (*Athene cunicularia hypugaea*)[1,2]	Open, well-drained grasslands, steppes, and deserts.	Omnivorous—invertebrates, small mammals, fish, reptiles, amphibians, birds.
Northern goshawk (*Accipiter gentilis*)[2,3]	Nest in ponderosa pine forest on Kaibab Plateau.	Carnivorous—tree squirrels, rock squirrels, cottontail rabbits, birds.
	Fish	
Flannelmouth sucker (*Catostomus latipinnis*)[1]	Large rivers.	Omnivorous—primarily invertebrates and microorganisms.
Apache trout (*Oncorhynchus apache*)	Introduced and established population in North Canyon Creek in Kaibab National Forest.	Insectivorous—aquatic and terrestrial.
Speckled dace (*Rhinichthys osculus*)[1]	Native to Colorado River system.	Omnivorous—primarily algae, crustaceans, insect larvae, small snails.
	Mammals	
Merriam's shrew (*Sorex merriami leucogenys*)[2]	Sagebrush steppe, grassland, brushland, woodland.	Insectivorous—beetles, spiders, caterpillars, crickets, wasps.
Dwarf shrew (*Sorex nanus*)[2]	Pinyon-juniper woodland.	Omnivorous—insects, spiders, small invertebrates, plant material.
Western red bat (*Lasiurus blossevillii*)[2,3]	Migratory—summer roosts in tree foliage, occasionally in saguaro boots and cave-like structures.	Insectivorous—moths, flies, beetles, cicadas, ground dwelling crickets.
Allen's big-eared bat (*Idionycteris phyllotis*)[1,2,3]	Taken most often in ponderosa pine, pinyon-juniper, Mexican woodland. Boulder piles, cliffs, rocky outcrops, or lava flows at or near most collection locations. Roosts in caves and abandoned mineshafts.	Insectivorous—soft bodied insects such as moths; beetles, roaches, ants.

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Name	Habitat	Diet
	Mammals—Continued	
Big brown bat (*Eptesicus fuscus*)	Taken most often in ponderosa pine, pinyon-juniper. Roosts in a variety of sites including attics, barns, bridge joints, hollow trees, mines, rock crevices, caves, and other similar locations.	Insectivorous—Coleoptera is important.
Big free-tailed bat (*Nyctinomops macrotis*)[1]	Inhabit rugged, rocky country and riparian areas. Roost in caves and holes in trees.	Insectivorous—moths, crickets, grasshoppers, flying ants, stinkbugs, leafhoppers.
California myotis (*Myotis californicus*)	Roost in crevices and cracks in canyon walls, under loose bark or in old snags, sometimes in caves and mine shafts.	Insectivorous—moths, flues, beetles, bugs.
Fringed myotis (*Myotis thysanodes*)[1]	Oak-pinyon woodlands and other open, coniferous, middle-elevation forests. Roost sites have been found in caves, mine tunnels, in large snags, under exfoliating bark, and in buildings. May use lower elevation caves and mines as hibernation sites.	Insectivorous—beetles, moths.
Greater western mastiff bat (*Eumops perotis californicus*)[3]	Desert scrub cliffs; rugged rocky canyons with abundant crevices.	Insectivorous—moths, crickets, grasshoppers, dragonflies, beetles, bees, wasps, ants.
Hoary bat (*Lasiurus cinereus*)	Roost in foliage of deciduous and coniferous trees.	Insectivorous—moths.
Long-eared myotis (*Myotis evotis*)[1]	Inhabit ponderosa pine or spruce-fir forests of Arizona. Summer roosts in rock outcroppings, tree cavities, under peeling bark, in stumps, caves, mines, sink holes, lava tubes, or in abandoned buildings. Likely use caves and abandoned mines during hibernation.	Insectivorous—primarily Lepidopterans.
Long-legged myotis (*Myotis volans*)[1,3]	Primarily coniferous forest; also riparian and desert habitats. Roosts including cracks in the ground, crevices in cliff faces, and spaces behind exfoliating tree bark. Use caves and mine tunnels for hibernation.	Insectivorous—flies, termites, lacewings, wasps, beetles.
Mexican free-tailed bat (*Tadarida brasiliensis*)	Migratory—roost in caves, mine tunnels, and crevices.	Insectivorous—primarily moths.
Pale Townsend's big-eared bat (*Corynorhinsus townsendii pallescens*)[2,3]	Summer roosts include caves and mines from desertscrub to coniferous forests. Winter hibernation in cold caves, lava tubes, and mines in vicinity of Grand Canyon.	Insectivorous—primarily moths.
Silver-haired bat (*Lasioncteris noctivagans*)	Broad-leafed riparian and coniferous woodlands near water.	Insectivorous—Tricoptera and Coleoptera.
Spotted bat (*Euderma maculatum*)[2,3]	Dry, rough desertscrub.	Insectivorous—primarily moths.
Western small-footed myotis (*Myotis ciliolabrum*)[1]	Deserts, chaparral, riparian areas, and oak-juniper forests. Hibernates in caves and old mines; summers in crevices, cracks, holes, snags, hollow trees, under rocks, and in buildings.	Insectivorous—flying insects.
Kaibab least chipmunk (*Neotamias minimus consobrinus*)[2]	Rock cliffs, river bluffs, and forest edges.	Omnivorous—plants, fungi, invertebrates, small mammals and birds.
Kaibab northern pocket gopher (*Thomomys talpoides kaibabensis*)[2]	Live underground (fossorial) in sagebrush steppe and valley grasslands.	Herbivorous—roots and stems of forbs and herbs.
House Rock Valley chisel-toothed kangaroo rat (*Dipodomys microps leucotis*)[2]	Burrow in desertscrub communities with high shrub cover and sparse grass cover.	Herbivorous—saltbush leaves.
Navajo Mexican vole (*Microtus mexicanus navaho*)[3]	Burrow in dense shrub thickets; dry grassy areas adjacent to ponderosa pine forests.	Herbivorous—grasses, forbs.
Navajo Mogollon vole (*Microtus mogollonensis navaho*)[2]	Burrow in dense shrub patches in ponderosa pine forests, sagebrush stands, thick grasses.	Herbivorous—grasses, forbs, other vegetation.
Long-tailed vole (*Microtus longicaudus*)[2]	Scrubby and grassy meadows; high elevations (>2,400 m).	Herbivorous—fruit, seeds, bark, leaves.

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Name	Habitat	Diet
	Mammals—Continued	
Kit fox (*Vulpes macrotis*)	Sandy areas; spends day underground.	Omnivorous—rodents, rabbits, birds, snakes, insects, seeds, berries.
Mule deer (*Odocoileus hemionus*)	Desert shrub, grasslands, pinyon-juniper, pine, aspen-fir, and mountain meadows.	Herbivorous—mountain-mahogany, buckbrush, cliffrose, sagebrush, buckthorn, juniper, and oak.
Elk (*Cervus elaphus*)[3]	Fir-aspen and pine-juniper forests.	Herbivorous—weeds, grasses, sedges, shrubs, willows, trees.
Bighorn sheep (*Ovis canadensis canadensis*)[2,3]	Mountain ledges and grassy basins.	Herbivorous—grass, sage, sedges.
Bobcat (*Lynx rufus*)	Ubiquitous in Arizona.	Carnivorous—cottontail rabbits, jackrabbits, mice, rats, birds.
Coyote (*Canis latrans*)	Ubiquitous in Arizona.	Omnivorous—small mammals, carrion, bird eggs, plants (juniper and Manzanita berries).
Mountain lion (*Puma concolor*)[3]	Desert mountains with broken terrain and steep slopes.	Carnivorous—mule deer, whitetail deer, javelina, livestock.
	Reptiles	
Gopher snake (*Pituophis melanoleucus*)	Ubiquitous in Arizona.	Omnivorous—small mammals, birds, bird eggs.
Common kingsnake (*Lampropeltis getulus*)	Ubiquitous in Arizona.	Omnivorous—lizards, birds, mammals, frogs, bird eggs, snakes, large invertebrates.
Northern sagebrush lizard (*Sceloporus garciosus graciosus*)[1]	Generally ground-dweller near bushes, brush heaps, logs, or rocks. Sagebrush, Manzanita, pinyon-juniper woodlands, pine and fir forests of canyon bottoms.	Insectivorous—wide variety of arthropods.
Desert spiny lizard (*Sceloporus magister*)	Desertscrub and thornscrub.	Insectivorous.
Desert tortoise (*Gopherus agassizii*)[3]	Rocky soils; desert scrub.	Herbivorous—grasses, forbs, succulents.
Banded Gila monster (*Heloderma suspectum cinctum*)[1]	Undulating foothills, bajadas, and canyons.	Omnivorous—small mammals, lizards, eggs of birds and reptiles.
Common chuckwalla (*Sauromalus ater*)[1]	Crevices of boulder fields, rock outcroppings, lava fields. Note: Eggs buried in a nest of soil.	Herbivorous—plants.

[1]Included as a BLM species of concern.

[2]Included as a USFS species of concern.

[3]Included as a NPS species of management concern.

Table 4. Exposure pathway matrix for aquatic and terrestrial biological receptors.

Receptor	Ingestion	Inhalation	Cell membrane-mediated uptake	Cutaneous absorption	Biotic uptake or trophic transfer
Aquatic habitats: Lentic, lotic, and wetland systems					
Algae, cyanobacteria, and microorganisms			•		•
Aquatic vascular plants			•	•	•
Aquatic invertebrates	•		•	•	•
Fish	•		•	•	•
Terrestrial habitats: Upland and riparian systems					
Soil microorganisms			•	•	•
Terrestrial plants			•	•	•
Terrestrial invertebrates	•	•	•	•	•
Amphibians	•	•	•	•	•
Reptiles	•	•	•	•	•
Birds	•	•	•	•	•
Mammals	•	•	•	•	•

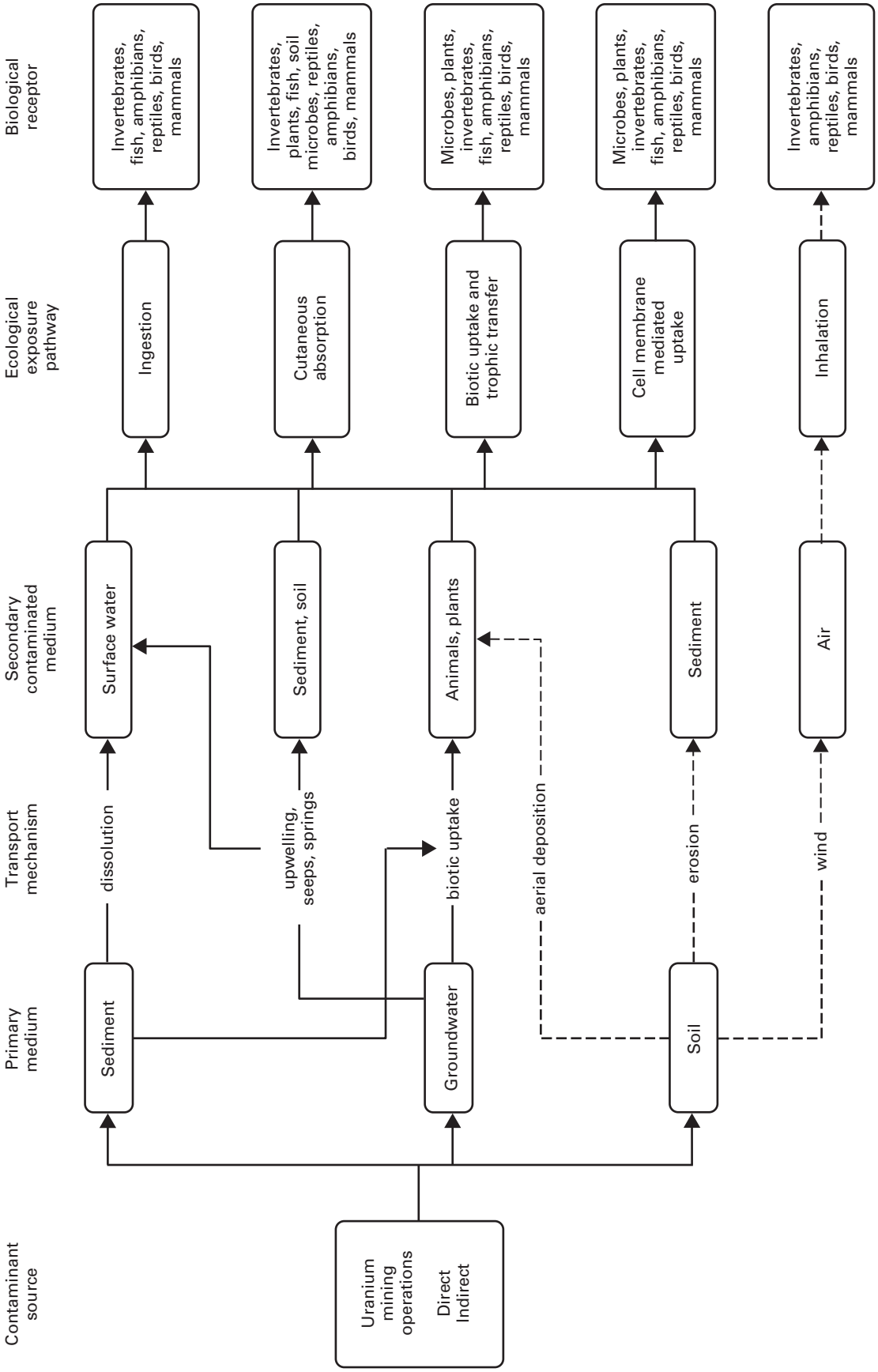

Figure 8. Exposure pathways among generalized terrestrial and aquatic habitats.

Biological data related to uranium that is specific to the segregation areas are limited. Concentrations of uranium in biota within the segregation areas have not been reported by USFS (Angela Gatto and Jeff Waters, personal commun., October 2009), BLM (Elroy Masters, personal commun., October 2009), Arizona Game and Fish Department (Sabra Schwartz, personal commun., October 2009), U.S. Fish and Wildlife Service (USFWS; Carrie Marr, personal commun., September 2009), or National Park Service (NPS; Martha Hahn, personal commun., October 2009). The only information identified for uranium concentrations in biota is from a study east of the segregation areas. Uranium concentrations were less than 0.09 mg/kg dry weight in invertebrates (aquatic beetles, water boatmen, and annelids), 0.27 mg/kg dry weight in whole-body fish (plains killifish, *Fundulus zebrinus*; Lower Colorado River sucker, *Catostomus* sp.; common carp, *Cyprinus carpio*; fathead minnow, *Pimephales promelas*; and green sunfish, *Lepomis cyanellus*), and 0.02 mg/kg dry weight in bird carcasses (killdeer, *Charadrius vociferus*; least sandpiper, *Calidris minutilla*; semipalmated plover, *Charadrius semipalmatus*; and spotted sandpiper, *Actitis macularia*) in the Rio Puerco and Little Colorado River drainages in Arizona (Andrews and others, 1995).

Ecotoxicity Values

The ecotoxicity data that follow are intended to provide overviews of biological responses potentially associated with uranium exposure in the environment. This information is best presented in two major categories: chemical hazards and radiation hazards. Within these two categories, biological receptor groups are further subdivided into aquatic and terrestrial environments. This chapter will also distinguish between two types of effects endpoints available in the scientific literature: effects thresholds and guidance values. Effects thresholds are acute and chronic values derived from empirical data based on a primary study. Guidance values are consensus values commonly derived from effect thresholds (acute and chronic empirical data) from multiple sources that establish benchmark values for the protection of biota or based on expert judgment. These two types of data will be presented separately in this document. While it is important to recognize that guidance values exist, the application of such toxicity benchmarks to any study should not be done without understanding how they were derived.

Biological receptors (fish and wildlife, aquatic and terrestrial invertebrates, vascular and nonvascular plants, a wide array of soil microorganisms) are potentially exposed to elements of the ^{238}U decay series, particularly as those elements occur within minerals and ores in deposits such as breccia pipes. These chemicals may attain concentrations that are toxic to biota in the segregation areas when encountered through the ingestion of prey and water, incidental ingestion of soil, inhalation of airborne contaminants, and dermal uptake. These radionuclides also present radiation hazards if exposure pathways are complete and exposure is sufficient to yield adverse effects in receptors. Therefore, radiation hazards were summarized separately from chemical hazards for each radionuclide.

Radionuclides from the ^{238}U decay series were the focus of this chapter, although toxicity data for radionuclides from other actinide decay series (for example, thallium) were included when available. Existing scientific literature provided the basis for the synoptic data compilation, which represents the status of the current literature and identifies data gaps in toxicity data. Available information was compiled on microbial, plant, and animal species and on effects linked to exposures to uranium and other radionuclides, including an overview of transfer of the contaminants from water and soil to biota. To help identify data gaps for receptor groups, the chemical toxicity data from the literature compilation were subjectively categorized by biological receptor as none (no chemical toxicity data available), minimal (chemical toxicity data available for <5 species and from <3 individual studies), low (chemical toxicity data available for 5–10 species and from 3–5 individual studies), or moderate (chemical toxicity data available for >10 species and from >5 individual studies) (table 5). Existing guidance values from the scientific literature were also included, although the methodologies for deriving these values were not critiqued. Data regarding biota sensitivity to radionuclides were also included when available. Data for special status and nonstatus species occurring within the segregation areas were not available except for several fish species. Tabulations of chemical exposure concentrations and radiation dose based on available guidance are also included as a linkage to the pathway analysis (tables 6–8). Derivation of radiation dose is detailed in Turner (2007) and Bréchignac and Desmet (2005), and exposure units used to characterize radiation toxicity are detailed in appendix 2.

Ecotoxicity data were compiled to provide relevant information on chemical hazards to aquatic and terrestrial biota of concern; data were limited to radionuclides of the ^{238}U decay series including uranium, thallium, thorium, radium, and radon because they are relatively long-lived (table 7). Availability of ecotoxicological data varies among those radionuclides likely encountered in field exposures, but was most abundant for uranium and thallium (tables 7–8). As a radionuclide of ^{234}U and ^{235}U decay series, thallium has a short half life (< 5 minutes), but in field settings it occurs predominately in its stable isotopes (^{203}Tl and ^{205}Tl). As a result, ecotoxicological data for the chemical toxicity of thallium are available for multiple receptors (table 8). The ecotoxicological data consist of a compilation of existing acute and chronic guidance values and effects thresholds. Information—such as exposure route, exposure duration, exposure compound, life stage exposed, and endpoint evaluated—that is relevant to characterizing benchmarks was included in the summary (tables 6–8). There were no selection criteria for including or excluding any guidance value or effects threshold into the summary of the

Table 5. Amount of chemical toxicity data available for biological receptors exposed to radionuclides in the uranium decay series considered in this summary.

[None, no chemical toxicity data; Minimal, chemical toxicity data available for <5 species and <3 primary studies; Low, chemical toxicity data available for 5–10 species and 3–5 primary studies; Moderate, chemical toxicity data available for >10 species and >5 primary studies]

Receptor	Uranium	Thallium	Thorium	Bismuth	Radium	Radon*	Protactinium*	Polonium*
Algae	Minimal	Minimal	Minimal	Minimal	None	None	None	None
Cyanobacteria	None	None	None	None	None	None	None	None
Microorganisms	Minimal	Minimal	None	None	None	None	None	None
Aquatic vascular plants	Minimal	Minimal	None	None	None	None	None	None
Aquatic invertebrates	Moderate	Moderate	Minimal	Minimal	None	None	None	None
Fish	Moderate	Low	Minimal	None	None	None	None	None
Soil microorganisms	Minimal	None	None	None	None	None	None	None
Terrestrial plants	Low	None	None	None	None	None	None	None
Terrestrial invertebrates	Minimal	None	None	None	None	None	None	None
Amphibians	Minimal	Minimal	None	None	None	None	None	None
Reptiles	None	None	None	None	None	None	None	None
Birds	None	None	None	None	None	None	None	None
Mammals, wild	None	None	None	None	None	None	None	None
Mammals, standard laboratory species	Moderate	Low	Minimal	None	None	None	None	None

*Primarily a radiation hazard.

literature compilation because (1) the objective was to compile as much uranium toxicity literature possible, and (2) the toxicity data available was expected to be limited for certain biological receptors. Also, because available data were limited for some receptors, the toxicity tables include dose and dose rates. Guidance values from the scientific literature are presented (table 6). The derivations of these benchmark toxicity values need to be understood before they can be incorporated into risk assessments for the segregation areas, but information on how these benchmarks were derived is beyond the scope of this document. Chronic endpoints such as no observed effect concentrations (NOECs) and lowest observed effect concentrations (LOECs) were reported instead of acute endpoints such as LC50s and EC50s (lethal and effect concentrations for 50 percent of the organisms tested) from the same study in order to provide the most conservative effects data available and to minimize the length of summary tables. However, acute data (LC50s and EC50s) would be important to consider when deriving or estimating benchmark or guidance values (Crane and Newman, 2000; Scholze and others, 2001; European Commission, 2003; Lepper, 2005; Canadian Council of Ministers of the Environment, 2007). In addition, NOECs presented may have included bounded (both NOEC and LOEC determined) and unbounded (no effect at the highest exposure evaluated) values. Such information should be indentified when guidance values are being derived because unbounded NOECs from studies that did not evaluate very high exposures can imply a potential sensitivity that is unfounded. Ecotoxicity data was not provided for lead, the stable end-state of the uranium decay series; the chemical toxicity of lead is well characterized for aquatic and terrestrial biota (for example, U.S. Environmental Protection Agency, 2005). Similarly, benchmark values are available for many metals co-occurring in uranium deposits characteristic of field settings (for example, see *http://www.epa.gov/ecotox/ecossl/*).

Chemical Hazards of Uranium

The chemical toxicity of uranium presents a variety of concerns for biota and ecosystem, particularly in areas where ore deposits are associated with mining operations. Driver (1994) and Eisler (1994) provided early compilations of chemical hazards associated with uranium. Data on the chemical toxicity, bioconcentration, and bioaccumulation of uranium for terrestrial systems are less developed compared to aquatic systems. The range of reported toxicity values for uranium varies widely, presumably because its toxicity is heavily influenced by the chemistry of the associated matrix (water, sediment, soil). For example, uranium tends to be more toxic to aquatic biota in soft water than in hard water (Paquin and others, 2003; Meyer and others, 2007).

Biological Receptors in the Aquatic Food Chain

Algae, Cyanobacteria, and Aquatic Microorganisms

Chemical toxicity data for algae, cyanobacteria, and aquatic microorganisms are limited, and responses to uranium exposure varies among receptors (table 7). Gus'Kova and others (1966, cited in Driver, 1994) reported that uranium (specifically the uranyl ion) inhibited the growth of aquatic microflora at about 1.0 mg/L in freshwater systems and was bactericidal at 100 mg/L. Diatom survival was reduced at an exposure of 1.0 mg/L (Gross and Koczy, 1946, cited in Driver, 1994), whereas a field study (Ruggles and others, 1979, cited in Driver, 1994) reported abundant diatom populations in tailing waters with 17 mg/L of uranium. Algae (*Scenedesmus*) experienced growth inhibition at 22 mg/L, and a protozoan (*Microregma*) had reduced food intake at 28 mg/L of uranium (Bringman and Kuhn, 1959, cited in Driver, 1994).

Table 6. Guidance values for chemical and radiation toxicity. Guidance values for the protection of biota are consensus values based on expert judgment or derived from models using effect thresholds (acute and chronic empirical data) from multiple sources. The guidance values are not recommended for use as toxicity benchmarks in any study without understanding how they were derived. Natural background concentrations of uranium in the segregation areas are also presented.

[LC, lethal concentration; EC, effect concentration; NOAEL, no adverse effect level; LOAEL, lowest adverse effect level; ADEQ, Arizona Department of Environmental Quality; CCME, Canadian Council of Ministers of the Environment; U.S. DOE, U.S. Department of Energy; NCRP, National Council on Radiation Protection and Measurements; IAEA, International Atomic Energy Agency; UNSCEAR, United Nations Scientific Committee on the Effects of Atomic Radiation; ACRP, Advisory Committee on Radiological Protection; ICRP, International Commission on Radiological Protection; EC, European Commission]

Hazard and receptor	Exposure matrix	Concentration/ rate	Comment	Reference
			Uranium—Chemical toxicity	
Aquatic biota				
Aquatic life	Water	0.0026 mg/L	Chronic Tier II threshold for aquatic life.	Suter and Tsao, 1996
Aquatic life	Water	0.0035 mg/L	Arizona Water Quality Criteria.	ADEQ, 2003
Aquatic life	Water	0.011 mg/L	Aquatic life threshold based on geometric mean of estimated no effect values with hardness <100 mg $CaCO_3$/L.	Franklin and others, 2000
Aquatic life	Water	0.046 mg/L	Acute Tier II threshold for aquatic life.	Suter and Tsao, 1996
Aquatic life	Water	0.142 mg/L	Lowest chronic threshold for aquatic life.	Suter and Tsao, 1996
Aquatic life	Water	0.92 mg/L	EC25 derived from acute LC50 for aquatic life.	Liber and Sobey, 1999
Aquatic life	Water	0.457–6.915 mg/L	Range based on all aquatic life uses for Arizona, Colorado, New Mexico, and Utah.	Brown, 2008
Aquatic vascular plants	Water	0.005 mg/L	Predicted no effect concentration.	Sheppard and others, 2005
Aquatic invertebrates	Water	0.005 mg/L	Predicted no effect concentration.	Sheppard and others, 2005
Aquatic invertebrates	Sediment	100 mg/kg dry weight	Predicted no effect concentration.	Sheppard and others, 2005
Fish	Water	0.4 mg/L	Predicted no effect concentration; very soft water (<10 mg $CaCO_3$/L).	Sheppard and others, 2005
Fish	Water	2.8 mg/L	Predicted no effect concentration; soft water (10–100 mg $CaCO_3$/L).	Sheppard and others, 2005
Fish	Water	23 mg/L	Predicted no effect concentration; hard water (>100 mg $CaCO_3$/L).	Sheppard and others, 2005
Terrestrial biota				
Soil microorganisms	Soil	100 mg/kg dry soil	Predicted no effect concentration.	Sheppard and others, 2005
Terrestrial plants	Soil	5 mg/kg	Based on phytotoxicity.	Efroymson and others, 1997a,b
Terrestrial plants	Soil	5 mg/kg	Based on Model Toxics Control Act on Oak Ridge National Laboratory plants.	U.S. DOE, 2005
Terrestrial plants	Soil	250 mg/kg dry soil	Predicted no effect concentration.	Sheppard and others, 2005
Terrestrial plants	Tissue residue	270 mg/kg dry soil	Toxicity threshold based on root tissue; depleted uranium.	Meyer and others, 2004
Birds	Water	69 mg/L	LOAEL benchmark based on drinking water for rough winged swallow; benchmark not available for piscivorous birds.	U.S. DOE, 2005
Mammals	Internal dose	0.1 mg/kg/d	Predicted no effect concentration.	Sheppard and others, 2005
Mammals	Water	7 mg/L	NOAEL wild mammal benchmark based on drinking water for white tailed deer; benchmark not available for piscivorous mammals; LOAEL is 14 mg/L.	U.S. DOE, 2005
Mammals	Soil	200 mg/kg	Arizona Soil Remediation Level.	ADEQ, 2007
Background				
Soil	Soil	2.3 mg/kg dry weight	Natural background in segregation area.	Smith, 1997
Sediment	Sediment	2 mg/kg dry weight	Natural background in segregation area.	Hopkins and others, 1984
Water	Water	0.032 mg/L	Natural background in segregation area.	see Bills and others, this report
Aquatic invertebrates	Tissue residue	<0.09 mg/kg	Background concentration in invertebrates from Rio Puerco and Little Colorado River in northern Arizona.	Andrews and others, 1995
Fish	Tissue residue	<0.27 mg/kg	Background concentration in whole-body fish from Rio Puerco and Little Colorado River in northern Arizona.	Andrews and others, 1995
Birds	Tissue residue	<0.02 mg/kg	Background concentration in bird carcasses from Rio Puerco and Little Colorado River in northern Arizona.	Andrews and others, 1995

Table 6. Guidance values for chemical and radiation toxicity. Guidance values for the protection of biota are consensus values based on expert judgment or derived from models using effect thresholds (acute and chronic empirical data) from multiple sources. The guidance values are not recommended for use as toxicity benchmarks in any study without understanding how they were derived. Natural background concentrations of uranium in the segregation areas are also presented.—Continued

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Hazard and receptor	Exposure matrix	Concentration/ rate	Comment	Reference
			Thallium—Chemical toxicity	
Aquatic biota				
Aquatic life	Water	0.012 mg/L	Chronic Tier II threshold for aquatic life.	Suter and Tsao, 1996
Aquatic life	Water	0.11 mg/L	Acute Tier II threshold for aquatic life.	Suter and Tsao, 1996
Aquatic plants	Water	0.1 mg/L	Lowest chronic threshold.	Suter and Tsao, 1996
Aquatic invertebrate	Water	0.13 mg/L	Lowest chronic threshold for daphnids.	Suter and Tsao, 1996
Fish	Water	0.057 mg/L	Lowest chronic threshold.	Suter and Tsao, 1996
			Radionuclides—Radiation toxicity	
Aquatic biota				
Freshwater biota	Environmental[a]	0.01 mGy/h	Proposed for protection of nonhuman populations.	Garnier-Laplace and others, 2006
Freshwater biota	Environmental[a]	0.4 mGy/h	Proposed for protection of nonhuman populations.	NCRP, 1991; IAEA, 1992; UNSCEAR, 1996
Algae and aquatic plants	Environmental[a]	2.4–10 mGy/d	Estimated No Effect Value based on a statistical model.	NCRP, 1991; UNSCEAR, 1996; Bird and others, 2003; Thompson and others, 2003
Algae	Environmental[a]	0.1 mGy/h	Proposed for protection of nonhuman populations.	CCME, 2007
Aquatic plants	Environmental[a]	0.1 mGy/h	Proposed for protection of nonhuman populations.	CCME, 2007
Benthic invertebrates	Environmental[a]	0.2 mGy/h	Proposed for protection of nonhuman populations.	CCME, 2007
Benthic invertebrates	Environmental[a]	1.1–5.5 mGy/d	Estimated No Effect Value based on a statistical model.	Harrison and Anderson, 1994; EC, 2001, cited in ACRP, 2002, Bird and others, 2003; Thompson and others, 2003
Frog (Reference)	Environmental[a]	0.004–0.04 mGy/h	Proposed for protection of nonhuman populations.	ICRP, 2007
Trout (Reference)	Environmental[a]	0.040–0.4 mGy/h	Proposed for protection of nonhuman populations.	ICRP, 2007
Fish	Environmental[a]	0.02 mGy/h	Proposed for protection of nonhuman populations.	CCME, 2007
Fish and aquatic biota	Environmental[a]	0.55–10,951 mGy/d	Estimated No Effect Value based on a statistical model.	NCRP, 1991; Marshall, 1962, 1967, and Trabalka and others, 1977, all cited in Rose, 1992; UNSCEAR, 1996; Konno, 1980, cited in Le Francois and others, 1999; Bird and others, 2003; Higley and others, 2003; Jones and others, 2003; Knowles, 2003; Thompson and others, 2003
Terrestrial biota				
Fungi	Environmental[a]	4928 mGy/d	Estimated No Effect Value based on a statistical model.	Rose, 1992
Terrestrial plants	Environmental[a]	0.01 mGy/h	Proposed for protection of nonhuman populations.	Garnier-Laplace and others, 2006
Terrestrial plants	Environmental[a]	0.100 mGy/h	Proposed for protection of nonhuman populations.	CCME, 2007
Terrestrial plants	Environmental[a]	0.4 mGy/h	Proposed for protection of nonhuman populations.	IAEA, 1992; UNSCEAR, 1996
Terrestrial plants	Environmental[a]	2.7–10 mGy/d	Estimated No Effect Value based on a statistical model.	IAEA, 1992; Rose, 1992; UNSCEAR, 1996; Bird and others, 2003; Higley and others, 2003; Jones and others, 2003
Wild grass (Reference)	Environmental[a]	0.040–0.4 mGy/h	Proposed for protection of nonhuman populations.	ICRP, 2007
Deciduous plants	Environmental[a]	40 mGy/h	Threshold for damage.	Sazykina, 2005
Conifers	Environmental[a]	0.4–4 mGy/h	Threshold for damage.	Sazykina, 2005
Conifers	Environmental[a]	4–40 mGy/h	Threshold for mortality.	Sazykina, 2005
Pine tree (Reference)	Environmental[a]	4–40 µGy/h	Proposed for protection of nonhuman populations.	ICRP, 2007
Terrestrial invertebrates	Environmental[a]	1.1–5.5 mGy/d	Estimated No Effect Value based on a statistical model.	EC, 2001, cited in ACRP, 2002; Bird and others, 2003
Terrestrial invertebrates	Environmental[a]	0.20–0.40 mGy/h	Threshold for morbidity derived from vertebrate threshold.	Sazykina, 2005
Terrestrial invertebrates	Environmental[a]	4–40 mGy/h	Threshold for reproduction.	Sazykina, 2005
Terrestrial invertebrates	Environmental[a]	40 mGy/h	Threshold for mortality.	Sazykina, 2005
Terrestrial invertebrates	Environmental[a]	0.2 mGy/h	Proposed for protection of nonhuman populations.	CCME, 2007
Bee (Reference)	Environmental[a]	0.4–4 mGy/h	Proposed for protection of nonhuman populations.	ICRP, 2007

Table 6. Guidance values for chemical and radiation toxicity. Guidance values for the protection of biota are consensus values based on expert judgment or derived from models using effect thresholds (acute and chronic empirical data) from multiple sources. The guidance values are not recommended for use as toxicity benchmarks in any study without understanding how they were derived. Natural background concentrations of uranium in the segregation areas are also presented.—Continued

[LC, lethal concentration; EC, effect concentration; NOAEL, no adverse effect level; LOAEL, lowest adverse effect level; ADEQ, Arizona Department of Environmental Quality; CCME, Canadian Council of Ministers of the Environment; U.S. DOE, U.S. Department of Energy; NCRP, National Council on Radiation Protection and Measurements; IAEA, International Atomic Energy Agency; UNSCEAR, United Nations Scientific Committee on the Effects of Atomic Radiation; ACRP, Advisory Committee on Radiological Protection; ICRP, International Commission on Radiological Protection; EC, European Commission]

Hazard and receptor	Exposure matrix	Concentration/ rate	Comment	Reference
			Radionuclides—Radiation toxicity—Continued	
Earthworm (Reference)	Environmental[a]	0.4–4 mGy/h	Proposed for protection of nonhuman populations.	ICRP, 2007
Amphibians and reptiles	Environmental[a]	2–2.7 mGy/d	Estimated No Effect Value based on a statistical model.	Bird and others, 2003; Thompson and others, 2003
Birds	Environmental[a]	0.14–5 mGy/d	Estimated No Effect Value.	Zach and Mayoh, 1986; Zach and others, 1993
Duck (Reference)	Environmental[a]	0.004–0.04 mGy/h	Proposed for protection of nonhuman populations.	ICRP, 2007
Mammals	Environmental[a]	0.100 mGy/h	Proposed for protection of nonhuman populations.	CCME, 2007
Mammals (Sensitive)	Environmental[a]	0.004–0.02 mGy/h	Threshold for minor cytogenetic effects.	Sazykina, 2005
Mammals (Sensitive)	Environmental[a]	0.02–0.08 mGy/h	Threshold for minor morbidity effects.	Sazykina, 2005
Mammals	Environmental[a]	0.08–0.2 mGy/h	Threshold for reproduction.	Sazykina, 2005
Mammals	Environmental[a]	4–40 mGy/h	Acute radiation sickness in vertebrates.	Sazykina, 2005
Mammals	Environmental[a]	40 mGy/h	Lethal dose within several days for vertebrates.	Sazykina, 2005
Deer (Reference)	Environmental[a]	0.004–0.04 mGy/h	Proposed for protection of nonhuman populations.	ICRP, 2007
Rat (Reference)	Environmental[a]	0.004–0.04 mGy/h	Proposed for protection of nonhuman populations.	ICRP, 2007
Animals	Environmental[a]	0.04 mGy/h	Proposed for protection of nonhuman populations.	IAEA, 1992
Animals	Environmental[a]	1–44 mGy/d	Estimated No Effect Value based on a statistical model.	IAEA, 1992; Childs and others, 1966, and French and others, 1966, both cited in Rose, 1992; UNSCEAR, 1996; Bird and others, 2003; Higley and others, 2003; Jones and others, 2003; Mihok, 2003
Animals	Environmental[a]	0.040–0.1 mGy/h	Proposed for protection of nonhuman populations.	UNSCEAR, 1996
Animals	Environmental[a]	0.01 mGy/h	Proposed for protection of nonhuman populations.	Garnier-Laplace and others, 2006
Background				
Animals (Terrestrial)	Environmental[a]	0.00001–0.00044 mGy/h	Estimated background concentration but not specific to the segregation area.	Garnier-Laplace and others, 2006
Plants (Terrestrial)	Environmental[a]	0.00002–0.0007 mGy/h	Estimated background concentration but not specific to the segregation area.	Garnier-Laplace and others, 2006
Freshwater organisms	Environmental[a]	0.000022–0.00018 mGy/h	Estimated background concentration but not specific to the segregation area.	Garnier-Laplace and others, 2006
Natural	Environmental[a]	<0.00004 mGy/h	Estimated background concentration but not specific to the segregation area.	Sazykina, 2005

[a]Dose rate accounts for multiple routes of exposure for individual receptors to yield a total dose rate for ionizing radiation.

Table 7. Biological effects thresholds from empirical data related to the chemical toxicity of uranium.

[LC, lethal concentration; EC, effect concentration; NOEC, no effect concentration; LOEC, lowest effect concentration; NOAEL, no adverse effect level; LOAEL, lowest adverse effect level; dw, dry weight; uranyl nitrate, depleted uranium; BLM, biotic ligand model; ATSDR, Agency for Toxic Substances and Disease Registry. Units were changed to have consistent units within a receptor class. To simplify the table, the following measures were taken. The LC50 was not reported if a NOEC or LOEC was available from study. The lowest NOEC value was retained if multiple NOECs were available for the same species from the same study. The lowest LC50 or EC50 was reported to be conservative regardless of exposure length (for example, Khangarot, 1991). The lowest NOEC was reported on repeated toxicity tests from Pickett and others (1993). See text for further explanation]

Biological receptor (species)	Exposure route	Concentration/ rate	Effect	Comment[a]	Reference
			Algae, cyanobacteria, and aquatic microorganisms		
Green algae (*Chlorella* sp.)	Water	0.0007 mg/L	Growth	NOEC; LOEC = 0.0016 mg/L; pH 7.0; hardness 8 mg $CaCO_3$/L.	Charles and others, 2002
Green algae (*Chlorella* sp.)	Water	0.0045 mg/L	Growth	NOEC; LOEC = 0.012 mg/L; pH 7.0; hardness 400 mg $CaCO_3$/L.	Charles and others, 2002
Green algae (*Chlorella* sp.)	Water	0.011 mg/L	Growth	NOEC; pH 6.5; hardness 2–4 mg $CaCO_3$/L.	Franklin and others, 2000
Green algae (*Chlorella* sp.)	Water	0.072–0.157 mg/L	Growth	NOEC; LOEC = 0.120–0.187 mg/L; natural creek water.	Hogan and others, 2005
Periphyton communities (algae, diatoms, fungi, bacteria)	Water	0.1 mg/L	Growth	NOEC for biomass; pH 9.1; exposed to river water and uranium trioxide.	Small and others, 2008
Aquatic microflora	Water	1.0 mg/L	Growth	Inhibit growth.	Gus'Kova and others,1966, cited in Driver, 1994
Green algae (*Chlorella* sp.)	Water	2 mg/L	Growth	NOEC; LOEC = 4 mg/L; pH 2.7; uranyl nitrate.	De Jong, 1965
Algae (*Scenedesmus* sp.)	Water	22 mg/L	Growth	Inhibit cell division.	Bringman and Kuhn, 1959, cited in Driver, 1994
Diatoms	Water	1.0 mg/L	Mortality	Reduced survival.	Gross and Koczy, 1946, cited in Driver, 1994
Algae (*Scenedesmus subspicatus*)	Water	36.3 mg/L	Mortality	LC50; 5 d; water from uranium processing plant.	Vinot and Larpent, 1984
Aquatic microflora	Water	100 mg/L	Mortality	Bactericidal.	Gus'Kova and others,1966, cited in Driver, 1994
Diatoms	Water	17 mg/L	None	Abundant populations in waters on uranium mill tailings.	Ruggles and others, 1979, cited in Driver, 1994
Protozoan (*Microregma* sp.)	Water	28 mg/L	Systemic	Inhibition of food intake; uranyl nitrate.	Bringman and Kuhn, 1959, cited in Driver, 1994
			Aquatic vascular plants		
Duckweed (*Lemna aequinoctialis*)	Water	0.112 mg/L	Growth	LOEC; pH 6.5; hardness 38 mg $CaCO_3$/L; primarily uranyl hydroxycarbonate (76%) and uranyl carbonate (13%).	Charles and others, 2006
			Aquatic invertebrates		
Midge (*Chironomus riparius*)	Sediment	5.3 mg/kg dw	Mortality	LC50; 10-d exposure; artificial sediments contaminated with uranyl nitrate.	Dias and others, 2008
Amphipod (*Hyalella azteca*)	Sediment	57 mg/kg dw	Mortality	LC50 for juveniles; 14-d exposure; artificial sediments contaminated with uranium.	CCME, 2007
Midge (*Chironomus riparius*)	Sediment	856 mg/kg dw	Mortality	LC50; spiked sediments with uranyl nitrate.	Lagauzere and others, 2009b
Worm (*Tubifex tubifex*)	Sediment	2910 mg/kg dw	Mortality	LC50; pH 8.2; spiked sediments with uranyl nitrate.	Lagauzere and others, 2009b
Water flea (*Ceriodaphnia dubia*)	Sediment	1.97 mg/L	Reproduction	NOEC; LOEC = 3.91 mg/L; hardness 190 mg $CaCO_3$/L; pH 8.49; 7-d exposure with depleted uranium in soil.	Kuhne and others, 2002
Worm (*Tubifex tubifex*)	Sediment	547 mg/kg dw	Systemic	EC50; spiked sediments with uranyl nitrate.	Lagauzere and others, 2009a
Midge (*Chironomus riparius*)	Sediment	787 mg/kg dw	Systemic	EC50; spiked sediments with uranyl nitrate.	Lagauzere and others, 2009a
Green hydra (*Hydra viridissima*)	Water	0.032 mg/L	Growth	LOEC for asexual reproduction in soft water (hardness, 6.6 mg $CaCO_3$/L); synthetic water.	Riethmuller and others, 2001

Table 7. Biological effects thresholds from empirical data related to the chemical toxicity of uranium.—Continued

[LC, lethal concentration; EC, effect concentration; NOEC, no effect concentration; LOEC, lowest effect concentration; NOAEL, no adverse effect level; LOAEL, lowest adverse effect level; dw, dry weight; uranyl nitrate, depleted uranium; BLM, biotic ligand model; ATSDR, Agency for Toxic Substances and Disease Registry. Units were changed to have consistent units within a receptor class. To simplify the table, the following measures were taken. The LC50 was not reported if a NOEC or LOEC was available from study. The lowest NOEC value was retained if multiple NOECs were available for the same species from the same study. The lowest LC50 or EC50 was reported to be conservative regardless of exposure length (for example, Khangarot, 1991). The lowest NOEC was reported on repeated toxicity tests from Pickett and others (1993). See text for further explanation]

Biological receptor (species)	Exposure route	Concentration/ rate	Effect	Comment[a]	Reference
			Aquatic invertebrates—Continued		
Green hydra (*Hydra viridissima*)	Water	0.032 mg/L	Growth	LOEC for asexual reproduction; hardness 6.6 mg $CaCO_3$/L.	Riethmuller and others, 2000, cited in Charles and others, 2006
Midge (*Chironomus tentans*)	Water	0.039 mg/L	Growth	NOEC for growth; LOEC = 0.157 mg/L; 10 d; larvae; uranyl nitrate.	Muscatello and Liber, 2009
Green hydra (*Hydra viridissima*)	Water	0.042 mg/L	Growth	LOEC for asexual reproduction in high alkalinity hard water (hardness 165 mg $CaCO_3$/L); synthetic water.	Riethmuller and others, 2001
Green hydra (*Hydra viridissima*)	Water	0.061 mg/L	Growth	LOEC; pH 6.0; hardness 4 mg $CaCO_3$/L.	Markich and Camilleri, 1997, cited in Charles and others, 2006
Green hydra (*Hydra viridissima*)	Water	0.062 mg/L	Growth	LOEC for asexual reproduction in low alkalinity hard water (hardness 330 mg $CaCO_3$/L); synthetic water.	Riethmuller and others, 2001
Green hydra (*Hydra viridissima*)	Water	0.09 mg/L	Growth	LOEC for asexual reproduction in low alkalinity moderately hard water (hardness 165 mg $CaCO_3$/L); synthetic water.	Riethmuller and others, 2001
Green hydra (*Hydra viridissima*)	Water	0.15 mg/L	Growth	LOEC for population growth in dry season, related to ability to capture live prey; uranyl sulfate.	Hyne and others, 1992
Green hydra (*Hydra viridissima*)	Water	0.15 mg/L	Growth	Growth inhibition.	Hyne and others, 1991, cited in Driver, 1994
Green hydra (*Hydra viridissima*)	Water	0.2 mg/L	Growth	LOEC for population growth in wet season, related to ability to capture live prey; uranyl sulfate.	Hyne and others, 1992
Green hydra (*Hydra vulgaris*)	Water	0.4 mg/L	Growth	LOEC for population growth in wet season; uranyl sulfate.	Hyne and others, 1992
Green hydra (*Hydra vulgaris*)	Water	0.55 mg/L	Growth	LOEC for population growth in dry season; uranyl sulfate.	Hyne and others, 1992
Cladoceran (*Moindaphnia macleayi*)	Water	0.01 mg/L	Mortality	NOEC; LOEC = 0.025 mg/L; 120 h; uranyl sulfate.	Hyne and others, 1993
Water flea (*Daphnia pulex*)	Water	0.220 mg/L	Mortality	LC50; 48 h; 3 mg $CaCO_3$/L; uranyl nitrate.	Trapp and others, 1986, cited in Pickett and others, 1993
Cladoceran (various tropical species)	Water	0.41–1.10 mg/L	Mortality	LC50; 24 h; uranyl sulfate; soft water.	Bywater and others,1991
Amphipod (*Hyalella azteca*)	Water	1.52 mg/L	Mortality	LC50; 96 h; depleted uranium.	Kuhne and others, 2002
Water flea (*Daphnia magna*)	Water	5.87 mg/L	Mortality	LC50; 48 h; moderate-hard water; uranyl sulfate.	Barata and others, 1998
Water flea (*Daphnia magna*)	Water	6.4 mg/L	Mortality	Mean LC50; 48 h; hardness 70 mg $CaCO_3$/L; uranyl nitrate.	Poston and others, 1984
Water flea (*Daphnia magna*)	Water	32.7 mg/L	Mortality	LC50; 24 h; water from uranium processing plant.	Vinot and Larpent, 1984
Water flea (*Daphnia magna*)	Water	37.5 mg/L	Mortality	Mean LC50; 48 h; hardness 133 mg $CaCO_3$/L; uranyl nitrate.	Poston and others, 1984
Water flea (*Daphnia magna*)	Water	52 mg/L	Mortality	Mean LC50; 48 h; hardness 197 mg $CaCO_3$/L; uranyl nitrate.	Poston and others, 1984
Amphipod (*Hyalella azteca*)	Water	186 nmol/L	Mortality	LC50 for juveniles; moderately hard water with low alkalinity; 28-d exposure to spiked sediments with uranyl nitrate trihydrate; BLM model used.	Alves and others, 2008

Table 7. Biological effects thresholds from empirical data related to the chemical toxicity of uranium.—Continued

[LC, lethal concentration; EC, effect concentration; NOEC, no effect concentration; LOEC, lowest effect concentration; NOAEL, no adverse effect level; LOAEL, lowest adverse effect level; dw, dry weight; uranyl nitrate, depleted uranium; BLM, biotic ligand model; ATSDR, Agency for Toxic Substances and Disease Registry. Units were changed to have consistent units within a receptor class. To simplify the table, the following measures were taken. The LC50 was not reported if a NOEC or LOEC was available from study. The lowest NOEC value was retained if multiple NOECs were available for the same species from the same study. The lowest LC50 or EC50 was reported to be conservative regardless of exposure length (for example, Khangarot, 1991). The lowest NOEC was reported on repeated toxicity tests from Pickett and others (1993). See text for further explanation]

Biological receptor (species)	Exposure route	Concentration/ rate	Effect	Comment[a]	Reference
			Aquatic invertebrates—Continued		
Asian clam (*Corbicula fluminea*)	Water	1872 mg/L	Mortality	LC50; uranyl acetate.	Labrot and others, 1999
Amphipod (*Hyalella azteca*)	Water	0.053 mg/L	Mortality	LC50 for juveniles; moderately soft water with high alkalinity; 28-d exposure to spiked sediments with uranyl nitrate trihydrate; BLM model used.	Alves and others, 2008
Water flea (*Ceriodaphnia dubia*)	Water	0.0015 mg/L	Reproduction	NOEC; LOEC = 0.0027 mg/L; uranyl nitrate; Shealy Environmental data.	Pickett and others, 1993
Water flea (*Ceriodaphnia dubia*)	Water	<0.002 mg/L	Reproduction	NOEC; LOEC = <0.002 mg/L; hydrogen uranyl phosphate; Shealy Environmental data.	Pickett and others, 1993
Water flea (*Ceriodaphnia dubia*)	Water	0.002 mg/L	Reproduction	NOEC; LOEC = 0.006 mg/L; hydrogen uranyl phosphate; Normandeau Associates data.	Pickett and others, 1993
Water flea (*Ceriodaphnia dubia*)	Water	<0.008 mg/L	Reproduction	NOEC; LOEC = 0.008 mg/L; uranyl nitrate; Normandeau Associates data.	Pickett and others, 1993
Cladoceran (*Moindaphnia macleayi*)	Water	0.008–0.031 mg/L	Reproduction	NOEC for reproduction; LOEC = 0.020–0.049 mg/L; uranyl sulfate.	Semaan and others, 2001
Water flea (*Ceriodaphnia dubia*)	Water	0.03 mg/L	Reproduction	NOEC; LOEC = 0.05 mg/L; uranium dioxide (IV); Normandeau Associates data.	Pickett and others, 1993
Water flea (*Daphnia magna*)	Water	30 mg/L	Swimming performance	EC50; uranyl acetate.	Knie and others, 1983
Green hydra (*Hydra viridissima*)	Water	1 mg/L	Systemic	Lysis of hydrae; 48 h exposure.	Hyne and others, 1991, cited in Driver, 1994
Worm (*Tubifex tubifex*)	Water	2.05 mg/L	Systemic	EC50; 96 h; uranyl acetate dihydrate.	Khangarot, 1991
Asian clam (*Corbicula fluminea*)	Water	0.00057–0.0059 mg/L	Valve closure/ movement	EC50 for valve closure calculated for uranyl free ion species; pH 5.5 to 6.5; adults.	Fournier and others, 2004
Asian clam (*Corbicula fluminea*)	Water	0.014–0.07 mg/L	Valve closure/ movement	EC50 for valve closure; integration time of response—5 h; pH 5.5 to 6.5; uranyl nitrate solution and adults.	Fournier and others, 2004
Asian clam (*Corbicula fluminea*)	Water	0.048–0.178 mg/L	Valve closure/ movement	EC50 for valve closure; integration time of response—10 to 60 min; pH 5.5 to 6.5; uranyl nitrate solution and adults.	Fournier and others, 2004
Asian clam (*Corbicula fluminea*)	Water	0.102 mg/L	Valve closure/ movement	EC50 for valve closure; pH 7.0; uranyl nitrate solution and adults.	Tran and others, 2008
Mussel (*Velesunio angasi*)	Water	0.285 mg/L	Valve closure/ movement	NOEC; pH 5.5; exposed to synthetic water.	Markich and others, 2000
Mussel (*Velesunio angasi*)	Water	0.365 mg/L	Valve closure/ movement	NOEC for duration of valve opening (mean for males and females); pH 6.0; exposed to synthetic water.	Markich, 2003
			Fish		
Various tropical species	Water	0.2 mg/L	Growth	Threshold effect level—growth; water exposure using uranium sulfate.	Holdway, 1992
Gudgeon (*Mogurnda mogurnda*)	Water	0.404 mg/L	Growth	NOEC; 1-day-old tropical species; uranium sulfate.	Holdway, 1992
Gudgeon (*Mogurnda mogurnda*)	Water	1.22 mg/L	Growth	LOEC; pH 6.0; hardness 6.6 mg $CaCO_3$/L.	Riethmuller and others, 2000, cited in Charles and others, 2006
Gudgeon (*Mogurnda mogurnda*)	Water	1.3 mg/L	Growth	LOEC; pH 6.0; hardness 4 mg $CaCO_3$/L.	Markich and Camilleri, 1997, cited in Charles and others, 2006

Table 7. Biological effects thresholds from empirical data related to the chemical toxicity of uranium.—Continued

[LC, lethal concentration; EC, effect concentration; NOEC, no effect concentration; LOEC, lowest effect concentration; NOAEL, no adverse effect level; LOAEL, lowest adverse effect level; dw, dry weight; uranyl nitrate, depleted uranium; BLM, biotic ligand model; ATSDR, Agency for Toxic Substances and Disease Registry. Units were changed to have consistent units within a receptor class. To simplify the table, the following measures were taken. The LC50 was not reported if a NOEC or LOEC was available from study. The lowest NOEC value was retained if multiple NOECs were available for the same species from the same study. The lowest LC50 or EC50 was reported to be conservative regardless of exposure length (for example, Khangarot, 1991). The lowest NOEC was reported on repeated toxicity tests from Pickett and others (1993). See text for further explanation]

Biological receptor (species)	Exposure route	Concentration/ rate	Effect	Comment[a]	Reference
			Fish—Continued		
Zebrafish (*Danio rerio*)	Water	0.02 mg/L	Mortality	LC50; larval fish; 15-d exposure of depleted uranium.	Bourrachot and others, 2008
Various tropical species	Water	0.73–3.46 mg/L	Mortality	LC50; 96 h; uranyl sulfate; soft water; adults.	Bywater and others, 1991
Rainbowfish (*Melanotaenia splendida inornata*) and gudeon (*Mogurnda morgurnda*)	Water	0.81 mg/L	Mortality	NOEC; LOEC = 1560 µg/L; uranium sulfate.	Holdway, 1992
Bluegill (*Lepomis macrochirus*)	Water	1.67 mg/L	Mortality	LC50; 96 h; uranyl nitrate; hardness 3 mg $CaCO_3$/L.	Trapp and others, 1986, cited in Pickett and others, 1993
Fathead minnow (*Pimephales promelas*)	Water	2.8 mg/L	Mortality	LC50; 96 h; soft water; uranyl sulfate; hard water LC50 = 135 mg/L.	Tarzwell and Henderson, 1956
Fathead minnow (*Pimephales promelas*)	Water	3 mg/L	Mortality	LC50; 96 h; pH 7.4; hardness 210 mg $CaCO_3$/L; as UO2+2.	McKee and Wolf, 1963, cited in Driver, 1994
Zebrafish (*Danio rerio*)	Water	3.1 mg/L	Mortality	LC50; adult fish; uranyl acetate.	Labrot and others, 1999
Fathead minnow (*Pimephales promelas*)	Water	3.1 mg/L	Mortality	LC50; 96 h; soft water; uranyl nitrate.	Tarzwell and Henderson, 1956
Fathead minnow (*Pimephales promelas*)	Water	3.7 mg/L	Mortality	LC50; 96 h; soft water; uranyl acetate.	Tarzwell and Henderson, 1956
Brook trout (*Salvelinus fontinalis*)	Water	3.9 mg/L	Mortality	NOEC; LC50 = 5.5 mg/L; soft water; embryos and larvae.	Parkhurst and others, 1984
Rainbow trout (*Oncorhynchus mykiss*)	Water	6.2 mg/L	Mortality	LC50; 96 h; hardness 31 mg $CaCO_3$/L.	Davies, 1980, cited in Driver, 1994
Zebrafish (*Danio rerio*)	Water	6.4 mg/L	Mortality	LC50; 24 h; water from uranium processing plant.	Vinot and Larpent, 1984
Brook trout (*Salvelinus fontinalis*)	Water	8.0 mg/L	Mortality	LC50; 96 h; hardness 31 mg $CaCO_3$/L.	Davies, 1980, cited in Driver, 1994
Brook trout (*Salvelinus fontinalis*)	Water	16.4 mg/L	Mortality	NOEC; hard water; embryos and larvae.	Parkhurst and others, 1984
Fathead minnow (*Pimephales promelas*)	Water	16.7 mg/L	Mortality	LC50; 96 h; hardness 70 mg $CaCO_3$/L.	Poston and others, 1984, cited in Driver, 1994
Flannelmouth sucker (*Catostomus latipinnis*)	Water	43.5 mg/L	Mortality	LC50; acute; uranyl nitrate.	Hamilton and Buhl, 1997
Bonytail (*Gila elegans*)	Water	46 mg/L	Mortality	LC50; 96 h; uranyl nitrate; hard water; fry-juvenile.	Hamilton, 1995
Colorado pikeminnow (*Ptychocheilus lucius*)	Water	46 mg/L	Mortality	LC50; 96 h; uranyl nitrate; hard water; fry-juvenile.	Hamilton, 1995
Razorback sucker (*Xyrauchen texanus*)	Water	46 mg/L	Mortality	LC50; 96 h; uranyl nitrate; hard water; fry-juvenile.	Hamilton, 1995
Fathead minnow (*Pimephales promelas*)	Water	135 mg/L	Mortality	LC50; 96 h; pH 8.2; hardness 135 mg $CaCO_3$/L; as UO2+2.	McKee and Wolf, 1963, cited in Driver, 1994
Common carp (*Cyprinus carpio*)	Water	60 mg/L	None	No effect on hatchability.	Till and Blaylock, 1976, cited in Driver, 1994
Zebrafish (*Danio rerio*)	Water	23.13 mg/L	Systemic	Increased bioaccumulation in liver and gills versus brain and skeletal muscle. Upregulated gene expression in liver for apoptosis, immune responses, and detoxification processes; depleted uranium.	Lerebours and others, 2009

Table 7. Biological effects thresholds from empirical data related to the chemical toxicity of uranium.—Continued

[LC, lethal concentration; EC, effect concentration; NOEC, no effect concentration; LOEC, lowest effect concentration; NOAEL, no adverse effect level; LOAEL, lowest adverse effect level; dw, dry weight; uranyl nitrate, depleted uranium; BLM, biotic ligand model; ATSDR, Agency for Toxic Substances and Disease Registry. Units were changed to have consistent units within a receptor class. To simplify the table, the following measures were taken. The LC50 was not reported if a NOEC or LOEC was available from study. The lowest NOEC value was retained if multiple NOECs were available for the same species from the same study. The lowest LC50 or EC50 was reported to be conservative regardless of exposure length (for example, Khangarot, 1991). The lowest NOEC was reported on repeated toxicity tests from Pickett and others (1993). See text for further explanation]

Biological receptor (species)	Exposure route	Concentration/ rate	Effect	Comment[a]	Reference
Soil microorganisms and protists					
Soil microbes	Soil	500 mg/kg dry soil	Systemic	LOEC for respiration; depleted uranium.	Meyer and others, 1998, cited in Sheppard and others, 2005
Terrestrial plants					
Carrot (*Daucus carota*)	Gel	9.4 mg/L	Growth	EC50 for root length; 34 d; uranyl nitrate; nutriant gel.	Straczek and others, 2009
Scots pine (*Pinus sylvestris*)	Soil	>100 mg/kg dry soil	Growth	NOEC; uranium mill tailings waste disposal soil.	Sheppard and others, 1985
Swiss chard (*Beta vulgaris*)	Soil	10 mg/kg	Growth	Inhibited root growth in mature plants.	Sheppard and others, 1983, cited in Driver, 1994
Purple threeawn (*Aristida purpurea*); Buffalograss (*Buchloe dactyloides*); Little bluestem (*Schizachyrium scoparium*)	Soil	50 mg/kg dry soil	Growth	LOEC for biomass increase; depleted uranium.	Meyer and others, 1998, cited in Sheppard and others, 2005
Soybean (*Glycine max*)	Water (hydroponic)	0.42 mg/L	Mortality, systemic	Adverse effects—chlorosis, early leaf abscission, reduced root growth; root conc. = 57 µg/g dw; shoot conc. = 1.37 µg/g.	Murthy and others, 1984, cited in Driver, 1994
Soybean (*Glycine max*)	Water (hydroponic)	42 mg/L	Mortality, systemic	Adverse effects—widespread necrosis, reduced leaf chlorophyll; root conc. = 938 mg/kg dw; shoot conc. = 91.5 mg/kg.	Murthy and others, 1984, cited in Driver, 1994
Terrestrial invertebrates					
Earthworm (*Eisenia fetida andrei*)	Media	13.5 µg/cm^2	Mortality	LC50; 96 h; filter paper media saturated with uranium acetate salt solution.	Labrot and others, 1999
Earthworm (*Eisenia fetida andrei*)	Media	40 µg/cm^2	Mortality	LC50; 96 h; filter paper media saturated with uranium acetate salt solution.	Riberia and others, 1996
Earthworm (*Lumbricus* sp.)	Soil	1000 mg/kg dry soil	Mortality	NOEC; LOEC = 300 mg/kg; >10% clay.	Sheppard and Evenden, 1992, cited in Sheppard and others, 2005
Springtail (*Folsomia candida*)	Soil	350–1030 mg/kg dry soil	Mortality	EC20; EC20 = 840–2200 mg/kg for reproduction; 35 d; adults; various soil types.	Sheppard and others, 2004, cited in Sheppard and others, 2005
Springtail (*Onychiurus folsomi*)	Soil	92–480 mg/kg dry soil	Mortality	EC20; EC20 = 150–1030 mg/kg for reproduction; 35 d; adults; various soil types.	Sheppard and others, 2004, cited in Sheppard and others, 2005
Earthworm (*Eisenia andrei*)	Soil	1000 mg/kg dry soil	Mortality, reproduction	NOEC; adults; various soil types.	Sheppard and others, 2004, cited in Sheppard and others, 2005
Amphibian and reptiles					
Frog (primarily *Rana catesbeiana*)	Injection	15 mg/kg	Systemic	Kidney lesions; uranium nitrate; lymph sac injections.	Oliver and Smith, 1930
African clawed frog (*Xenopus laevis*)	Water	13.1–54.3 mg/L	Growth	Delay in metamorphosis in 64-d tests (embryo to tadpoles).	Mitchell and others, 2005
Iberian green frog (*Rana perezi*)	Water	1.75 mg/L	Mortality, growth	Reduced survival and growth; 96 h; larvae exposed to 100% uranium mine effluent.	Marques and others, 2008
African clawed frog (*Xenopus laevis*)	Water	77 mg/L	None	No effects in 96-h tests with depleted uranium and embryos.	Mitchell and others, 2005

Table 7. Biological effects thresholds from empirical data related to the chemical toxicity of uranium.—Continued

[LC, lethal concentration; EC, effect concentration; NOEC, no effect concentration; LOEC, lowest effect concentration; NOAEL, no adverse effect level; LOAEL, lowest adverse effect level; dw, dry weight; uranyl nitrate, depleted uranium; BLM, biotic ligand model; ATSDR, Agency for Toxic Substances and Disease Registry. Units were changed to have consistent units within a receptor class. To simplify the table, the following measures were taken. The LC50 was not reported if a NOEC or LOEC was available from study. The lowest NOEC value was retained if multiple NOECs were available for the same species from the same study. The lowest LC50 or EC50 was reported to be conservative regardless of exposure length (for example, Khangarot, 1991). The lowest NOEC was reported on repeated toxicity tests from Pickett and others (1993). See text for further explanation]

Biological receptor (species)	Exposure route	Concentration/ rate	Effect	Comment[a]	Reference
			Mammals		
Rat	Air	0.05 mg/m^3	Systemic	Kidney injury; 1 yr.	ATSDR, 1990a, cited in Driver, 1994
Guinea pig	Air	0.20 mg/m^3	Systemic	Kidney damage; 7.5 mos.	ATSDR, 1990a, cited in Driver, 1994
Dog	Air	0.25 mg/m^3	Systemic	Kidney damage; 1 yr.	ATSDR, 1990a, cited in Driver, 1994
Rabbit	Air	0.25 mg/m3	Systemic	Kidney damage; 6.5 mos.	ATSDR, 1990a, cited in Driver, 1994
Guinea pig	Dermal	2110 mg/kg	Mortality	LC50; uranyl nitrate.	Orcutt, 1949, cited in Driver, 1994
Rat	Dermal	490 mg/kg	Mortality	LC50; uranyl nitrate.	Orcutt, 1949, cited in Driver, 1994
Mice	Dermal	7600 mg/kg	Mortality	LC50; uranyl nitrate.	Orcutt, 1949, cited in Driver, 1994
Rabbit	Dermal	5 mg/kg/d	Mortality	LOAEL; uranyl nitrate hexahydrate.	Orcutt, 1949, cited in Driver, 1994
Dog	Diet	5.7 mg/kg	Mortality	LC50; uranyl nitrate hexahydrate.	Domingo and others, 1987, cited in Driver, 1994
Mice	Diet	25 mg/kg	Mortality	NOAEL; LD50 = 136 mg/kg; uranyl acetate dihydrate.	Domingo and others, 1987, cited in Driver, 1994
Rat	Diet	1940 mg/kg	Mortality	Death; 2 yrs.	ATSDR, 1990a, cited in Driver, 1994
Mice	Diet	<5 mg/kg/d	Reproduction	NOAEL for maternal toxicity, fetotoxicity; LOEC = 2.8 mg/kg/d for growth of pups; uranyl acetate dihydrate.	Domingo and others, 1989a, cited in Domingo, 2001, and Sheppard and others, 2005
Rat	Diet	97 mg/kg/d	Reproduction	Testicular atrophy; uranyl fluoride.	Maynard and Hodge, 1949, and Maynard and others, 1953, cited in Driver, 1994
Rat	Diet	11 mg/kg	Systemic	NOAEL; LOAEL = 118 mg/kg for hepatic lesions, proteinuria, weight loss; uranyl acetate dihydrate.	Domingo and others, 1987, cited in Driver, 1994
Rat	Diet	24 mg/kg/d	Systemic	NOAEL for testicular lesions; uranyl nitrate hexahydrate.	Maynard and others, 1953, cited in Driver, 1994
Mice	Diet	5 mg/kg/d	Systemic	NOAEL for viability and lactation indices; LOEC = 28 mg/kg/d for c litter size; uranyl acetate dihydrate.	Domingo and others, 1989b, cited in Domingo, 2001, and Sheppard and others, 2005
Rabbit	Diet	3 mg/kg/d	Systemic	LOEC for weight loss and limited renal damage.	Maynard and Hodge, 1949, cited in Sheppard and others, 2005
Dog	Diet	9 mg/kg/d	Systemic	LOEC for hepatic effects.	Maynard and others, 1949, cited in Sheppard and others, 2005
Mice	Drinking water	21 mg/L	Mortality	Maternal death.	ATSDR, 1990a, cited in Driver, 1994
Mice	Drinking water	1.9–5.6 mg/kg/d	Reproduction	Threshold effects for reproduction.	Llobet and others, 1991; cited in Sheppard and others, 2005; Feugier and others, 2008; Kundt and others, 2009
Rabbit	Drinking water	0.05–1.36 mg/kg/d	Systemic	LOEC for renal damage; uranyl nitrate.	Gilman and others, 1998a,c, cited in Sheppard and others, 2005
Mice	Drinking water	16 mg/L	Systemic	Decreased body weight.	ATSDR, 1990a, and Domingo and others, 1989a, cited in Driver, 1994
Rat	Drinking water	0.06 mg/kg/d	Systemic	LOEC for renal damage; uranyl nitrate.	Gilman and others, 1998b, cited in Sheppard and others, 2005
Rat	Drinking water	40-471 mg/L	Systemic	Effects in nervous system and major organs.	ATSDR, 1990a, cited in Driver, 1994; Houpert and others, 2007

[a]U(VI) was generally used in all toxicity tests unless otherwise noted.

Table 8. Biological effects thresholds from empirical data related to the chemical toxicity of thallium.

[USACHPPM, U.S. Army Center for Health Promotion and Preventative Medicine; IC, inhibition concentration; LC, lethal concentration; EC, effect concentration; NOEC, no effect concentration; LOEC, lowest effect concentration; LOAEL, lowest adverse effect level. All data from water-borne exposures unless otherwise noted. Single dose exposure data from USACHPPM (2007) were excluded]

Receptor category and criteria species	Concentration (mg/L)	Effect	Comment	Reference
Algae, cyanobacteria, and aquatic microorganisms				
Green algae (*Chlorella vulgaris*)	0.02	Growth	NOEC; LOEC = 0.04; thallium sulfate.	De Jong, 1965
Freshwater rotifer (*Brachionus calyciflorus*)	0.0188	Mortality	LC50; 24 h; thallium sulfate.	Calleja and others, 1994
Fungus (*Geotrichum candidum*)	0.00038	Systemic	IC50 for glucose inhibition; 4 h; thallium sulfate.	Jacobsen, 1995
Aquatic vascular plants				
Duckweed (*Lemna minor*)	0.04–0.08 nmol/cm^3	Growth	Threshold value; EC50 = 0.16–0.23 nmol/cm^3; thallous acetate.	Kwan and Smith, 1988
Duckweed (*Lemna minor*)	0.00008	Growth	LOEC; EC50 = 0.0002 mg/L.	Smith and Kwan, 1989
Aquatic invertebrates				
Amphipod (*Gammarus minus*)	0.1	Mortality	LC50; thallium sulfate.	Horne and others, 1983
Amphipod (*Hyalella azteca*)	0.190–0.310	Mortality	LC50; thallous nitrate added to tap water.	Borgmann and others, 1998
Amphipod (*Hyalella azteca*)	0.01	Reproduction	LC25 for reproduction; 10 wk; thallous nitrate.	Borgmann and others, 1998
Fairy shrimp (*Streptocephalus proboscideus*)	0.0008	Mortality	LC50; 24 h; thallium sulfate.	Calleja and others, 1994
Midge (*Chironomus riparius*)	229	Mortality	LC50; thallium sulfate.	Horne and others, 1983
Nematode (*Caenorhabditis elegans*)	123–200	Mortality	LC50; 96 h; elemental thallium.	Williams and Dusenbery, 1990
Roachfly (*Peltoperla maria*)	1460	Mortality	LC50; thallium sulfate.	Horne and others, 1983
Snail (*Physa heterostropha*)	2.7	Mortality	LC50; thallium sulfate.	Horne and others,1983
Water flea (*Daphnia magna*)	0.0083	Mortality	LC50; 24 h; thallium sulfate.	Calleja and others, 1994
Water flea (*Daphnia magna*)	0.1	Reproduction	NOEC for reproduction in chronic test; LOEC = 0.181; thallium sulfate.	Kimball, 1978
Water flea (*Daphnia magna*)	0.024	Mortality	LC50; 48 h; thallium nitrate.	Lan and Lin, 2005
Water flea (*Daphnia magna*)	0.061	Mortality	LC50; 48 h; thallium chloride.	Lan and Lin, 2005
Water flea (*Daphnia magna*)	0.203	Mortality	LC50; 48 h; thallium acetate.	Lan and Lin, 2005
Water flea (*Daphnia magna*)	0.000016	Systemic	EC50 for immobilization; 24 h; thallium sulfate.	Lilius and others, 1995
Water flea (*Daphnia pulex*)	0.00000918	Systemic	EC50 for immobilization; 24 h; thallium sulfate.	Lilius and others, 1995

Table 8. Biological effects thresholds from empirical data related to the chemical toxicity of thallium.—Continued

[USACHPPM, U.S. Army Center for Health Promotion and Preventative Medicine; IC, inhibition concentration; LC, lethal concentration; EC, effect concentration; NOEC, no effect concentration; LOEC, lowest effect concentration; LOAEL, lowest adverse effect level. All data from water-borne exposures unless otherwise noted. Single dose exposure data from USACHPPM (2007) were excluded]

Receptor category and criteria species	Concentration (mg/L)	Effect	Comment	Reference
			Fish	
Goldfish (*Carassius auratus*)	7	Mortality	LC50; thallium chloride.	Birge, 1978
Bluegill (*Lepomis macrochirus*)	120	Mortality	LC50; 96 h; thallous sulfate; young of year.	Buccafusco and others, 1981
Bluegill (*Lepomis macrochirus*)	170	Mortality	LC50; thallium acetate.	Dawson and others, 1977
Rainbow trout (*Oncorhynchus mykiss*)	0.17	Mortality	LC50; thallium chloride.	Birge, 1978
Rainbow trout (*Oncorhynchus mykiss*)	0.18	Mortality	LC50; embryo-larval; thallium chloride.	Birge and others, 1980
Goldfish (*Carassius auratus*)	0.17	Mortality	LC50; egg-larval exposure.	Birge and others, 1977
Rainbow trout (*Oncorhynchus mykiss*)	1.5	Mortality	LC50; 10 d; thallous sulfate; fry.	Craig and Beggs, 1979
Rainbow trout (Oncorhynchus mykiss)	10–15	Mortality	Lethal concentration.	Nehring, 1962, cited in Craig and Beggs, 1979
Rainbow trout (*Oncorhynchus mykiss*)	0.24	Mortality	LC50; 10 d.	Zitko, 1975, cited in Craig and Beggs, 1979
Rainbow trout (*Oncorhynchus mykiss*)	2.3	Mortality	LC50; 96 h; thallium sulfate.	Horne and others, 1983
Fathead minnow (*Pimephales promelas*)	0.145	Mortality	NOEC in chronic tests; LOEC = 0.292; thallium sulfate; juveniles.	Kimball, 1978
Fathead minnow (*Pimephales promelas*)	0.04	Mortality	LOEC; thallium sulfate; embyros.	LeBlanc and Dean, 1984
			Amphibians	
Eastern narrow-mouthed toad (*Gastrophryne carolinensis*)	0.11	Mortality	LC50; thallium chloride.	Birge, 1978
Eastern narrow-mouthed toad (*Gastrophryne carolinensis*)	0.11	Mortality	LC50; egg-larval exposure.	Birge and others, 1977
			Mammals	
Rats, mice, dogs	20–40 mg/kg	Mortality	LD50s for various monovalent thallium salts.	USACHPPM, 2007
Mice	150 mg/kg	Mortality	LD50 for 36 h; thallium sulfate.	Achenback and others, 1980, cited in USACHPPM, 2007
Rats	1.51 mg/kg/d	Mortality	LOAEL; thallic oxide.	Downs and others, 1960, cited in USACHPPM, 2007
Rats	0.3 mg/kg/d	Reproduction	LOAEL for male reproduction; thallium sulfate.	USACHPPM, 2007
Rats	1.4 mg/kg/d	Systemic	LOAEL for nerve function 36 wks; thallium sulfate.	Manzo and others, 1983, cited in USACHPPM, 2007

Few studies focused on the toxicity of uranium to aquatic microorganisms have been published since the review by Driver (1994). Studies have reported microbe activity in soils and sediments related to uranium exposures, but threshold effects levels for mortality, growth, and reproduction are minimally reported in the literature. Most research has focused on establishing benchmark concentrations for aquatic bacteria exposed to uranium, although studies for freshwater algae and periphyton have also been published. For example, Small and others (2008) observed no adverse effects on periphyton communities exposed to river water having 0.1 mg/L of uranium. Water quality conditions, particularly pH, dissolved organic carbon, and differential chemical speciation, affect uranium toxicity to green algae (Franklin and others, 2000; Charles and others, 2002; Fortin and others, 2004, 2007; Hogan and others, 2005). The bioavailability of uranium to aquatic microorganisms and algae remains a critical issue for aquatic habitats potentially exposed to uranium releases from ore deposits. Guidance values to protect aquatic life (which includes algae and aquatic microorganisms) have been estimated by various organizations (table 6).

Aquatic Vascular Plants

Toxicity data for aquatic vascular plants are limited (table 7). The uptake and incorporation of uranium from water to plant tissues yield relatively low tissue residues (Pettersson and others, 1993). Translocation of uranium from root to foliage is low; therefore, foliage generally has lower uranium concentrations than roots (Pettersson and others, 1993). Mkandawire and others (2005, 2007) noted that water quality characteristics, such as phosphate concentration, affected the uranium toxicity to duckweed (*Lemna aequinoctialis*) under field conditions. Charles and others (2006) reported that joint exposures to copper and uranium reduced growth inhibition relative to single-compound exposures. Charles and others (2006) also reported a LOEC of 0.112 mg/L for growth. Sheppard and others (2005) have suggested a predicted no effect concentration for uranium toxicity to freshwater plants (table 6).

Aquatic Invertebrates

A moderate amount of chemical toxicity data is available for aquatic invertebrates including midges, cladocerans, hydrae, amphipods, worms, and molluscs (table 7). Similar to other metals, the toxicity of uranium varied with total hardness and alkalinity for aquatic invertebrates. For example, Poston and others (1984) reported greater mean acute values (as LC50s) for cladocerans in hard water (37.5 mg/L) than in soft water (6.4 mg/L) (table 7). Chronic values (as NOECs) were available for the water flea *Ceriodaphnia dubia* (<0.002–0.03 mg/L), the cladoceran *Moindaphnia macleayi* (0.008–0.031 mg/L), the midge *Chironomus tentans* (0.039 mg/L), and the mussel *Velesunio angasi* (0.280–0.388 mg/L), some of which are lower than guidance thresholds available for aquatic invertebrates (table 6). Uranium toxicity data for freshwater green hydra (*Hydra viridissima*) was also available from multiple studies (table 7).

Fish

A moderate amount of chemical toxicity data is available for fish (table 7). Uranium toxicity varies widely in fishes and is dependent on water quality conditions such as total hardness and alkalinity. For example, acute values (as LC50s) for fathead minnow were 3 mg/L with a water hardness of 210 mg $CaCO_3$/L and pH of 7.4, and 135 mg/L with a water hardness of 400 mg $CaCO_3$/L and pH of 8.2 (McKee and Wolf, 1963, cited in Driver, 1994). Some estimated guidance values to protect fish have also included hardness in their recommendations (table 6).

Toxicity data for uranium were available for threatened and endangered species of the Colorado River system. Hamilton (1995) examined the acute toxicity of uranium on swim-up fry and juvenile Colorado pikeminnow (*Ptychocheilus lucius*), razorback sucker (*Xyrauchen texanus*), and bonytail (*Gila elegans*). Acute values for were 46 mg/L (96-hr LC50s) for each species, which indicated uranium sensitivity did not differ between species. Common carp, a nonstatus species in the Colorado River system, were not affected by uranium exposures of 60 mg/L in areas of high water hardness (Till and Blaylock, 1976). Chronic values (as NOECs) for uranium toxicity were available for the gudgeon *Mogurnda mogurnda* (0.404 mg/L), the rainbowfish *Melanotaenia splendida inornata* (0.81 mg/L), and brook trout (*Salvelinus fontinalis*) (3.9 mg/L soft water and 16.4 mg/L hard water) (table 7).

Biological Receptors in the Terrestrial Food Chain

Soil Microorganisms

Although the soil crust community in arid ecosystems typical of the segregation areas are critical to the system's structure and function, existing uranium toxicity data for soil microorganisms and other soil-dwelling biota are limited (table 7). Biological soil crusts are assemblages of lichens, fungi, cyanobacteria, and mosses that colonize soil surfaces and represent up to 70 percent of the living groundcover in arid land environments (Belnap and Lange, 2001; Belnap and others, 2005). Biological soil crusts are critical to the transfer of nutrients from seasonal surface runoff (Ludwig and others, 1997, 2005). Empirical data related to the uranium toxicity for soil microbes was found in one study. Meyer and others (1998) indicated that an LOEC of 500 mg/kg dry soil for soil microbe respiration may also be applicable for some terrestrial systems. Characterization of effects cannot be developed for soil microbes in the absence of chemical toxicity data. Nevertheless, guidance values for soil microbes have been estimated (table 6).

Terrestrial Nonvascular and Vascular Plants

Chemical toxicity data for uranium were available for terrestrial plants (table 7). Uranium concentrations as low as 0.42 mg/L reduced seedling survival based on measures of

chlorosis, early leaf abscission, and reduction in root growth in hydroponically-grown soybean plants (*Glycine max*) (Murthy and others, 1984). Chronic values available for uranium concentrations in soil include a NOEC of >100 mg/kg for Scots pine (*Pinus sylvestris*) and a LOEC of 50 mg/kg dry soil for a variety of grasses (Sheppard and others, 1985; Meyer and others, 1998). Sheppard and Evenden (1992) suggested that sublethal effects may occur in plants grown in soils containing between 10 to 100 mg/kg of uranium. In this exposure range, reduced root growth in test plants was observed at soil concentrations of 10 mg/kg in both sand and peat soils, but aboveground growth was not affected (Sheppard and others, 1983). Several guidance thresholds for the protection of terrestrial plants have been estimated by various organizations (table 6).

Other factors should be considered when evaluating the toxicity of uranium to terrestrial plants. The mycorrhizae (fungus–plant root relationships) are critical to soil structure and function, and biological processes acting within the rhizosphere (the zone surrounding the roots of plants) complement the physicochemical factors influencing uranium solubility and control uranium uptake into plant tissues. Rufyikiri and others (2004) observed that mycorrhizal fungi may limit uranium accumulation by mycorrhizae-dependent plants exposed to high uranium concentrations in soil. Mechanisms explaining these observations are numerous, including the interactions of soil pH, calcium, and phosphorus. Such confounding factors encourage caution in interpretations of soil guidance values for metals in soils and threshold effects concentrations of uranium and other radionuclides.

Terrestrial Invertebrates

Chemical toxicity data for uranium were limited to earthworms and springtails for terrestrial invertebrates (table 7); guidance values specific to terrestrial invertebrates have not been estimated (table 6). Sheppard and Evenden (1992) observed reduced survival in earthworms (*Lumbricus* spp.) exposed to 1,000 mg/kg dry weight soil, a concentration associated with adverse effects to other organisms in different soil types. Following these earlier studies, Sheppard and others (2004, cited in Sheppard and others, 2005) evaluated the toxicity of uranium to the earthworm *Eisenia andrei* using three different soil types as exposure matrices and reported NOECs of 1,000 mg/kg dry weight soil for survival and reproduction in each soil type. Effects levels observed for two species of springtail (*Onychiurus folsomi* and *Folsomia candida*) were similar to those for earthworms following 35-day exposures (Sheppard and others, 2004). In a similar study, adult *O. folsomi* were more sensitive to uranium than *F. candida* as measured by mortality and reproduction (Sheppard and others, 2004, cited in Sheppard and others, 2005).

Alternative test methods have also been used to evaluate chemical toxicity to earthworms (table 7). Ribera and others (1996) exposed the earthworm *Eisenia fetida andrei* to filter paper media saturated with a uranium acetate salt solution and reported a 96-hour LC50 of 40 $\mu g/cm^2$, whereas Labrot and others (1999) using a similar test method reported a 96-hour LC50 of 13.5 $\mu g/cm^2$. Labrot and others (1996) also examined various biomarkers in *E. fetida andrei* exposed to a uranyl acetate solution and observed decreased levels of malondialdehyde at 1.0 $\mu g/cm^2$. These studies relied on exposures of individuals to filter paper saturated with uranium solutions and should not be compared directly with exposure on bulk soils. Regardless of that methodological difference, the findings of Labrot and others (1996, 1999) indicate that toxicity and bioconcentration may be mediated by uranium concentrations occurring in soil solution.

Amphibians and Reptiles

Amphibians and reptiles are key components of the food web in the segregation areas; however, data for the chemical toxicity of naturally occurring uranium in amphibians was minimal and no data was found for reptilian species (table 7). Decreased survival and reduced growth in survivors was observed in larvae of the frog *Rana perezi* exposed to 100 percent uranium mine effluent (uranium concentration, 1.75 mg/L), although effects could not be specifically associated with uranium (Marques and others, 2008). Marques and others (2008) also noted that other adverse effects related to growth and malformations were observed at effluent exposures >50 percent, but mortality was not significant relative to controls. Mitchell and others (2005) observed no effects in exposures of the African clawed frog (*Xenopus laevis*) to concentrations of depleted uranium greater than 77 mg/L in 96-hour tests, but delayed metamorphosis was observed at concentrations greater than 13.1 mg/L in 64-day exposures. However, no mortality or malformations were observed in these longer-term exposures (Mitchell and others, 2005). Kidney lesions developed in various frog species after the lymph sacs were injected with 15 mg/kg of uranium nitrate (Oliver and Smith, 1930). Guidance values for the protection of amphibians and reptiles have not been estimated (table 6).

Birds

Chemical toxicity data for effects of uranium in birds were not found, which is consistent with previous uranium reviews (Driver, 1994; Eisler, 1994). Despite the lack of empirical chemical toxicity data, a guidance value for uranium in drinking water has recommended for non-piscivorous birds in southern Utah (table 6) (U.S. Department of Energy, 2005).

Mammals

Chemical toxicity data for uranium in mammalian wildlife were not found; however, mammalian wildlife receptors and animal models used to evaluate human health share common pathways of uranium exposure. Laboratory mammals provide a starting point for evaluating effects levels in wild mammals, which undoubtedly have a range of sensitivities to uranium (table 7). Rabbits, dogs, and guinea pigs were more sensitive to uranium exposure than rats (Driver, 1994), and

rabbits were most sensitive to uranium in laboratory tests (Morrow and others, 1981, 1982, and Leach and others, 1984, all cited in Driver, 1994). As with other biological receptors, the increased solubility of uranium in U(VI) makes the hexavalent form more toxic to mammals than U(IV). More recent studies (for example, Domingo, 2001; Sheppard and others, 2005) indicate that mammalian toxicity values for uranium remain consistent with those of previous reviews (Driver, 1994; Eisler, 1994). Field observations (for example, tissue concentrations) related to uranium mining activities and releases of radionuclides from energy-generating facilities dominate the recent literature for mammals, particularly with respect to exposures and bioaccumulation. Most effects data available were associated with kidney function (table 7). Guidance values for the protection of mammals have been estimated by various organizations (table 6).

Bioconcentration Factors and Trophic Transfers

The bioavailability of uranium depends on its speciation in the environment. Metals, including uranium, partition between solid and liquid phases and may occur as dissolved, exchangeable, carbonate, iron-manganese oxide, organic, or crystalline species. Partitioning or speciation of metals in the environment is influenced to varying degrees by pH, redox state, organic content, and other environmental factors such as temperature, precipitation, and periodic events (for example, storms). Hydrogen ion activity (pH) is likely one of the more critical factors governing metal speciation, solubility from mineral surfaces, transport, and eventually bioavailability to plants or animals. Factors such as particulate size, mineral properties, and total surface area available for adsorption or desorption processes affect metal speciation and metal bioavailability. For example, finely milled ore may release smaller particles that are likely more widely dispersed by water and wind, and thus can enhance metal transport and availability to biological receptors. Therefore, the form of the uranium and the nature of the environment can strongly influence the transport (movement) and bioavailability of uranium and the uranium decay series products in the segregation areas. In particular, interception rates are strongly influenced by the amount of precipitation (Pröhl, 2009). Interception is the fractional rate of adsorption of radionuclides from atmospheric sources (both dry and wet deposition). Rates of interception are inversely proportional to precipitation on a logarithmic scale (Pröhl, 2009). The greatest rates of interception (that is, fractional adsorption to plants) occur in arid regions like the segregation areas. The transport of radionuclides into a plant is measured by a transfer factor (TF) that is specific to the plant (Leclerc and others, 2009) and in general is defined by the amount of radionuclide taken up into the edible portions relative to the amount on the foliar areas of the plant. Therefore, when environmental pathways are considered for transport of uranium to biological receptors, such coefficients must be evaluated separately. Databases of plant-specific TFs for uranium and other radionuclides have been developed (Leclerc and others, 2009) and will be important for quantitative modeling of exposure pathways of species of interest within the segregation areas.

The uptake of uranium and uranium decay series products into animals is similar to that of other metals. The uptake of metals can be regulated or nonregulated, depending on the size (atomic radius) and valence (charge) of the metal species (Gray and others, 2006; Nordberg and others, 2007). Metals that have a similar size and charge to essential trace metal nutrients can be taken up across biological membranes through specific transport mechanisms (for example, sodium/potassium exchange pumps). These same properties of atomic size and charge regulate binding once metals are inside an animal. These physical properties of the metals, along with the physiological characteristics of an animal, determine the internal dose and pharmacokinetics of metals, including uranium. In general, the liver and kidney are the primary sites of uranium accumulation, with bones, scales, gonads, gills, and gastrointestinal tract variously contributing to the accumulated uranium load (Colley and Thomson, 1991; Holdway, 1992). As with other non-essential metals, semi-metals, and non-essential metalloids, uranium tissue residues tend to be inversely related to body size. For example, higher mass-specific tissue residues are observed in small-bodied organisms provided uptake and depuration kinetics are similar across species of varying masses. Thus, target organ dose metrics of uranium and the decay series products will be determined by the speciation of each radionuclide and species-specific factors of the animals of interest.

Aquatic Ecosystem Exposure

Metal uptake by aquatic organisms is associated with two major pathways: (1) ingestion of metal-enriched sediment or particles and metal-enriched food items, and (2) uptake directly from water across biological membranes (generally respiratory membranes). Therefore, reactions of metals in water and sediment influence the bioavailability of metals in natural waters, and biological food webs influence the ultimate uptake of metals in aquatic ecosystems. However, detailed quantitative models that incorporate both biotic and abiotic factors which control uranium uptake in aquatic organisms are incompletely characterized. Therefore, empirical values for bioconcentration or bioaccumulation of uranium (and decay products) into aquatic organisms are used to derive estimates of exposure (Driver, 1994).

Linkages among aquatic and terrestrial habitats assure transfers across environmental compartments. Uranium in ore deposits accumulates in soils and reaches surface waters and sediments through physical processes mediated by natural or human-aided mechanisms, or the two combined. Uranium behaves similarly to other metals upon its entry in food

chains (Meyer and others, 2005); uranium is adsorbed onto the surfaces of plants and animals and can then be ingested by consumers and predators through their diets. Coincidental ingestion of suspended or bed sediments may also contribute significantly to uranium exposure for some species of aquatic vertebrates; therefore, bottom-feeding fishes tend to accumulate greater uranium concentrations than piscivorous fish (see Swanson, 1983, 1985, and Waite and others, 1988, all cited in Driver, 1994).

Uranium tends to adsorb to surfaces in sediment systems, and its interactions depend on the physicochemical properties of these solid matrices. Sediments act as a sink for uranium with concentrations consistently exceeding that in overlying water. Uptake of uranium from water to organisms occurs primarily through sources in sediment (Swanson, 1985, and Brunskill and Wilkinson, 1987, both cited in Driver, 1994) or through equilibriums established among sediment, water column, and fish. Early studies (Swanson, 1985) observed that organisms feeding on or near stream and lake sediments receiving drainage from uranium mill tailings contained higher concentrations of uranium than pelagic or predatory species. Transfer pathways and effects of uranium-series radionuclides in surface waters are poorly characterized quantitatively because there are few data available for uranium under field or laboratory conditions. Overall, sediment-to-fish transfer coefficients were 0.02–0.05 for sediment-to-fish and 5.7–11.0 for water-to-fish (Swanson, 1985). Driver (1994) postulated that a decline of about one order of magnitude in bioconcentration factors (BCFs) or bioaccumulation factors (BAFs) occurred at each step in an aquatic food chain and that biomagnification of uranium would not occur in aquatic or semi-aquatic food chains with species such as amphibians, fish-eating birds or waterfowl, and mammals such as muskrats (*Ondatra zibethicus*) or otters (*Lontra canadensis*).

Bioconcentration factors for uranium vary across aquatic species and were available for a limited number of species. For aquatic microbes, BCFs for uranium ranged from 2,794 to 354,200 (Driver, 1994). High BCFs for uranium by algae results from the relatively high adsorption of radionuclides on cell surfaces rather than actual uptake by the organisms (Atkins, 1977, and Horikoshi and others, 1981, both cited in Driver, 1994). For example, cell-bound uranium may account for only 10–15 percent of the total uranium on a dry weight basis in green algae and other aquatic microorganisms (Strandberg and others, 1981, cited in Driver, 1994). Physicochemical interactions among water quality factors and biological interactions can affect the binding of uranium to cell walls (for example, in phytoplankton) or cell membranes (for example, in zooplankton) and in solution.

The transfer of uranium in aquatic-sediment systems to rooted or floating aquatic vascular plants has been characterized in some species. A water-to-plant BCF of 0.55 was reported for uranium in aquatic macrophytes (Thompson and others, 1972, cited in Driver, 1994), which was lower than BCFs for the pondweed *Potamogeton* sp. (1.13) and the water milfoil *Myriophyllum* sp. (1.15) in a lake receiving uranium mine tailings (Waite and others, 1988, cited in Driver, 1994). Sediment-to-plant BCFs were lower for *Potamogeton* sp. (0.16) and *Myriophyllum* sp. (0.20) than water-to-plant BCFs (Waite and others, 1988). Water-to-invertebrate BCFs for uranium are highly variable, ranging from 1 to 10,000 (Thompson and others, 1972; Mahon, 1982; Driver, 1994), which reflects greater variation in trophic and spatial niches among invertebrates collected under field conditions (Swanson, 1985, cited in Driver, 1994).

Numerous field studies demonstrate that uranium concentrates in fish. Values derived from field-collected individuals reflect bioaccumulation factors (BAFs), which capture all potential routes of exposure beyond the uptake of material solely from ambient waters. Uranium BAFs in fish were less than 10 when exposed to contaminated surface waters (Thompson and others, 1972). The greatest reported BAFs for uranium in rainbow trout (*Oncorhynchus mykiss*), longnose suckers (*Catostomus catostomus*), and lake whitefish (*Coregonus clupeaformis*) were less than 38 (Mahon, 1982; Poston, 1982; Swanson, 1985, cited in Driver, 1994). BAFs reported in the literature for early life stage fish are consistently less than 20 (Driver, 1994; Yankovich, 2009). For example, BCFs for uranium ranged from 1.9 to 4.3 in hard water (210 mg/L as $CaCO_3$) in brook trout eggs and fry (Parkhurst and others, 1984) and 3.3 in eyed carp eggs (Till and Blaylock, 1976). Till and Blaylock (1976) also noted that uranium accumulated in yolk material more than in developing embryo. Driver (1994) suggested that in lieu of species or site-specific data, default values for concentration factors (BCFs or BAFs) should be 10 for the flesh of freshwater fish, 20 for whole body piscivorous or planktivorous fish, and 50 for benthic species that might have greater exposures through incidental ingestion of sediments.

Terrestrial Ecosystem Exposure

Partitioning of dissolved metals including uranium in soils is primarily affected by pH, and acidic soils are characterized by free metal ions in soil solution. Routine characterization of physicochemical properties of soils such as pH, cation exchange capacity, organic carbon, and soil texture (particle size distribution) are key attributes that govern uranium speciation. Uranium is poorly bioconcentrated in terrestrial plants through direct uptake because of the reduced bioavailability of insoluble compounds formed under naturally occurring conditions.

Factors that influence bioavailability of uranium in soil are similar to those physicochemical properties that influence its bioavailability in aquatic and sediment systems (International Atomic Energy Agency, 2009). For example, for terrestrial plants, uptake of uranium and other trace elements depends on (1) movement of elements from the soil to the

plant root, (2) movement of elements across the membrane of epidermal cells of the root, (3) transport of elements from the epidermal cells to the xylem, in which a solution of elements is transported from roots to shoots, and (4) possible mobilization, from leaves to storage tissues used as food (seeds, tubers, and fruit), in the phloem transport system (Sansharova and others, 2009a, cited in International Atomic Energy Agency, 2009). Metals including uranium are made available to terrestrial wildlife directly through plant uptake and the food chain, but the limiting step for elemental entry into the food chain is usually passage from the soil to the root, which depends on element concentrations in soil pore solutions. Climate strongly influences metal speciation, primarily because of climate's role in developing soil types. These local- and regional-scale factors ultimately control elemental—metals and metalloids—mobility and availability. For example, in the arid climates of the western United States, soils are commonly characterized by small abundances of soil organic matter and large abundances of salt and carbonate, the latter phases often containing metals in varying states of solution.

Uranium bioavailability in terrestrial food chains is strongly influenced by the nature of soils and their complex interactions with metals (Driver, 1994; Bohn and others, 2001). In general, uranium enters food chains via adsorption on plant surfaces (known as interception) or as tissue residues in prey. Natural history attributes related to feeding strategy can influence exposure of herbivores (Meyer and others, 2005). Coincidental ingestion of particles of sediment or soils also contributes to dietary exposures (Beyer and others, 1994). For plants, uranium exposures occur primarily in the rhizosphere where the soil directly interacts with the root (Ross, 1994). However, uranium uptake across root membranes is limited and little translocation to aboveground structures has been observed (Sheppard and others, 1983, and Van Netten and Morley, 1983, both cited in Driver, 1994; Vandenhove and others, 2009). Plant accumulations of uranium are limited, as reflected in low soil-to-plant concentration factors, while the greatest concentration ratios are associated with dusty conditions (Garten and others, 1987, cited in Driver, 1994). Concentration ratios for a variety of plants (food crops, pasture grass, fruits, and vegetables) were generally reported to be below 1 (Driver, 1994; Sansharova and others, 2009b, cited in International Atomic Energy Agency, 2009).

Bioaccumulation of uranium is also relatively low into herbivorous vertebrates, although multiple exposure routes (ingestion, inhalation, dermal contact, etc.) enhance the likelihood that uranium will be accumulated in tissues of exposed biota. Biomagnification of uranium does not occur through food-chain transfers, with transfer coefficients less than 1 from plants to foraging grazers in terrestrial environments (Driver, 1994). In addition, Mahon (1982) reported that vertebrate (bird and mammal) tissue concentrations of uranium in terrestrial food chains decreased by an order of magnitude at each trophic level. More information on soil-to-biota BCFs for uranium is available and should be consulted for detail (for example, Driver, 1994; see also Calmon and Fesenko, 2009).

Chemical Hazards of Radionuclides

Data on the chemical hazards of other radionuclides in the ^{238}U decay series are much more limited. Chemical toxicity data is available for thallium to terrestrial and aquatic receptors, but toxicity data for thorium, bismuth, radium, radon, polonium, and protactinium are limited. Brief characterizations of each element's toxicity and capacity for bioconcentration or bioaccumulation under field conditions are presented.

Thallium

Thallium is highly reactive, and concentrations range from 0.3 to 0.6 mg/kg in the environment. Elemental thallium occurs as relatively short-lived radionuclides, such as those encountered in the uranium decay series, and as stable isotopes, ^{203}Tl and ^{205}Tl. Stable isotopes of thallium occur as Tl(I) and Tl(III) in minerals (potassium and sulfur-containing ores), as thallium salts, and as alloys and amalgams with mercury (World Health Organization, 1996). Atmospheric releases of thallium from coal burning plants and the smelting of zinc, copper, and lead ores contribute to wet and (or) dry deposition from these industrial or mining sources. Thallium can also be released to surface waters through erosional processes during mining operations of other economically important minerals. Such releases of thallium through atmospheric deposition or erosional processes in undisturbed or disturbed source areas have the potential to contaminate surface waters, sediments, or soils, creating an exposure pathway to biological receptors (Peter and Viraraghavan, 2005).

Thallium in the aquatic environment exists in inorganic forms and as the stable organic dimethylthallium ($(CH_3)_2Tl^+$). Dimethylthallium is produced by oxidative methylation of Tl(I) in anaerobic freshwater sediment, but no measurement of dimethylthallium in the freshwater column has yet been made (Huber and Kirchmann, 1978). The mobility of thallium in soil varies as a function of its physiochemical properties (for example, oxidation state) and the nature and properties of the soils in which it occurs. The concentration of thallium in soil interstitial water may enable its uptake by terrestrial plants, and the release to groundwater may occur as a function of soil pH in soils where thallium is highly mobile. Thallium adsorbs to clays, organic matter, and iron oxides (World Health Organization, 1996). Deposited thallium may be resuspended as dust in terrestrial habitats or solubilized in aquatic habitats (Peter and Viraraghavan, 2005). The bioavailability of Tl(III) in freshwater is less than that of Tl(I) because of the oxidation of Tl(I) to Tl(III) by biota, despite Tl(III) being the dominant form of dissolved thallium in the water column of the Great Lakes (Canada/United States) (Ralph and Twiss, 2002; Twining and others, 2003). Potassium is important in determining the aquatic toxicity of thallium. Thallium and potassium are interchangeable in mineral crystal lattices and membrane transport because of similar atomic radii and ionic mobility (Hassler and others, 2007). Therefore, potassium has been used to help establish water quality guidelines and predicting the fate of thallium in the aquatic environment.

Chemical toxicity data for thallium are available (table 8), although much of the literature for terrestrial and aquatic biota reflects an emphasis on human health implications of environmental exposures (Agency for Toxic Substances and Disease Registry, 1992). Toxicity data for biota was limited to algae, aquatic microorganisms, aquatic vascular plants, aquatic invertebrates, fish, amphibians, and mammals; studies on the chemical toxicity of thallium were not available for cyanobacteria, soil microorganisms, terrestrial invertebrates, reptiles, or birds. The following sections briefly characterize toxicity data from the literature compilation. A wildlife toxicity assessment on the potential toxicity, bioconcentration, and bioaccumulation hazards associated with thallium exposure most often encountered in the environment is available (U.S. Army Center for Health Promotion and Preventive Medicine, 2007). In general, thallium bioaccumulates but does not biomagnify in terrestrial and aquatic food chains (Zitko and Carson, 1975, Zitko and others, 1975, Sharma and others, 1986, and Ewers, 1988, all cited in U.S. Army Center for Health Promotion and Preventive Medicine, 2007). Field experiments have measured thallium content of various food crops because terrestrial plants can absorb thallium from soil (Cataldo and Wildung, 1983, cited in Agency for Toxic Substances and Disease Registry, 1992), but limited uptake data are available. Characterization of food-chain transfer of the metal is lacking, despite observations that thallium is bioconcentrated (Bunzl and others, 2001). Bioconcentration and bioaccumulation is incompletely characterized for thallium, primarily because of insufficient data. The bioconcentration of thallium appears to be greater in aquatic vascular plants and aquatic macroinvertebrates than in fishes. For example, BCFs were 27–1,430 for Atlantic salmon (*Salmo salar*) (Zitko and others, 1975) and 34 for bluegill (*Lepomis macrochirus*) (Barrows and others, 1978). BCF values varied widely from 6,000 to 88,000 for duckweed (Kwan and Smith, 1988) and 5,500 to 26,000 for the amphipod *Hyalella azteca* (Borgmann and others, 1998).

Algae, Cyanobacteria, and Aquatic Microorganisms

Chemical toxicity data for thallium are limited for algae, and no toxicity data were found for cyanobacteria or aquatic microorganisms (table 8). Acute toxicity values (as EC50s) for algae ranged from 0.13 to 0.43 mg/L, whereas a chronic NOEC of 0.02 mg/L was reported for the green alga *Chlorella vulgaris* (De Jong, 1965). Acute toxicity data were also available for the freshwater rotifer *Brachionus calyciflorus* (EC50 =18.8 µg/L) and the fungus *Geotrichum candidum* (IC50 = 0.38 µg/L) (table 8). Guidance values to protect aquatic life have been estimated for thallium (table 6).

Aquatic Vascular Plants

Chemical toxicity data for thallium were limited to two studies for aquatic vascular plants (table 8). Chronic effects data (LOEC) were available for duckweed (*Lemna minor*) (Smith and Kwan, 1989). Despite the lack of empirical chemical toxicity data, a guidance threshold to protect aquatic plants has been estimated (table 6).

Aquatic Invertebrates

Chemical toxicity data for thallium were available for aquatic invertebrates including amphipods, shrimp, snails, daphnids, midges, nematodes, and roachflies (table 8). Chronic toxicity values (NOECs) have been reported for daphnids by Kimball (1978); they are lower than the guidance thresholds for daphnids proposed by Suter and Tsao (1996).

Fish

Chemical toxicity data for thallium were available for fish (table 8). Acute toxicity (as LC50s) ranged over three orders of magnitude for various fish species (table 8), and a chronic value (NOECs) has been reported for fathead minnows (Kimball, 1978). Guidance thresholds for chronic toxicity of thallium have been suggested for fish (table 6).

Terrestrial Vascular Plants

Thallium is not essential for plant growth, but terrestrial vascular plants will take up and translocate thallium to aboveground vegetation when soluble species are present in the soil. Toxic effects of thallium in terrestrial plants includes impaired chlorophyll synthesis, seed germination, reduced transpiration due to interference in stomatal processes, growth reduction, stunting of roots, and leaf chlorosis (Adriano, 1986). Kabata-Pendias and Pendias (1984) reported toxic effects on plants grown in a surface soil having 1.0 mg/kg thallium; this served as the basis for a guidance value developed by Efroymson and others (1997a, b) for thallium released to soils. Efroymson and others (1997a, b) noted that other studies focusing on thallium relied on hydroponic studies.

Amphibians and Reptiles

Chemical toxicity data for thallium were minimal for amphibians, and no toxicity data were found for reptiles (table 8). An acute toxicity value (as LC50) of 0.11 mg/L was reported for the eastern narrow-mouthed toad (*Gastrophryne carolinensis*) (Birge and others, 1977; Birge, 1978). Guidance thresholds for chronic toxicity of thallium have not been suggested for amphibians or reptiles (table 6).

Mammals

Toxicity data were available for laboratory animal studies related to human health but not wild mammals (table 8) (U.S. Army Center for Health Promotion and Preventive Medicine, 2007). Acute toxicity data (as LD50s) were consistently 20–150 mg/kg for thallium in rats, mice, and dogs (U.S. Army Center for Health Promotion and Preventive Medicine, 2007). Chronic dose rates (as lowest observed adverse effect levels (LOAELs)) for thallium in rats ranged from 0.3 to 1.51 mg/kg/d (U.S. Army Center for Health Promotion and Preventive Medicine, 2007). All of these data were exposures through drinking water; toxicity data for ingestion (soil or prey items) were not available. Guidance thresholds for chronic toxicity of thallium have not been estimated for mammals (table 6).

Thorium

Thorium generally occurs at concentrations three times greater than uranium (Zhang and Brady, 2002) as a rare earth phosphate (monazite) that is found in igneous rocks and placer deposits. Thorium also occurs as a relatively common silicate mineral thorite ($ThSiO_4$), which occurs in ore-grade deposits in North America (for example, in Idaho; see Mackin and Schmidt, 1957; Staatz, 1972) and as a trace constituent in phosphates (simple and multiple oxides; Gascoyne, 1992). The thorium isotope ^{232}Th has a very long half-life compared to other thorium isotopes, is more likely to be a radiation hazard, and contributes little to thorium occurrence in the lithosphere (Underhill, 1996; Zhang and Brady, 2002).

Releases of thorium to the atmosphere can occur from natural and anthropogenic sources, but pathways linking sources with biological receptors are lacking. Movement and partitioning of thorium in the environment, particularly from naturally occurring sources, involve ^{232}Th as particulates in the atmosphere that can subsequently reach terrestrial and aquatic habitats through wet and dry deposition (Jiang and Kuroda, 1987, cited in Agency for Toxic Substances and Disease Registry, 1990). Wet and dry deposition are the chief removal processes for atmospheric thorium, with deposition rates dependent on weather conditions and the physicochemical properties of particulates (for example, particle size and density, and chemical form; see Agency for Toxic Substances and Disease Registry, 1990). Natural hazards such as volcanic eruptions release thorium, resulting in increased concentrations in rain water (Fruchter and others, 1980; Kuroda and others, 1987, cited in Agency for Toxic Substances and Disease Registry, 1990). Thorium concentrations in soils vary regionally, with windblown dusts as likely natural sources of thorium in the atmosphere. From an exposure perspective, established background concentrations (Agency for Toxic Substances and Disease Registry, 1990) are applicable to biological receptors whether releases of thorium are linked to natural or anthropogenic (for example, uranium mining and ore processing) sources.

Thorium enters surface waters as ThO_2 in suspended particles or deposited into sediments because of low water solubility (Hem, 1992); colloids can dominate the concentrations of thorium in waters (Orlandini and others, 1990). Thorium concentrations in solution may be greater in waters with soluble complexes of carbonate, humic materials, or other ligands (Hem, 1992; LaFlamme and Murray, 1987). Dissolved thorium levels in most surface waters will be very low, but they will be higher in naturally occurring alkaline waters (Hem, 1992). In freshwater environments, thorium is relatively unavailable for biological uptake because it adsorbs strongly to inorganic sediments (Whicker and Schultz, 1982; Cowart and Burnett, 1994). Most environmental transport of thorium is through physical processes where thorium adheres to particulate matter; direct accumulation in aquatic plants is very low (approx. 1×10^{-3}; Pettersson and others, 1993). However, bottom-feeding aquatic organisms ingest thorium through inadvertent consumption of sediments while foraging (Whicker and Schultz, 1982). Although most thorium passes through the digestive tract, some is deposited in and strongly bound to bone, where it is removed only very slowly. Thorium BCFs (57.6–465) were available for rainbow trout (*Oncorhynchus mykiss*) (Poston, 1982). Poston (1982) also noted that the majority of thorium in fish was associated with the gastrointestinal tract, which may indicate that thorium incorporation into biological matrices was limited or predominately an adsorption process in the absence of assimilation. Based on very few studies, BCFs decrease as the trophic levels of aquatic animals increase (Poston, 1982; Fisher and others, 1987, cited in Agency for Toxic Substances and Disease Registry, 1990). Pyle and Clulow (1998) gives a BCF of 286–1,180 on white sucker (*Catostomus commersoni*).

Altered physicochemical characteristics of soils associated with uranium mining processes may be linked to thorium releases to surface water and groundwater (for example, acid-leaching of uranium tailing piles; see Moffett and Tellier, 1978, and Platford and Joshi, 1988, both cited in Agency for Toxic Substances and Disease Registry, 1990). The mobility of thorium in soils is determined by geochemical processes, and thorium will generally remain strongly adsorbed onto soil and be relatively immobile (Torstenfelt, 1986).

Thorium binds preferentially to donor atoms of oxygen and strong bases. The sorption of thorium on iron oxides is fast at low pH, which indicates the formation of strong complexes with surface sites (Murphy and others, 1999). Thorium also binds with the oxy-hydroxides, like goethite. Conversely, the presence of sulfates decreases adsorption by the means of competition (Syed, 1999). Thorium(IV) forms strong complexes with humic and fulvic acids (Olofsson and Allard, 1983). Adsorption with organic matter, clays, and oxides, which limits the mobility and bioavailability of thorium, increases with pH (Syed, 1999). The various isotopes of thorium do not have the same apparent behavior in soils because of their mode of genesis (primary radionuclide or daughter product), the differences in chemical solubility of the minerals which contain them, and their radioactive half-lives (Leslie and others, 1999). The mobility of the thorium isotopes goes in the direction $^{228}Th > ^{230}Th > ^{232}Th$.

As in aquatic systems, the presence of ions or ligands (for example, CO_3^{-2}, humic matter) will increase the formation of soluble complexes of thorium and increase its mobility in soil. Leaching into groundwater may be increased in soils having low sorption capacity and capacity to form soluble complexes (for example, hydroxylated forms). Plant-to-soil transfer ratios for thorium are consistently less than 0.01 (Garten, 1978; Vandenhove and others, 2009), indicating poor bioconcentration in plants from soil. Partitioning of thorium between soil matrix (for example, as an adsorbed fraction) and soil interstitial water also indicates that thorium BCFs in plants will be very low (Vandenhove and others, 2009). Plants grown in highly disturbed soils or waste materials (for example, uranium tailings containing elevated levels of thorium) will have increased BCFs (Ibrahim and Whicker, 1988).

Characterization of thorium exposure is relatively incomplete for human populations outside of occupational and hazardous waste site settings (Agency for Toxic Substances and Disease Registry, 1990), and data for exposure of biological receptors are even more limited. The following sections summarize the existing data related to the toxicity, bioconcentration, and bioaccumulation of thorium. Chemical toxicity data for thorium were not found for aquatic vascular plants, soil microorganisms, terrestrial plants, terrestrial invertebrates, amphibians, reptiles, or birds. However, recent publications indicate an increased focus on thorium's environmental fate relative to its availability to and incorporation into biological receptors exposed in the field (for example, Calmon and others, 2009).

Algae, Cyanobacteria, and Aquatic Microorganisms

Chemical toxicity data of thorium were limited to one study of freshwater algae. De Jong (1965) reported that the alga *Chlorella vulgaris* presented an NOEC for growth of 0.8 mg/L and an LOEC of 1.2 mg/L for thorium.

Aquatic Invertebrates and Fish

Thorium accumulates in the organs, skin, and gastrointestinal tract of fish, indicating that a significant portion of this radionuclide remains adsorbed (Poston, 1982). Chemical toxicity data of thorium in aquatic invertebrates and fish were limited. Acute toxicity of thorium to the amphipod *Hyalella azteca* varied as a function of water hardness in studies reported by Borgmann and others (2005); LC50s were 0.0052 mg/L in soft water (approx. 18 mg $CaCO_3$/L) and 3.15 mg/L in hard water (approx. 124 mg $CaCO_3$/L). In the catfish *Rhamdia quelen*, Borgmann and others (2005) reported that thorium exposure to 70–210 µg/L could stimulate enzyme activities related to oxidative stress in adults and was cytogenotoxic in juveniles. Data from field studies reported that releases of thorium from uranium mining and milling operations and radium and uranium recovery plants to surface waters have contributed to the exposure of benthic organisms (Hart and others, 1986, and McKee and others, 1987, both cited in Agency for Toxic Substances and Disease Registry, 1990).

Mammals

Thorium toxicity data for mammals is available for rodents from laboratory toxicity evaluations, but data for wild mammals are lacking. Exposure differences among laboratory studies using traditional biomedical test species yield pertinent data for human health (Agency for Toxic Substances and Disease Registry, 1990), but may be of limited use in identifying threshold effects levels for wild mammals. Nevertheless, these data identify pathways of concern for biota exposed in field settings. Thorium exposures to wild mammals may occur via inhalation, ingestion of food and water, and dermal/cutaneous uptake. Bone is the target organ for thorium; laboratory rodents deposit thorium into bone (70 percent), other organs and tissues (16 percent), urine (10 percent), liver (4 percent), and intestines (<0.05 percent) (Agency for Toxic Substances and Disease Registry, 1990). Thorium is excreted via renal mechanisms in mammalian species used in biomedical testing (National Research Council, 1988; Agency for Toxic Substances and Disease Registry, 1990), and that method would presumably apply to wild mammals as well. However, the natural history and life history attributes of wild mammals exposed in the field are different from laboratory rodents, which may affect exposure.

Bismuth

Bismuth is rare in the Earth's crust and is in the same elemental family as phosphorus, arsenic, and antimony. Bismuth is used as a cooling agent and fuel support in high-power nuclear plants. Bismuth is present at valences +3 and +5, with +3 being the most stable form. The isotopes of bismuth have atomic masses from 195 to 215, but only ^{209}Bi is stable. In the natural state, bismuth is in the form of bismuthine (Bi_2S_3), whose principal ores are associated with lead, silver, and tin (Li and Thornton, 1993). Bismuth is used as indicator of volcanic activity in connection with sulfide emissions because the contribution of anthropogenic sources (primarily mining extraction) to the total distribution of bismuth remains very limited (Ferrari and others, 2000). The use of fungicides and certain natural or synthetic manures that contain bismuth increase the content of this element in soils; however, the increase is negligible compared to the geochemical background of the majority of the cultivated soils (Senesi and others, 1979). Data relative to the behavior of bismuth in soils and plant transfer are very limited. In soils, bismuth is often oxidized and is found in the carbonate form. The element can accumulate in horizons rich in organic matter or oxidized iron (Kabata-Pendias, 2001), and its mobility can be affected by the pH.

Bismuth is one of the least toxic metals and is often used for the treatment of stomach ulcers and intestinal affections. Bismuth(III) is particularly used with a therapeutic aim as an antacid, astringent, disinfectant, antiprotozoaires, and radiocontrastant (Bi(V) mainly). However, it presents relatively greater radiation hazards. Bismuth is poorly characterized with respect to its environmental fate and effects on aquatic and terrestrial biota, although some data is available for maximum mineral tolerance in diets of domestic and laboratory animals (National Research Council, 2005). The National Research Council (2005) recommended maximum tolerable levels for bismuth of 500 mg/kg for rodents, swine, and horses and 1,000 mg/kg for poultry, based on animal health.

Data for the chemical toxicity of bismuth to biota were limited. De Jong (1965) reported chronic effects endpoints from laboratory exposures of bismuth to alga *Chlorella vulgaris*, which included an LOEC of 7.2 mg/L and a NOEC of 3.6 mg/L. Two studies regarding effect levels related to the

chemical toxicity of bismuth in aquatic invertebrates were identified. For the amphipod *Hyalella azteca*, the LC50s were 0.025 mg/L in soft water (approx. 18 mg $CaCO_3$/L) and 2.543 mg/L in hard water (approx. 124 mg $CaCO_3$/L) (Borgmann and others, 2005). Median effective concentrations (EC50s) for bismuth ranged from 0.662 to 14.79 mg/L for *Tubifex tubifex* (Khangarot, 1991). Chemical toxicity data for bismuth were not found for aquatic vascular plants, soil microorganisms, terrestrial plants, terrestrial invertebrates, fish, amphibians, reptiles, birds, or mammals. The effects of bismuth on terrestrial animals is not known, except for humans, for whom the toxic effects are not related to exposure time or dose (Martin and others, 1980, cited in Pamphlett and others, 2000). However, the increasing use of shotgun pellets containing bismuth (91 percent) for hunting may be a hazard to biota that survive after being shot. Pamphlett and others (2000) found significant amounts of bismuth in the cytoplasm within nervous system cells, the tubular cells of the kidneys, the dendritic cells of the liver, and the macrophages of the lungs of exposed mice in a laboratory study. However, the consequences of such a long-term accumulation over the life of the animal are not known (Pamphlett and others, 2000). Studies reporting bioconcentration and bioaccumulation were also not found for aquatic or terrestrial habitats. Additional investigations are necessary to better characterize the ecotoxicity of bismuth.

Radium

As a decay product of uranium and thorium, radium commonly occurs in all rock, soil, and water at very low concentrations. As an alkaline earth metal, radium behaves environmentally and physiologically like calcium (Whicker and Schultz, 1982; Cowart and Burnett, 1994). Environmental migration of radium is facilitated by its ability to form soluble sulfates, carbonates, and chlorides. Radium readily deposits in bone tissue once taken up by aquatic organisms. Concentrations of radium in the groundwater are typically elevated when high concentrations occur in bedrock (Selinus and others, 2005). All isotopes of radium (^{224}Ra, ^{226}Ra, ^{228}Ra) are radioactive. The most common isotope, ^{226}Ra, is an alpha emitter with accompanying gamma radiation; ^{228}Ra is principally a beta emitter and ^{224}Ra is an alpha emitter. Radium decays to form radioactive radon gas isotopes, which are not chemically reactive. Radiation hazards dominate exposure in field settings, and radon chemical hazards are limited.

The ecotoxicology of radium is incompletely characterized, particularly with respect to its chemical toxicity. Data are not sufficient to characterize threshold effects levels for the chemical toxicity of radium to biota, but radiation hazards likely outweigh concerns for chemical toxicity of radium. Routes of exposure to terrestrial vertebrates are dominated by inhalation, ingestion, and dermal or cutaneous exposures (Agency for Toxic Substances and Disease Registry, 1990). Radium from inhalation or ingestion is partitioned into fecal material (80 percent) and the gastrointestinal tract (20 percent), where radium enters the bloodstream and preferentially accumulates in bone tissues (Agency for Toxic Substances and Disease Registry, 1990). This fraction of the radium dose may be excreted through the feces and urine in time, but a portion will remain in the bones as part of the tissue matrix (Agency for Toxic Substances and Disease Registry, 1990; Casarett and others, 2007).

Chemical toxicity data for radium were not found for algae, cyanobacteria, aquatic microorganisms, aquatic vascular plants, aquatic invertebrates, soil microorganisms, terrestrial plants, terrestrial invertebrates, fish, amphibians, reptiles, birds, or mammals. As noted above, nearly all naturally occurring radium is present as ^{226}Ra, which occurs in plants, animals, soil, rocks, surface water, and groundwater. Radium generally occurs at very low concentrations in plants and animals. Higher concentrations of radium in plants are associated with uranium ores and other geologic materials, with estimates of 3 percent of that in soil (Rayno, 1983; Tracy and others, 1983; Watson and others, 1984). Plant BCFs for radium across all plants species and soil types have been summarized by Vandenhove and others (2009). Transfer coefficients summarized for feedstuff-to-domestic animals could potentially be used to evaluate dietary exposures in wild mammals, particularly for herbivores and for preliminary estimates for maternal transfer of nuclides based on milk consumption in offspring (Howard and others, 2009).

Radon

Radon is a noble gas that has limited data available to characterize its ecotoxicity (Cothern and Smith, 1988; Vincolli, 1996). Radon naturally occurs as a gas and is highly mobile in the earth's crust (Corbett and others, 1997); its presence in the atmosphere results from the transfer of the gas from near-surface soils and rocks to the surface. Numerous isotopes occur, but ^{220}Rn and ^{222}Rn are the most common. The geochemistry of nuclides in the natural decay series and links to release of radon are complex. For example, uranium concentrations vary with rock type, which affects the transport and emission of the gas. Highly permeable rocks, such as limestone, or rocks that are fractured or faulted provide more spaces for radon gas to pass through the material; radon may be released at the surface if these rock types occur within breccia pipes. Furthermore, radon dissolves in water and may be transported through permeable and fractured rocks for long distances, where the gas may be released when groundwater reaches the surface. Radon readily enters the gas phase because of its low vapor pressure (Cothern, 1988). Therefore, radon does not persist in the water and has an aqueous half-life of 2 days (Vincolli, 1996). Given this brief summary of the environmental fate and transfer of radon, the primary hazards associated with exposures to ecological receptors in the field will be radiation toxicity, primarily via inhalation routes of exposure.

Polonium

Polonium occurs in the earth's crust at about one part in 10^{15}. All 25 isotopes are radioactive, but only three have appreciable half-lives (>140 days: ^{208}Po, ^{209}Po, ^{210}Po). From an ecological perspective, polonium occurs naturally at concentrations that preclude chemical toxicity as a primary hazard. The isotope ^{210}Po is produced during the decay of ^{238}U, is widely distributed in small amounts in the earth's crust, and occurs in uranium ores at less than 0.1 mg ^{210}Po per ton. Radiation hazards cannot be dismissed given the specific activities of the three most frequently encountered isotopes.

In terrestrial ecosystems, foliar transfer is the dominant pathway for contamination of vegetation by ^{210}Po (Francis and others, 1968; Pietrzak-Flis and Skowronska-Smolak, 1995; Skwarzec and others, 2001). The relative weakness of root transfer of ^{210}Po (because of its strong retention by soils), as well as an almost negligible translocation among plant tissues, results in ^{210}Po mainly being concentrated in the leaves of plants. For animals, specific activities of ^{210}Po vary by four orders of magnitude, depending on the species selected and organs examined. For example, ^{210}Po concentrations range from 0.037 Bq/kg wet weight for ox muscle (Globel and Muth, 1980, cited in Beaugelin-Seiller and others, 2004) to 332 Bq/kg for caribou liver (Thomas and Gates, 1999). This broad interval is related to the quantity of ^{210}Po ingested by the animal (including incidental ingestion of soil particles) and to the lifespan of the animal. In addition, ^{210}Po transfer to animals is proportionally weaker at sites close to uranium mines than at undisturbed sites because the larger particle sizes associated with mining activities reduce the ^{210}Po bioavailability for plants and animals (Thomas, 2000).

In lakes, the specific activities of ^{210}Po are much higher in sediments than in water (Haridasan and others, 2001). The specific activities measured in aquatic plants are relatively high, particularly in phytoplankton (approximately 20 Bq/kg wet weight) with concentration factors of approximately 10^3 to 10^4 (Hameed and others, 1997; Shaheed and others, 1997, cited in Beaugelin-Seiller and others, 2004). In aquatic animals, specific activity of ^{210}Po is more important in invertebrates than fish. Polonium concentrates in the soft tissues of molluscs. Moreover, the carapaces (chitinous) of shellfish accumulate more ^{210}Po than the shells (calcium carbonates) of bivalves because of a strong affinity of ^{210}Po to organic matter (Cherry and Heyraud, 1981). In fish, soft tissues in contact with the digestive system and the gills have an activity of ^{210}Po that is greater than the muscles and skin. Fish incorporate ^{210}Po from ingestion of food and filtration of water by the gills. The concentration factors in molluscs and fish are very high, from 1 to 1,000 L/kg wet weight.

Protactinium

Protactinium is a radioactive metal that does not readily oxidize when exposed to air. Three isotopes (^{231}Pa, ^{234}Pa, ^{234m}Pa) naturally occur, but ^{231}Pa is the most abundant. As a decay product of ^{235}U, protactinium is naturally present in soil, rocks, surface water, groundwater, plants, and animals in very low concentrations. Greater concentrations are present in uranium ores and other geologic materials. Protactinium occurs in uranium ores at a concentration of about 1 part protactinium to 3 million parts uranium. Protactinium preferentially adsorbs to soil, with concentrations in sandy soil particles 550 times greater than in interstitial water; concentration ratios are even higher (2,000 and above) for loam and clay soils. Protactinium is generally not a concern for groundwater. Few data specific to protactinium behavior in terrestrial ecosystems are available. The rare data that does exist indicate that protactinium has low mobility in soils and low transfers in the food chains of terrestrial origin (see Colle and Mourlon, 2003).

These radionuclides present little chemical hazards based on their short half-lives but may contribute significantly to radiation hazards for ecological receptors exposed in the field. For example, Thomas and Liber (2001) showed that external beta radiation from ^{234}Pa and alpha radiation from uranium contributed most of the dose at the affected sites, whereas ^{210}Po was most important at the control site in a study on benthic invertebrates. Polonium (Po) and protactinium (Pa) present the greatest hazards when ingested or inhaled. For example, studies have demonstrated that ^{210}Po is accumulated to exceptionally high levels in tissues of a variety of marine organisms, well above levels of the parent radionuclide ^{210}Pb (Carvalho and Fowler, 1994; Stepnowski and Skwarzec, 2000). The behavior of ^{210}Po differs from that of ^{210}Pb, especially because of the higher affinity of ^{210}Po for organic matter. The hazards are mediated by adverse effects linked to internal dose of radiation. External radiation dose may be associated with gamma rays emitted by ^{231}Po and a number of short-lived decay products of ^{227}Ac (actinium).

Radiation Hazards of Radionuclides

Biological receptors may be exposed to radiation from undisturbed and disturbed naturally occurring radioactive materials. Undisturbed natural radiation sources have not been manipulated through human activities such as mineral extraction. Disturbed natural radiation sources have been modified through human interventions and include mining, collateral events associated with the use of fossil fuels, production and use of fertilizers (for example, phosphate fertilizers), and use of natural material for construction activities (for example, granite countertops). Radiation exposure can also occur through the release of technologically enhanced, naturally occurring radioactive materials or refined sources associated with nuclear fuel cycles, but natural radiation remains a significant contributor to radiation dose in the environment (United Nations Scientific Committee on the Effects of Atomic Radiation, 2000). A person in the United States is estimated to receive a natural background radiation dose of approximately 360 mrem per year (3.6 mGy/yr) (United Nations Scientific Committee on the Effects of Atomic Radiation, 2000). Naturally occurring radionuclides, including ^{238}U and its daughter

products, contribute 63 percent of the overall total dose received; nearly 50 percent of that dose is from inhaled radionuclides such as radon. The remaining 37 percent comes from exposure to cosmic radiation and radionuclides internally present within the human body (United Nations Scientific Committee on the Effects of Atomic Radiation, 2000). The background radiation dose for nonhuman biota will differ because life history and feeding strategies can influence exposure.

Ionizing radiation is produced when naturally occurring radioactive materials decay—the nuclei of unstable atoms or radionuclides release energy in the form of radioactivity to increase their nuclear stability. This process is characterized by emissions of subatomic particles and high-energy photons (gamma rays). Radioactive decay of uranium produces daughter products such as radium, radon, and thorium and releases alpha particles, beta particles, or gamma rays. All daughters and radiation emissions are naturally occurring in the ^{238}U decay process. Emissions of the ^{238}U decay series (alpha and beta particles and gamma rays) are the primary focus of this overview of radiation effects.

Collisions between ionizing radiation and molecules in cells and tissues of exposed organisms can cause adverse effects to biological receptors in field and laboratory settings. An alpha (α) particle has a relatively short range in air, generally no more than several centimeters (and approximately several micrometers in water), and does not penetrate deeply into biological structures such as the epidermis or cuticle of biota. Beta (β) particles can travel up to one meter in the air and can penetrate several centimeters into biological tissues. Gamma (γ) rays can travel relatively long distances through air, environmental materials, and biological tissues. These emission species are highly energetic and have the capacity to pass through biota; consequently, they pose a great risk to biota.

Gross alpha and beta radiation are measures of all alpha and beta activity, respectively, present in a sample regardless of the specific radionuclide source. As an assessment and monitoring tool, gross alpha and beta are often used to screen samples for relative levels of radioactivity (Underhill, 1996; Cooper and others, 2003). Measurements of exposures to natural radiation associated with elemental uranium, which emits alpha particles, is more problematic for pathways contributing to internal dose (for example, ingestion and inhalation in terrestrial vertebrates) than for those contributing to external dose. Adverse effects of internal alpha emission in biota could be significant (Blaylock and others, 1993; United Nations Scientific Committee on the Effects of Atomic Radiation, 1996; Sample and others, 1997). Effects of chemical toxicity and radiation toxicity may be jointly expressed in field settings.

Exposure to ionizing radiation may lead to adverse biological effects. Alpha particles released during radionuclide decay can cause adverse effects during radiation exposures through ingestion or inhalation in animals or uptake and translocation in plants (Sample and others, 1997). Early developmental stages or life stages with rapid growth are generally more sensitive to radiation exposure than older, relatively mature organisms of the same species. Embryos and fetuses are typically more sensitive to ionizing radiation because these early life stages are dominated by rapidly dividing cells (Huettermann and Koehnlein, 1978; Riley, 1994; Brenner and others, 2003). Cells undergoing division though mitosis are more susceptible than cells that are not proliferating, and damage to the cellular DNA often results in cell death (Wolff, 1998; Shackelford and others, 1999; Pawlik and Keyomarsi, 2004). The sensitivity of biota to radiation and chemical exposures is also influenced by body size. For example, large-bodied species are typically more vulnerable to high levels of radiation exposure than small-bodied species because of the greater collision potential (for example, larger target) between the ionizing radiation and biota (Bytwerk, 2006; Higley and Bytwerk, 2007). A species life history may also affect its sensitivity to radiation. Alonzo and others (2008) reported that fast growing invertebrates were more sensitive to reproductive effects than slow growing invertebrates. Life history strategies may not account for all differences in species sensitivity to radiation exposure, but they should be considered when characterizing pathways that link sources to receptors. Invertebrates (for example, herbivorous insects or filter-feeding aquatic invertebrates), vascular plants, unicellular plants and animals, bacteria, and viruses are generally more resistant to the acute effects of radiation (Whicker and Schultz, 1982). However, smaller burrowing mammals will tend to receive larger doses through external exposures to contaminated soils containing radionuclides because of their close and prolonged contact with the soil and their occupation of confined spaces such as burrow habitats wherein soil gases such as radon may accumulate (Macdonald and Laverock, 1998). Burrowing animals have the highest potential for external exposures because they are exposed to a 4-pi geometry (that is, 360 degrees) versus a 2-pi geometry (180 degrees) for animals that only nest or sleep on the soil surface. Fish and other aquatic organisms are typically less sensitive to radiation exposure than terrestrial wildlife (vertebrates). This radioresistance could be due to a smaller DNA content per cell as well as to longer and more variable cell cycle times as compared to mammals. Differences in exposure matrix (atmospheric versus aquatic) also influence dose, given the relative differences in radiation penetration in these media (Martin, 2006).

Exposures to high levels of ionizing radiation produce adverse biological effects, such as increased cell death, decreased life expectancy, reduced growth, and altered behavior. Much of the literature focused on adverse biological effects is related to external gamma acute exposure and associated observed effects (see reviews by Driver, 1994; Eisler, 1994) rather than naturally occurring radioactive materials such as those of the uranium decay series. Previous studies provide information on the biological effects associated with uranium mining operations, altered routes of exposure to radiation from decay series elements, and biological effects thresholds (U.S. Environmental Protection Agency, 2005, 2008). Species sensitivity to radiation is highly variable (Rose, 1992), and beneficial responses have been observed in

biota commonly used in evaluating human health effects and radiation safety (for example, see Luckey, 1991, regarding hormesis associated with exposures to low levels of radiation). Single-celled organisms and invertebrates tend to be more resistant to acute radiation exposures than vertebrates (fig. 9) (Eisler, 1994). Extrinsic factors (type of radiation and its associated energy, rate and length of exposure, and dose rate and absorbed dose) will be influenced by environmental factors such as spatial characteristics of exposure, physical stressors (for example, temperature and season), and chemicals occurring as part of the exposure matrix (Cooper and others, 2003). Biological responses to radiation exposure will also be affected by intrinsic factors such as species, age, sex, nutritional status, and biological and ecological interactions (Alpen, 1997; Van der Stricht and Kirchmann, 2001).

Literature on the environmental effects of radiation characteristic of the uranium decay series for microbial, plant, and animal species were reviewed and summarized (for example, see International Atomic Energy Agency, 1976, 1988, 1992; Woodhead, 1984; National Council on Radiation Protection and Measurements, 1991; Driver, 1994; Eisler, 1994; United Nations Scientific Committee on the Effects of Atomic Radiation, 1996; Thompson and Bird, 2003; Woodhead and Zinger, 2003; Andersson and others, 2009). Data available for the effects of ionizing radiation is primarily related to nuclear energy and weapons production rather than naturally occurring radionuclides and mining activities. Nevertheless, radiation effects associated with gross alpha, gross beta, or gamma rays are highly dose-dependent regardless of source. Radiation data, including toxicity and fate data, for species in the segregation areas were not found; therefore, data from other species reported in the literature are presented. Factors influencing uptake, loss, and toxicity of radionuclides in the uranium decay series are considered for aquatic and terrestrial biota. The biota's potential for being an emission source through bioconcentration or bioaccumulation is also considered. Dose and dose rates used to characterize radiation toxicity are detailed in appendix 2.

A brief summary of the transfer coefficients of these radiohazards from water or soil to biota are also presented. Physicochemical characteristics of water-column or water-sediment interface or sediments alone influence radionuclide uptake in fish, aquatic invertebrates, and sediment-dwelling invertebrates. For example, radiation uptake by fishes and aquatic invertebrates in water-column dominated exposures is dependent on the ionic chemical species, which interacts with external organs (for example, gills and skin) that provide surfaces for exposure or influences bioaccumulation via dietary exposures. In addition, radionuclide accumulation in biota can differ within organs (for example, preferential partitioning) and by feeding habits.

Biota are potentially exposed to external radiation through radionuclides in parent ore material, water, sediment, soil, and other biota such as vegetation. Sources of internal

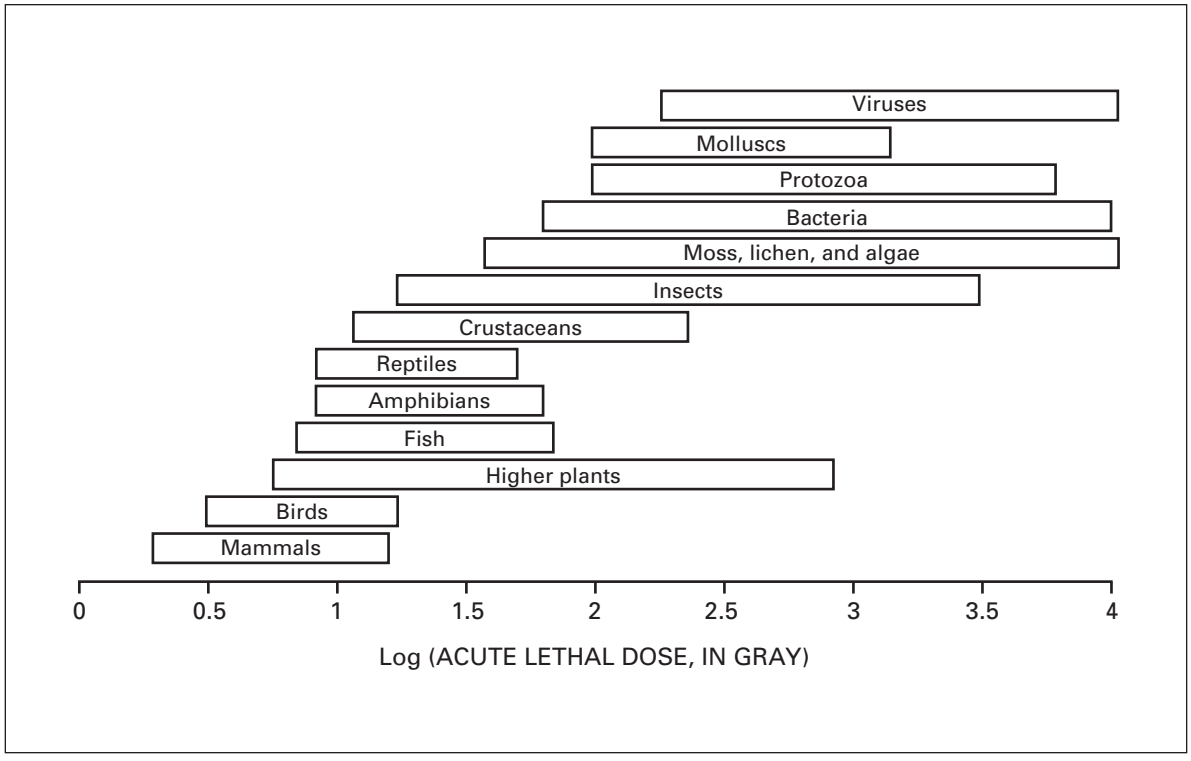

Figure 9. Comparative radiosensitivity of different organisms represented by the acute lethal dose ranges (Woodhead and Zinger, 2003; United Nations Scientific Committee on the Effects of Atomic Radiation, 1996, after Whicker and Schultz, 1982).

radiation exposure from radionuclides include ingestion via food and water and absorption through the epithelium of the skin or respiratory tissues. Effects thresholds from empirical data and consensus-based guidance values for biota associated with radiation exposures from published literature are presented in the following sections by receptor category. The majority of the toxicity data was from reviews by Driver (1994), Eisler (1994), and Woodhead and Zinger (2003). In the most recent review, Woodhead and Zinger (2003) used a database (FREDERICA) to identify publications containing effects information sufficient to estimate the dose rate for chronic exposures; however, most of these publications provided data for acute rather than chronic radiation exposures. Data from Woodhead and Zinger (2003) was expansive and complex; therefore, data from FREDERICA are presented only in the text to minimize the length and complexity of table 9. Woodhead and Zinger (2003) was the primary reference used to draw radiation toxicity data in the following sections. Woodhead and Zinger (2003) should be reviewed for additional risk assessment for radiological effects. In addition, the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) is developing a document that summarizes the effects of ionizing radiation on nonhuman biota that would be useful to consult for risk assessment purposes (Tom Hinton, French Institute of Radiation Protection and Nuclear Safety, personal commun., 2009; see also *www.unscear.org*, accessed December 1, 2009).

Biological Receptors in the Aquatic Food Chain

Algae, Cyanobacteria, and Aquatic Vascular Plants

Empirical data for radiation toxicity to algae, cyanobacteria, and aquatic plants were limited (table 9) (Woodhead and Zinger, 2003). References that describe morphological changes or survival of green algae indicate high acute doses ranging from 100 to 1,000 Gy (Woodhead and Zinger, 2003). Data on low doses and chronic irradiation experiments are limited, although low doses (1–5 Gy) resulted in chromosome aberrations for green algae. Woodhead and Zinger (2003) observed that acutely toxic doses for cyanobacteria and green algae ranged from 1 to several thousand Gy, with most threshold effects occurring at doses greater than 5 Gy. Woodhead and Zinger (2003) reported that aquatic plants were less radiosensitive to radiation exposure than higher trophic levels, although this observation was limited to five references. Dose rates of 1.3–8.5 μGy/h inhibiting growth in a cyanobacteria (*Syenechococcus lividus*) have been reported (Woodhead and Zinger, 2003). Guidance values for the protection of algae and aquatic plants have been estimated by various organizations (table 6).

Aquatic Invertebrates

Radionuclides accumulate in the gut of filter-feeding aquatic and sediment-dwelling invertebrates. Data for radiation effects in aquatic invertebrates are available (table 9) (Woodhead and Zinger, 2003). Hormetic effects have been noted; ionizing radiation increased growth rates at dose rates of 760 mGy/d for blue crab (*Callinectes sapidus*), 2,400–5,500 mGy/d for water snail (*Assiminea infima*), and 8,200–17,800 mGy/d for daphnids (table 9). Historically, data for ionizing radiation effects on crustaceans and molluscs focused on acute exposures with low linear energy transfer radiation (for example, x-rays and gamma rays; Woodhead and Zinger, 2003). Significant effects on invertebrate reproductive capacity have been observed for chronic irradiation at dose rates as low as 0.19 mGy/h (Woodhead and Zinger, 2003). More recently, Alonzo and others (2008) reported that exposures to dose rates of 0.11 mGy/h or higher resulted in a significant reduction in body mass and egg and neonate masses. Gilbin and others (2008) reported lower fecundity and decreases in egg mass for cladoceran *Daphnia magna* at 4.2 and 31 mGy/h, respectively. Overall, molluscs were less radiosensitive than crustaceans with expected chronic dose rates greater than 10 mGy/h (Woodhead and Zinger, 2003). Guidance values for the protection of aquatic invertebrates have been estimated by various organizations (table 6).

Fish

The literature for radiation effects on fishes is relatively well developed (table 9) (Woodhead and Zinger, 2003). More data are available for acute radiation exposures than for chronic irradiation in bony fish. For both irradiation durations, reproductive capacity was the most frequently studied endpoint, and eggs were identified as the most radiosensitive life stage, but it was noted that the most recent acute study was from 1999 (Woodhead and Zinger, 2003). The majority of acute irradiation data are for freshwater species such as medaka (*Oryzias latipes*) and salmonids (*Oncorhynchus mykiss, O. kisutch*, and *O. tshawytscsha*) (Woodhead and Zinger, 2003). Woodhead and Zinger (2003) reported that acute doses less than 1 Gy are unlikely to cause morbidity in irradiated fish, but acute exposure can affect reproduction by altering spermatogenesis, oogenesis, and embryo development. Life stage considerations were also found to be important. Woodhead and Zinger (2003) noted that acute exposures of 0.16 Gy in the single-cell stage of development could affect mortality, but radiation doses less than 0.5 Gy in later life stages would not likely affect reproduction in adults. Chronic endpoints commonly included growth data as body weight and length and reproductive effects as fertility of the irradiated parents, the viability of the resulting embryos, embryo development, and the fertility of offspring irradiated as developing embryos (Woodhead and Zinger, 2003). Woodhead and Zinger (2003) reported that chronic dose rates less than 12 mGy/h during embryonic development or less than 4 mGy/h during post-hatch life were unlikely to affect survival based on irradiation of external sources. Protective guidance values derived for fish by various organizations range over several orders of magnitude (table 6). Literature reviews to establish guidance values of ionizing radiation exposure must be used with caution because biological effects differ from one form of radiation to another.

Table 9. Biological effects thresholds from empirical data related to radiation toxicity.

[LD, lethal dose]

Receptor category and species	Radiation	Dose or dose/ rate	Effect	Comment**	Reference
Aquatic invertebrates					
Daphnid (*Daphnia magna*)	Alpha	15 mGy/h	Growth	Model predicted delayed population growth	Alonzo and others, 2008
Earthworm (*Eisenia fetida*)	Gamma	43 mGy/h	Mortality	Model predicted population extinction	Alonzo and others, 2008
Blue crab (*Callinectes sapidus*)	Ionizing	760 mGy/d	Growth	Increased growth rate	Eisler, 1994
Water snail (*Physa heterostropha*)	Ionizing	2400–5500 mGy/d	Growth	Increased growth rate	Eisler, 1994
Daphnid (*Daphnia pulex*)	Ionizing	8200–17,800 mGy/d	Growth	Increased growth rate	Eisler, 1994
Fish					
Goldfish (*Carassius auratus*)	Gamma	23,000 mGy	Mortality	LD50/30	Driver, 1994
Pinfish (*Lagodon rhomboides*)	Ionizing	197 mGy/d	Growth	Increased growth rate	Eisler, 1994
Soil microorganisms					
Bacteria	Gamma	5,000,000 mGy	Mortality	LD50/30	Driver, 1994
Virus	Gamma	18,000,000 mGy	Mortality	LD50/30	Driver, 1994
Terrestrial nonvascular and vascular plants					
Bean (*Vicia faba*)	Ionizing	58,000–100,000 mGy/yr	Morbidity	Growth stimulation	Eisler, 1994
Herbaceous rock-outcrop community	Ionizing	1,000,000 mGy/yr	Mortality	Lethal; severe sublethal effects at 400,000 mGy/yr	Eisler, 1994
Crop land, abandoned	Ionizing	1,500,000 mGy/yr	Mortality	Lethal; severe sublethal effects at 450,000 mGy/yr	Eisler, 1994
Terrestrial invertebrates					
Fruit fly (*Drosophila melanogaster*)	Gamma	1,000,000 mGy	Mortality	LD50/30	Driver, 1994
Amphibians and reptiles					
Frog	Gamma	7,000 mGy	Mortality	LD50/30	Driver, 1994
Leopard lizard (*Crotaphytus wislizenii*)	Ionizing	40–60 mGy/d	Reproduction	No female reproduction in years 3 and 4. Sterility in males and regression of ovaries in females in year 5. Year 6, 75% of females lacked ovaries and 25% had normal ovaries with signs of egg deposition; males appeared normal.	Eisler, 1994
Birds					
Linnet (*Acantis cannabina*)	Ionizing	4000 mGy	Mortality	LD50/30; adult	Kushniruk, 1964, cited in Driver, 1994
Green-winged teal (*Anas crecca*)	Ionizing	4850 mGy	Mortality	LD50/30; adult	Tester and others, 1968, cited in Driver, 1994
Bluebird (*Sialia sialis*)	Ionizing	5000–6000 mGy	Mortality	LD50/30; nestling to fledgling	Willard, 1963, cited in Driver, 1994

Table 9. Biological effects thresholds from empirical data related to radiation toxicity.—Continued

[LD, lethal dose]

Receptor category and species	Radiation	Dose or dose/ rate	Effect	Comment**	Reference
			Birds—Continued		
Greenfinch (*Chloris chloris*)	Ionizing	6000 mGy	Mortality	LD50/30; adult	Kushniruk, 1964, cited in Driver, 1994
European Goldfinch (*Carduelis carduelis*)	Ionizing	6000 mGy	Mortality	LD50/30; adult	Kushniruk, 1964, cited in Driver, 1994
House sparrow (*Passer domesticus*)	Ionizing	6250 mGy	Mortality	LD50/30; adult	Kushniruk, 1964, cited in Driver, 1994
Mallard (*Anas platyrhynchos*)	Ionizing	6300 mGy	Mortality	LD50/30; 12 months old	Curnow and others, 1970, cited in Driver, 1994
Serin (*Serinus canarius*)	Ionizing	5000 mGy	Mortality	LD50/30; adult	Kushniruk, 1964, cited in Driver, 1994
Mallard (*Anas platyrhynchos*)	Ionizing	7040 mGy	Mortality	LD50/30; 4 months old	Abraham, 1972, cited in Driver, 1994
Blue-winged teal (*Anas discors*)	Ionizing	7150 mGy	Mortality	LD50/30; adult	Tester and others, 1968, cited in Driver, 1994
Starling (*Sturnis vulgaris*)	Ionizing	8000 mGy	Mortality	LD50/30; adult	Garg and others, 1964, cited in Driver, 1994
Song Sparrow (*Melospiza melodia*)	Gamma	8000 mGy	Mortality	LD50/30	Driver, 1994
Shoveler (*Anas clypeata*)	Ionizing	8940 mGy	Mortality	LD50/30; adult	Tester and others, 1968, cited in Driver, 1994
Weaver finch (*Quelea quelea*)	Ionizing	10,600 mGy	Mortality	LD50/30; adult	Lofts and Rotblat, 1962, cited in Driver, 1994
Bluebird (*Sialia sialis*)	Ionizing	25,000 mGy	Mortality	LD50/30; adult	Willard, 1963, cited in Driver, 1994
Various species	Ionizing	0.001 mGy/d	Morbidity	Growth retardation; nestlings	Eisler, 1994
			Mammals		
Sheep	Gamma	2,000–3,000 mGy	Mortality	LD50/30	Driver, 1994
Pig	Gamma	2,000–3,000 mGy	Mortality	LD50/30	Driver, 1994
Dog	Gamma	3,500 mGy	Mortality	LD50/30	Driver, 1994
Guinea pig	Gamma	4,000 mGy	Mortality	LD50/30	Driver, 1994
Mice	Gamma	5,500 mGy	Mortality	LD50/30	Driver, 1994
Monkey	Gamma	6,000 mGy	Mortality	LD50/30	Driver, 1994
Chicken	Gamma	6,000 mGy	Mortality	LD50/30	Driver, 1994
Rat	Gamma	7,500 mGy	Mortality	LD50/30	Driver, 1994
Rabbit	Gamma	8,000 mGy	Mortality	LD50/30	Driver, 1994

**Within each receptor category, data are sorted by dose units (mGy, mGy/h, mGy/d, etc). Notes: For some studies, units were converted from rads to mGy for comparability purposes among dose and dose rates. References from Eisler (1994) were limited to those data that were not represented by the European numbers. For example, pine forest values from Eisler (1994) were not used. Data on single exposures were not used from Eisler (1994). Exposure length may be useful to include in this table; however, review of original studies found reporting of exposure duration was not well described.

Biological Receptors in the Terrestrial Food Chain

Radiation effects data for soil biota, terrestrial plants, and terrestrial animals include more acute studies than chronic studies and were generally too limited to establish presumptive no-effects levels (table 9) (Woodhead and Zinger, 2003). Reproductive capacity was the most frequently studied effect of acute radiation exposure in all biota; however, data on morbidity, mortality, and mutation were also available. Morbidity, or the general health of biota, was the most common effect reported for chronic exposures, although survival and effects on reproduction were also found. Radiation dose rates rarely exceeded 10 mGy/h, and threshold effects levels were generally 0.10 mGy/h. Responses to acute irradiation, particularly in terms of the LD50s, differed between taxonomic groups, but these differences became less pronounced in studies focused on continuous, low-dose rate radiation exposure measuring non-mortality endpoints (Woodhead and Zinger, 2003). Woodhead and Zinger (2003) also noted that increased responses were positively related to dose rate, which became unequivocal at dose rates greater than 10 mGy/h when conferred over a large fraction of the life span.

Terrestrial Nonvascular and Vascular Plants

Most studies of radiation effects on terrestrial plants were limited to field crops and woody species (table 9) (Woodhead and Zinger, 2003). Very few data were available from references derived from studies focused on mosses, lichens, and fungi, and there are few citations regarding radiation damage to plants resulting from the experimental incorporation of radioactive material into soil in field conditions. However, studies with enhanced natural radiation background at sites were available. Studies reporting the effects of high external radiation dose rate on forests in field conditions are also reported (for example, Sazykina, 2005).

The radiosensitivity of plants is commonly characterized by growth inhibition, reduced reproductive capacity, and reduced survival. The radiation exposure a plant receives is influenced by its morphology (for example, size, shape, density), age, and natural history attributes at the time of exposure (for example, differentiation of flowers). Seasonality also affects radiosensitivity, most often as a function of morphological or physiological changes (Woodhead and Zinger, 2003). Polyploid species, in which reproduction is achieved by vegetative growth, are generally more radioresistant and are typical of extreme environments such as the desert southwest of the United States. Abiotic (for example, light, temperature) and biotic (for example, interactions with neighboring plants) factors influence responses to ionizing radiation exposure (Sazykina, 2005).

Acute radiation effects data, primarily related to reduced growth, morphological changes, alteration in productivity, and abnormal shape and appearance, were available for seeds, seedlings, buds, meristamic tissues, and trees (table 9) (Woodhead and Zinger, 2003). Woodhead and Zinger (2003) noted that acute irradiation effects of coniferous trees in the spring varied (0.7–1.0 Gy), and the timing of irradiation of vegetative plants was critical. For example, survival endpoints (reported as LD50s) for pine trees ranged between 6 and 30 Gy after 2 years of exposure. Among agricultural crops, cereals such as wheat, barley, and oats were radiosensitive (LD50s of 16–22 Gy) (Woodhead and Zinger, 2003). Effects of chronic irradiation were generally related to morbidity, reproductive capacity, and mutation, but few were completed using low LET radiation (linear energy transfer; Woodhead and Zinger, 2003). Woodhead and Zinger (2003) noted threshold dose rates for morbidity (>0.10 mGy/h), reproduction (0.05 mGy/hr), and mutation (0.04 mGy/h) in terrestrial plants from a radiation database (table 9). Guidance values and estimated thresholds for the protection of terrestrial plants have been derived by various organizations (table 6).

Terrestrial Invertebrates

Soil fauna consist of a large variety of species ranging from protozoa to earthworms and arthropods. Chronic and acute radiation effects data, primarily related to mortality, are available but limited for these receptors (table 9) (Woodhead and Zinger, 2003). For example, effects data for low acute doses (less than 5 Gy) were rarely reported, and chronic exposure data relied predominantly on survival. Similar to fish, the acute sensitivity of soil fauna to radiation exposure is dependent on the developmental stage of organism considered. Woodhead and Zinger (2003) reported that a greater dose caused mortality (100–1,000 Gy) compared to morbidity and reproduction effects (5–20 Gy). Relatively sedentary animals, such as earthworms, are vulnerable to internal exposure by alpha radiation by directly foraging in the soil and can experience decreases in population sizes after chronic exposure (Woodhead and Zinger, 2003). Acute effects data less than 5 Gy were not reported for soil fauna including earthworms, springtails, and terrestrial isopods. Susceptibility to acute radiation exposure varies greatly between and within soil invertebrate groups (Woodhead and Zinger, 2003).

Data for chronic effects or radiation on soil fauna was limited. Woodhead and Zinger (2003) reported that soils with elevated natural background levels of radiation (0.001–0.002 mGy/h) contained fewer earthworms (*Eisenia nordenskioldi*, *Dendrobaena octaedra*, and *Octolasium lacteum*) and insect larvae (Dipterea, Elateridae) compared with reference areas; earthworms were particularly sensitive, possibly because of their close contact with soil. In a more recent study, Hertel-Aas and others (2007) reported that chronic gamma external radiation dose rates of 11 mGy/h reduced cocoon hatchability of the earthworm *Eisenia fetida* and concluded that dose rates of 43 mGy/h could collapse a population. Overall, Woodhead and Zinger (2003) observed there were too few data to draw conclusions on dose-effect relationships for terrestrial invertebrates in acute or chronic exposures. However, other studies have estimated a variety of dose rates as guidance values or thresholds for the protection of terrestrial invertebrates (table 6).

Amphibians and Reptiles

Amphibians and reptiles represent a significant portion of the vertebrate species in the segregation areas that may be exposed to fugitive radiation, but relatively few studies exist on their radiosensitivity (table 9) (Woodhead and Zinger, 2003). Amphibians are dependent on water for their reproduction and early life stages and shift from an herbivorous to carnivorous diet and to nocturnal activities as adults. Natural history is widely variable for each of the classes, and life span is species dependent. Too few data are available to estimate effects of chronic irradiation. Woodhead and Zinger (2003) noted that acute data are predominately focused on LD50s, which range between 2 and 22 Gy for a number of species of reptiles and amphibians, and that juvenile stages are more sensitive to radiation than other life stages.

The most common species with radiation data were the rough-skinned newt (*Taricha granulosa*), eastern newt (*Notophthalmus viridescens*), northern dusky salamander (*Desmognathus fuscus*), dwarf waterdog (*Necturus punctatus*), spotted grass frog (*Limnodynastes tasmaniensis*), northern leopard frog (*Rana pipiens*), squirrel treefrog (*Hyla squirella*), and toads (*Bufo* spp.). Estimated guidance values for the protection of amphibians and reptiles are also limited (table 6).

Birds

Acute and chronic irradiation data for birds are limited and dated (table 9) (Woodhead and Zinger, 2003). Woodhead and Zinger (2003) observed that data were not sufficient to draw conclusions on dose-effects relationships. However, lethal doses of radiation, regardless of their form, induce a wide range of pathologies because of disruptions in hemostasis and chromosome damage or breakage. For example, LD50/30s (radionuclide dose required to kill 50 percent of the animals in 30 days) for wild bird species exposed to ionizing radiation range from 4,000 to 25,000 mGy (Driver, 1994). Driver (1994) also characterized additional observational studies focused on birds, most in association with nuclear fuel production facilities or waste management operations.

For internal exposures to radiation, particularly alpha particles, vertebrates typically display hemorrhage in respiratory and gastrointestinal tracts in response to inhalation or ingestion of radioactive source materials. Birds may be at greater risk to radiation exposure than other wild vertebrates because of their natural history related to foraging and ingestion of grit, which effectively increases radiation dose (Bellamy and others, 1949, cited in Driver, 1994). Radiation dose will vary depending on food source, behavior, and habitat; therefore, these abiotic and biotic factors influence the accumulation of radionuclides in tissues. Despite the lack of empirical data, guidance values for the protection of birds have been estimated by various organizations (table 6).

Mammals

Biological effects of ionizing radiation in mammals have been extensively studied (table 9) (Woodhead and Zinger, 2003); however, nearly all the data detailing effects of acute or chronic irradiation in mammals are related to human health. These data provide a starting point for characterizing threshold-effects levels in wild mammals, but interspecies differences and field settings will affect radiation effects. Many studies focus on effects of acute exposure to high doses of low LET ionizing radiation administered to rodents (Woodhead and Zinger, 2003).

Data for acute radiation exposures typically measure survival, hematological measures (for example, cell counts, hemoglobin, piruvic acid content), and body and organ weights (for example, kidney, gonads, liver) in a variety of species and different life stages (for example, embryos, young, and adult) (Woodhead and Zinger, 2003). Exposure data on the effects of alpha emitters via inhalation or ingestion are limited compared to beta and gamma emitters; therefore, little data are available related to mortality, morbidity, or reproductive capacity for naturally occurring radioactive materials of the uranium decay series. However, the United Nations Scientific Committee on the Effects of Atomic Radiation (1993) provided a comprehensive review of effects of chronic exposure to low LET radiations on the mutation rate in the mouse. In addition, guidance values for the protection of mammals have been proposed by various organizations (table 6).

Radiation data for wildlife receptors other than mammals are relatively limited as suggested by the previous sections. The reliance on traditional laboratory animal data for extrapolation to terrestrial wildlife may encourage the continued comparative analysis of radiation dose for biota considered in ecological risk assessments. The inability to characterize the interspecies variability as a function of natural histories of wild mammals relative to laboratory mammals, principally rodents, is problematic. Biota concentration guides (BCGs) for radiation exposures to ecological receptors have been developed to help address these uncertainties (for example, Higley and others, 2003). Higley and others (2003) based BCGs on habitats rather than species, which are intended as conservative estimates of values protective of biota living in such habitats. For example, levels of radiation exposure associated with adverse effects to aquatic biota have been calculated as a dose-rate limit of 0.4 mGy/h. At a dose rate of less than 0.4 mGy/h, populations of the most sensitive aquatic organisms should be protected (National Council on Radiation Protection and Measurements, 1991). The National Council on Radiation Protection and Measurements (1991) developed their guidance values following a review of published literature on the effects of radiation on aquatic biota. Their findings indicated that the developing eggs and young of some fish are the most radiosensitive aquatic organisms. Similarly, the U.S. Department of Energy (2002) recommends that if the results of radiological models or dosimetric measurements exceed a radiation

dose rate of 0.1 mGy/h, then a more detailed evaluation of the effects of radiation exposure should be conducted. More recent studies should also be considered in future risk evaluations. For example, data related to external gamma irradiation effects (chronic and acute) are available, and the knowledge of relative biological effectiveness (RBE) values could be used to weight the biological effectiveness for other types of radiation (Chambers and others, 2006).

Radionuclide and Radiation Transfers in Aquatic and Terrestrial Ecosystems

Radionuclide trophic transfer has been discussed in previous sections of this document. In that section, bioconcentration and bioaccumulation processes of metals, including radionuclides, in ecotoxicological studies were evaluated. Bioconcentration or bioaccumulation factors and transfer coefficients have been characterized for a variety of environmental chemicals, yet empirical data detailing the movement and fate of radionuclides are relatively sparse. Driver (1994) had noted that there appeared to be discrimination against the movement of radionuclides of high atomic number from lower to higher trophic levels. Biotic factors influence the extent that a chemical is bioconcentrated. For example, aquatic invertebrates lose up to half of their body burden of adsorbed and absorbed radionuclides at each molt; hence, total accumulation over their entire life cycle is reduced (Wilhm, 1970). Additionally, life histories of animals dictate exposure pathways and scenarios (for example, exposures to burrow-dwelling vertebrates may be dominated by resuspended dusts or gases in their subterranean habitats). Moreover, abiotic factors, such as geochemical composition and specific activity, are also critical for anticipating which pathways may dominate exposure. For example, inhalation routes and dermal exposures will be increasingly critical in determining radiation dose when a physicochemical form suggests that atmospheric exposures (for example, dust) may dominate pathways linking radionuclides with biota. In addition, weathering will affect fate and transport of radionuclides, primarily by altering the matrix with respect to particle size distribution, crystallographic structures, porosity, and oxidation states.

These environmental factors strongly affect long-term processes that influence ecosystem transfers of radionuclides. The physicochemical form of nuclides in naturally occurring radioactive materials will influence biological uptake, accumulation, radiation doses, and biological effects differently in biota exposed to source materials. Sediments and soils act as sinks for radionuclides, particularly when these naturally occurring radioactive materials are present as particles and colloids. Radionuclide speciation is critical to biological uptake, regardless of whether that process results from a relatively simple pathway linking the abiotic environment to biota through direct uptake mediated by membranes in direct contact with the exposure medium (bioconcentration) or from a more complex pathway linking biota not only to the abiotic environment but also to biological sources such as dietary inputs that are manifested as bioaccumulation of radionuclides. As such, radionuclides may occur in various physicochemical forms, such as low molecular mass species (LMM), colloids, pseudocolloids, or particles (Salbu and others, 2004). In addition, each of these physicochemical forms will vary with respect to their mobility and fate in areas potentially open to disturbance linked to mining activities. For example, LMM species and colloids are relatively mobile, whereas smaller physicochemical species, such as particles, may be more likely to be entrapped in the soil or sediment matrix (Zhang and Brady, 2002). Within the context of physicochemical forms of radionuclides, LMM-species can cross biological membranes directly or indirectly following physicochemical interactions with low molecular weight ligands, whereas radionuclides interacting with high molecular weight ligands will display decreased uptake across membrane surfaces. Hence, high molecular weight forms are less bioavailable, and LMM species are more mobile (lower apparent dissociation constant [Kd]) and bioavailable (for example, higher apparent BAF) than colloids and particles.

Naturally occurring radioactive materials such as those characteristic of decay chains of ^{238}U, ^{235}U, and ^{232}Th produce a group of radionuclides with a wide range of half-lives. Most of these radionuclides are predominately alpha emitters, so internal exposures contribute significantly to the radiation dose. However, considering that uranium and thorium are always present in soils, particularly in areas identified as potentially of interest for mining, gamma radiation may become more prominent when characterizing external exposures and their associated absorbed doses in biota occupying habitats in these areas. Thus, exposure to the radioactivity hazards associated with the radionuclides uranium and thorium will be evaluated differently than the chemical hazards associated with these same chemicals. The hazards associated with external doses of ionizing radiation that can affect biota without actually being taken up require a different exposure paradigm for these types of radionuclides.

Data Gaps, Uncertainties, and Summary

Our overview of biological pathways and ecotoxicity values for radionuclides in the uranium decay series anticipates future studies focused on evaluations of risk or effects potentially associated with possible mining in northern Arizona. Given that anticipation, we (1) briefly identify data gaps and uncertainties we encountered that will likely be addressed in those future efforts, and (2) summarize our technical findings focused on pathways critical to linking sources with receptors that co-occur in the segregation areas.

Data Gaps and Uncertainties

The current distribution of uranium to the habitat surrounding segregation areas is not well defined. Other chapters of this assessment measured uranium in soil, sediment, and water, but biological samples were not collected. Therefore, site-specific uranium concentrations in biota inhabiting the segregation areas are not available, nor are they readily available from other sources. This type of information would provide baseline concentrations for various biological receptors that may be exposed to uranium and other inorganic chemicals associated with uranium mining activities. Information on wind dispersal (dust) of uranium also needs to be characterized for the mining areas. The use of dispersion models (for example, dust and particulates) would be useful to identify vulnerable habitats within the segregation areas. The physiochemical properties of water, soil, and sediment collected from the segregation areas will help to further characterize the potential exposure to biota. In addition, other elements present in breccia pipes, including silver, arsenic, cobalt, copper, molybdenum, nickel, lead, selenium, and zinc, may also pose a risk to biota utilizing the area (Weinrich, 1985), but they were not considered in this document. Selenium is of particular concern because elevated concentrations have been reported in biota in the Colorado River Basin (Radtke and others, 1988; Hinck and others, 2008), and adverse biological effects are well documented in aquatic and terrestrial organisms (see reviews by Jarvenin and Ankley, 1999; Ohlendorf, 2003).

Little biological information was found regarding habitat utilization in and near the segregation areas. Therefore, it is unclear where Federal and State species of concern inhabit the segregation areas and where habitat may be lost, degraded, or fragmented if mining operations are allowed. The physical habitat could be altered by the mine itself (for example, old, abandoned, and new tunnels if conventional mining is reinitiated), accessory buildings, waste rock disposal areas, roads, and traffic. Mining activities and roads may also influence infiltration rates, overland flow, and sediment movement important to the aquatic habitat. Site assessments of habitat utilization would help to better define exposure pathways. For example, some plant species of concern, such as the Grand Canyon rose (*Rosa stellata*), have been documented as growing in breccia pipes (Arizona Game and Fish Department, 2005). Such assessments would need to examine aquatic and terrestrial ecosystems and include microbes, plants, and animals. Assessments would also need to address if active and remediated mining sites could become attractive nuisances to species of concern (for example, bats utilizing refilled breccia pipes, talussnail [*Sonorella* spp.] using waste rock). Ingestion (water, diet, and incidental), inhalation, and dermal uptake rates of species of concern would further add to the exposure characterization.

The literature compilation was comprehensive, but it is unlikely that every pertinent reference on uranium decay series toxicity was obtained. However, extensive efforts were made to obtain data related to uranium mining operations in the region; uranium data related to depleted uranium from nuclear energy was included because information specific to uranium mining operations was sparse for some biological receptors. In addition, although literature compiled for this report was taken from peer-reviewed publications and Government-reviewed technical reports, our intent was to simply compile and report data and make them available to future studies focused on risks and effects. In future work, data inclusion and exclusion criteria would have to be developed, and the studies would necessarily undergo rigorous data quality analysis or validation required before using these effects data to establish benchmarks or guidance values. The empirical effects data compiled in tables 7–9 represent NOECs, LOECs, LC50s, and EC50s for a variety of endpoints (for example, survival, growth, reproduction), and they may contribute to the development of benchmarks or guidance values, but they were not intended to be applied to that end in our evaluation. Various resources on deriving or estimating guidance values (see Crane and Newman, 2000; Scholze and others, 2001; European Commission, 2003; Lepper, 2005; Canadian Council of Ministers of the Environment, 2007) should be consulted if site-specific benchmark or guidance values are to be developed. Moreover, existing guidance values to protect nonhuman biota (table 6) should not be applied to a risk assessment in the segregation areas unless the bases of these benchmarks are fully understood and are found to be appropriate to the site.

Uranium likely does not biomagnify in the aquatic or terrestrial food chain, and the toxicity, bioconcentration, and bioaccumulation of uranium are better characterized in aquatic systems than terrestrial systems. However, the absence of data should not be considered an indication that uranium is not a concern in seasonally variable terrestrial ecosystems such as those of the arid southwest United States. Precipitation in Arizona varies spatially, which affects exposure pathways that link aquatic biota to source materials potentially released from mining. The average annual precipitation is less than 250 mm (10 in) in much of the segregation areas, although wetter regions occurring at higher elevations (for example, the Kaibab Plateau) can have an annual precipitation of 635 mm (25 in) or more. Throughout the region, precipitation typically occurs during two rainy seasons: short, isolated, and intense summer rains and longer, widespread, and less intense winter rains (Peterson, 1994). Aquatic biota may be affected by releases of uranium or co-occurring constituents of mined ores in arid habitats like northern Arizona during rain events. However, little information is available regarding fate and environmental transport of these materials and their potential effects in desert flora and fauna in arid areas. Furthermore, these seasonal patterns in aquatic habitats (for example, ephemeral streams) and concern over climate change suggest that reduced water resources in the segregation areas are not cause to dismiss risk to aquatic species, as indicated by the occurrence of species of concern such as the Kanab ambersnail (*Oxyloma haydeni kanabensia*) in the area. As future risk or effect evaluations may detail, precipitation events will likely influence risk characterizations for aquatic species, particularly those species reliant on ephemeral aquatic habitats unique to arid landscapes such as that of the southwest United States. For example, the flash flooding typical to the area can cause increased runoff

and erosion that can erode weathered soils from waste rock or dry gullies and expose unweathered materials. Biota drinking from seasonal streams and ponds after flood events may experience increased exposure to radionuclides because of ingestion after flood events; such water may include runoff from contaminated soils that are relied upon heavily by wildlife when water becomes available after precipitation events. Wind dispersion is also a concern and should be considered when evaluating risks to biota.

Chemical and radiation effects thresholds for radionuclides were consistently limited to only a few species for most biological receptors, and limited data were available for wildlife species. Those species tested were few in number, generally traditional aquatic toxicity test species, and relatively undeveloped with respect to developing benchmark toxicity values. Minimal chemical toxicity data was available for microbes, aquatic vascular plants, terrestrial invertebrates, and amphibians, and no data were found for reptiles, birds, or mammalian wildlife. Toxicity data was most abundant, but still limited, for aquatic invertebrates, fish, and laboratory test mammals. The availability of only certain kinds of data creates extensive gaps in data on the effects of uranium exposure on the food web indicative of the segregation areas. We relied predominately on measures that captured effects based on gross alpha and gross beta exposures, in part because biota in field settings are exposed to a mixture of radiation sources. Nonetheless, data gaps were evident. No chemical or radiation toxicity data were available for species of snakes, lizards, birds, or mammals that are dependent on subterranean habitats such as burrows or caves. Furthermore, studies that focus on effects related to depleted uranium exposure may exclude important subterranean habitats like those found in the segregation areas because dispersed depleted uranium remains near the surface of the soil (Ribera and others, 1996). It is also unknown if these burrow-dwelling species are subject to increased exposures. We can say little, if anything, regarding the sensitivity of wildlife receptors to chemical and radiation exposure encountered in the field. For example, biota in burrow habitats may approach continuous exposure, as compared to surface-dwelling biota whose exposure to radionuclide-laden dust would vary relative to daily and seasonal patterns in wind direction and magnitude. In addition, State and Federal species of concern were found in most compartments of the generalized food web. Given the lack of toxicity data available for most biological receptors and the abundance of species of concern in the food web, the risk of uranium and its decay products to biological receptors using the segregation areas should not be underestimated. Future analysis of risks or effects would need to fully establish the comparative basis for evaluating exposure in the field. Specifically, given the geographic setting captured in this pathway analysis, future work must address the data gaps for characterizing "reference" versus "mined" lands in addition to ecotoxicity data; baseline conditions, such as uranium concentrations in materials collected from terrestrial habitats like soils and vegetation, would need to be determined. Similarly, a more completely characterized radiation survey would need to be completed.

Data related to the chemical toxicity of radionuclides, stable end-state constituents associated with uranium decay series, and other inorganic constituents of mined ores vary from absent to sufficient, but empirical data characterizing the radiation toxicity are consistently lacking. Radiation data were available for a limited number of species within each receptor category in our summary, and few data were found for desert flora and fauna. Radiation exposure is of specific concern for biota that spend prolonged periods of time (for example, hibernation, avoiding heat of the day) in the subterranean environment. Most radiation data available are for laboratory test species and are intended to address radiation safety concerns for humans; they may not include important exposure pathways like incidental ingestion of radionuclides through burrowing, preening, or caching food of wild mammals. More infrequently, wildlife have been considered in regulatory contexts related to nuclear energy development and waste management rather than uranium mining and processing. Few studies have attempted to quantify the risk to biota directly caused by the chemical or radiation released by means other than those linked to uranium mill tailings. Research into the biological effects is strongly biased towards human health, yet attention focused on the biological effects of the uranium production of nuclear energy development has increased. For example, plants and fish residing near mill tailings can take up radionuclides and introduce a health risks to individuals, communities, and ecosystems in the food chain, including humans (International Atomic Energy Agency, 2004).

A compilation of the existing literature has indicated that acute exposures to mill tailings often result in sublethal effects to the biota; however, effects resulting from chronic exposures are greatly unknown. The long term risks of chronic exposure are not understood, particularly in terms of potential genetic effects on species populations, density, ecosystem dynamics, and biodiversity. Additionally, the possible synergistic effects of radiation dose and chemical exposure on other metals, semi-metals, and other toxic compounds common at uranium mining and processing facilities also need to be considered.

Given the data gaps we encountered, monitoring and surveillance of radionuclide exposure to biological resources would be appropriate if the segregation areas are opened to mining activities. Research needs will vary among uranium ore deposits throughout the United States. For example, ecological studies or monitoring activities associated with uranium mining activities will likely identify and define stressor exposures to biological resources in areas of concern that are unlike those identified in this evaluation. As indicated by this report, initial evaluations of system vulnerabilities should focus on resources captured in the preliminary conceptual model. In part, the identification of activities associated with the development of energy resources (such as those linked to uranium mining—infrastructure, related surface and subsurface disturbance) would allow connections to be made that shape the pathways between mineral sources such as uranium deposits in breccia pipes and biological resources that risk hazardous exposure to those minerals.

Summary

This chapter focused on sources of and exposure to naturally occurring uranium and other radionuclides associated with uranium mining in northern Arizona, particularly in those segregation areas adjacent to Grand Canyon National Park. Federal and State species of concern for the area were identified in order to develop exposure pathways between biological receptors and uranium mining activities. Relevant scientific literature on toxicity threshold effects levels for uranium and associated radionuclides was then compiled for aquatic and terrestrial biota.

Our literature compilation included various biological receptor categories, including microbes, plants, invertebrates, fishes, amphibians, reptiles, birds, and mammals, that are integral parts of the food web in the segregation areas. Species of concern from all of these categories except microbes, identified by State and Federal agencies primarily based on their small home range and limited population size, inhabit the segregation areas. Certain biological receptors are potentially more susceptible to uranium exposure; herbivores, aquatic species, and burrowing animals are of particular concern given the likely exposure pathways and available toxicity data. For example, certain species of reptiles, amphibians, birds, and mammals in the segregation areas spend significant amounts of time in burrows or mine tunnels where they can inhale or ingest uranium and other radionuclides through digging, eating, preening, and hibernating. Toxicity data for the sensitivity of burrowing animals to radionuclides is not available in the existing scientific literature. Herbivores may also be exposed to radionuclides though the ingestion of radionuclides that have been aerially deposited on vegetation. Other receptors such as carnivorous birds and mammals that do not utilize subterranean habitats may be less sensitive to exposure in the segregation areas because uranium does not bioaccumulate or biomagnify in food chains. Microbes and other lower organisms may also be less sensitive to the chemical and radiation hazards of uranium and associated radionuclides, but certain invertebrates including the Kanab ambersnail need additional consideration given their endangered status.

Results of this literature compilation highlight that toxicity data for many radionuclides and biological receptors are lacking. Other authors have developed chemical and radiation toxicity guidance values for uranium decay products (for example, Suter and Tsao, 1996; Woodhead and Zinger, 2003; Sheppard and others, 2005; U.S. Department of Energy, 2005; United Nations Scientific Committee on the Effects of Atomic Radiation, 2008). However, directly applying these values to biota in the segregation areas may be inappropriate. The presumptive values are not limited to naturally occurring uranium at mining areas, which may have chronic exposures at low concentrations. In addition, some guidance values are based on little or no empirical data but rather are driven by mathematical models, the biological relevance of which cannot be determined. Empirical data for the toxicity of uranium and associated radionuclides is limited for the species of concern, specifically reptiles, birds, and mammalian wildlife, that represent essential components of the food web in the segregation areas and have unique habitats and life history strategies. Nevertheless, these existing guidance values were included in this chapter (tables 6–8) to highlight that guidance values for radionuclides have been recommended by various committees. Such recommendations could be useful in the environmental impact statement to be developed, as long as the derivations of the guidance values are clearly understood. Future evaluations of risks or effects may consider developing site-specific benchmarks related to uranium mining in the segregation areas.

Other inorganic constituents (for example, selenium) that are characteristic of uranium ores typical of breccia pipes were not considered in this evaluation of pathways. Whereas these constituents may not present radiation hazards in field exposures, some of these elements are potentially as toxic, if not more toxic, than uranium. Available toxicity data for these inorganic chemicals vary, but toxicity thresholds are available for some. The pathways linking sources with receptors are likely identical regardless of the chemical and radiation hazards associated with breccia pipes. Future studies focused on risks of uranium mining need to develop empirical data to characterize the ore materials included in the exposure mixtures that are potentially released consequent to mining operations. Radiation toxicity data are highly limited for biological receptors likely to be exposed in the field, and little empirical data are available to develop support needed for radiation measurements in biota in natural settings. In our present evaluation, effect thresholds for chemicals of concern and radiation hazards were not available for biological receptors essential to the food web in the segregation areas, including the soil crust community, vascular plants, terrestrial invertebrates, amphibians, reptiles, birds, and mammals.

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Appendix 1. ^{235}Uranium and ^{232}Thorium Decay Series

[Decay series information taken from Peterson and others, 2005]

Table 1.1. Half-life for constituent elements of ^{235}U decay series.

Element	Type of radiation released in daughter formation	Half-life
Uranium-235	Alpha	7.0×10^{9} years
Thorium-231	Beta	26 hours
Protactinium-231	Alpha	33,000 years
Actinium-227	Alpha	22 years
Thorium-227	Beta	19 years
Francium-223	Beta	22 minutes
Radon-223	Alpha	11 days
Radon-219	Alpha	4.0 seconds
Polonium-215	Beta	1.8 milliseconds
Lead-211	Beta	36 minutes
Bismuth-211	Alpha	2.1 minutes
Thallium-207	Beta	4.8 minutes
Lead-207	Stable	

Table 1.2. Half-life for constituent elements of ^{232}Th decay series.

Element	Type of radiation released in daughter formation	Half-life
Thorium-232	Alpha	1.405×10^{10} years
Radium-228	Beta	6.7 years
Actinium-228	Beta	6.13 hours
Thorium-228	Alpha	1.91 years
Radium-224	Alpha	3.64 days
Radon-220	Alpha	55 seconds
Polonium-216	Alpha	0.15 seconds
Lead-212	Beta	10.64 hours
Bismuth-212	Alpha	60.6 minutes
Polonium-212	Beta	304 nanoseconds
Thallium-208	Alpha	3.1 minutes
Lead-208	Stable	

Appendix 2. Radiation Measurement

This appendix briefly describes terms related to radiation exposure and dose, as a means to provide the reader with a background for interpreting the numeric values used to characterize radiation effects discussed in this report. For more in depth descriptions of the concepts and terms used to measure radiation (especially as related to exposure) and the total absorbed dose received by a receptor, see Blaylock and others (1993), Brechignac and Desmet (2005), and Turner (2007).

Not unlike other units of measure, units of measurement for radiation have been standardized by the International System of Units (SI), which can trace its lineage to the metric system. A second set of units of measurements, however, are commonly encountered in the United States and serve as a "conventional system" that remains in wide use today throughout the country.

Depending on the properties of radiation being characterized, different units of measure are applied. For characterizing exposure, the amount of radiation being emitted by a radioactive source is measured by the curie (Ci) in the conventional system or by the becquerel (Bq) in SI units. Because of an excess of energy and stability, energy is released from radioactive elements primarily in the form of alpha particles, beta particles, or gamma rays. The units Ci or Bq express the number of disintegrations of these radioactive atoms during an interval of time. One Ci equals 37 billion (3.7×10^{10}) disintegrations per second; one Bq equals one disintegration per second; hence, one Ci equals 37 billion (3.7×10^{10}) Bq. Standard prefixes common in scientific nomenclature are applied to alter the quantity expressed by the base units (for example, mCi = 0.001 Ci; kBq = 1,000 Bq).

Radiation dose refers to the energy absorbed per mass of biological tissues (for example, joules per kilogram, J/kg) from radioactive decay. Following conventional nomenclature, radiation dose is measured by the rad, which stands for radiation absorbed dose; the SI unit is the gray (Gy). The rad is a unit of absorbed dose for any ionizing radiation and equals 100 ergs absorbed per gram of material, or 0.01 J/kg; 1 Gy equals 100 rad, or 1 J/kg. In the United States, radiation exposures were historically measured in roentgens/hour (R/h); the roentgen is equivalent to 2.58×10^{-4} coulomb/kg (C/kg) in air in SI units. For water and soft tissues the absorbed dose per roentgen is between 0.93 and 0.98 rad; hence, the roentgen and rad are nearly equivalent numerically. Regardless of the units being used to characterize absorbed dose, the amount of energy intercepted by the receptor is expressed per unit of weight.

Roentgens were initially developed to address issues related to human radiation safety, particularly those risks associated with exposure to x-rays or gamma rays, and the derivative unit for radiation dose in humans was the roentgen equivalent in man, or rem. A rem is the product of the absorbed dose in roentgens (R) and the biological efficiency or dose equivalent (DE) of the radiation. DE may be regarded as an expression of dose in terms of its biological effect, which accounts for the variation in absorbed dose of radiation for different forms of radiation (for example, alpha, beta, or gamma). More specifically, each form of radiation is characterized by different energies; hence, each form of radiation may give rise to a greater biological effect at the same absorbed dose based on the Quality Factor (Q) characteristic of that type of radiation:

$$DE = \text{Absorbed Dose} \times \text{Quality Factor (Q)}$$

where

$Q = 1$ for gamma, x-ray, and beta radiation,

and

$Q = 10$ for alpha radiation.

Adverse effects associated with absorbed dose, then, vary as a function of Q, which is related to the energy loss of radiation per unit path length, or its linear energy transfer (LET). Generally, the greater the LET in tissue, the greater the biological effect (adverse). The SI equivalent of the rem is the sievert (Sv), where 1 Sv = 100 rem when a radiation weighting factor finds 1 rem = 1.07185 R. The official units for radiation measurement are the SI units: the gray (Gy) for absorbed dose and the sievert (Sv) for equivalent dose.

Table 2.1. Equivalent units of radiation measurement.

	International System (SI)		Conventional (US)
	Multiply	*By*	*To obtain*
Exposure (activity of source)	becquerel (Bq)	3.7×10^{10}	curie (Ci)
Absorbed dose	gray (Gy)	0.01	rad (rad)
Biologically effective dose	sievert (Si)	0.01	rem (rem)
Intensity	coulomb/kg in air	2.58×10^{-4}	roentgen (R)

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