

7.0 Screening-Level Ecological Risk Assessment

7.1 Introduction

In order to determine the potential for ecological risks posed by site-related chemicals at Range K, Parcel 203(7), a screening-level ecological risk assessment (SLERA) was conducted. This SLERA consisted of a description of the habitat(s) in and around the range, a discussion of the constituents detected in samples collected from environmental media at the range, a discussion of the conceptual site model, an estimation of the screening-level risk, the identification of the constituents of potential ecological concern, an uncertainty analysis, a discussion of the different lines of evidence, and a summary of the results and conclusions.

7.2 Environmental Setting

The environmental setting of Range K is entirely terrestrial in nature, although a small ephemeral drainage feature occurs approximately 300 feet south of the study area during periods of significant precipitation. The terrestrial habitat at Range K is best characterized as relatively immature mixed deciduous/ coniferous forest, although conifers are dominant. The canopy species typically found in this forest type at FTMC include shortleaf pine (*Pinus echinata*), loblolly pine (*Pinus taeda*), longleaf pine (*Pinus palustris*), yellow poplar (*Liriodendron tulipifera*), sweetgum (*Liquidambar styraciflua*), black gum (*Nyssa sylvatica*), white oak (*Quercus alba*), and northern red oak (*Quercus rubra*). The dominant understory species of this area are red maple (*Acer rubrum*), flowering dogwood (*Cornus florida*), witch hazel (*Hamamelis virginia*), sweetgum (*Liquidambar styraciflua*), wild black cherry (*Prunus serotina*), hackberry (*Celtis occidentalis*), black walnut (*Juglans nigra*), and sourwood (*Oxydendrum arboreum*). The shrub layer is dominated by chokeberry (*Aronia arbutifolia*), southern low blueberry (*Vaccinium pallidum*), southern wild raisin (*Viburnum nudum*), Virginia creeper (*Parthenocissus quinquefolia*), poison ivy (*Toxicodendron radicans*), and yellowroot (*Xanthorhiza simplicissima*). Numerous muscadine grape (*Vitis rotundifolia*) vines are also present in this habitat type. The herbaceous layer consists mainly of foxglove (*Aureolaria pectinata*), little bluestem (*Schizachyrium scoparium*), narrow-leaved sensitive brier (*Schrankia microphylla*), poison oak (*Toxicodendron toxicarium*), goat's rue (*Tephrosia virginiana*), and pencil flower (*Stylosanthes biflora*).

Typical terrestrial species that may inhabit the area of Range K include opossum, short-tailed shrew, raccoon, white-tail deer, red fox, coyote, gray squirrel, striped skunk, a number of species of mice and rats (e.g., white-footed mouse, eastern harvest mouse, cotton mouse, eastern

woodrat, and hispid cotton rat), and eastern cottontail. Approximately 200 avian species reside at FTMC at least part of the year (USACE, 1998). Common species expected to occur in the vicinity of Range K include northern cardinal (*Cardinalis cardinalis*), northern mockingbird (*Mimus polyglottus*), warblers (*Dendroica spp.*), indigo bunting (*Passerina cyanea*), red-eyed vireo (*Vireo olivaceus*), American crow (*Corvus brachyrhynchos*), bluejay (*Cyanocitta cristata*), several species of woodpeckers (*Melanerpes spp.*, *Picoices spp.*), and Carolina chickadee (*Parus carolinensis*). Game birds present in the vicinity of Range K may include northern bobwhite (*Colinus virginianus*), mourning dove (*Zenaida macroura*), and eastern wild turkey (*Meleagris gallopavo*). A variety of woodland hawks (e.g., sharp-shinned hawk) and other raptors (e.g., red-tailed hawk, barred owl, and great horned owl) are expected to use this area for a hunting and/or nesting area.

A small, ephemeral drainage feature is present approximately 300 feet south of the Range K study area. This drainage feature runs in a northeast-to-southwest direction, following a topographic depression and only contains water during periods of significant precipitation. When water is present, it is narrow (2 to 3 feet wide) and shallow (3 to 6 inches deep). The substrate is mostly cobbles and gravel with small depositional areas of sand and leaf litter, interspersed throughout. The small size and ephemeral nature of this drainage feature preclude the presence of most fish species and other animals that might prey on fish (piscivores); however, semi-aquatic species (amphibians) could occur in this drainage feature and drought-tolerant fish species could occur during periods of high precipitation.

7.3 Constituents Detected On Site

The sampling and analysis programs conducted at Range K were designed based on a number of factors, including:

- Site history;
- Results of the environmental baseline survey (EBS); and
- Results of previous sampling and analysis programs.

The sampling and analysis programs at Range K are described in Chapter 2.0 of this report. Constituents detected in surface soil and groundwater at Range K are presented in Chapter 4.0 of this report.

In general, inorganic constituents were commonly detected in surface soils at Range K, but organic compounds (i.e., pesticides, nitroaromatic compounds, and VOCs) were less frequently detected and at relatively low concentrations. Aluminum, arsenic, barium, beryllium, cadmium,

chromium, cobalt, iron, nickel, selenium, vanadium, and zinc were detected in at least one surface soil sample out of a total of 13 surface soil samples collected at concentrations that exceeded their respective ESVs and background threshold values (BTV). There was no discernable pattern to the elevated concentrations of these inorganic constituents in surface soil at Range K.

The nitroaromatic compound 1,3,5-trinitrobenzene was detected in one surface soil sample at a concentration that exceeded the ESV. 2-Nitrotoluene, 3-nitrotoluene, and p-nitrotoluene were detected in one surface soil sample each, but none of these compounds have ESVs associated with them. The VOC trichloroethene was detected in one surface soil sample at a concentration that exceeded its ESV.

Surface water and sediment were not sampled at Range K because there are no permanent surface water bodies directly associated with Range K. There is an intermittent stream located approximately 300 feet south of Range K. Because this drainage feature does not drain Range K itself, it was not sampled.

Inorganic constituents were commonly detected in groundwater at Range K, but organic compounds (i.e., pesticides, SVOCs, and VOCs) were less frequently detected and at relatively low concentrations. Aluminum, barium, cadmium, chromium, cobalt, iron, lead, manganese, mercury, and vanadium were detected in at least one sample out of a total of 18 groundwater samples collected at concentrations that exceeded their respective ESVs and BTVs. The pesticides alpha-chlordane, heptachlor, and methoxychlor were detected in one sample out of 14 groundwater samples analyzed for pesticides. The SVOC bis(2-ethylhexyl)phthalate and the volatile organic compounds 1,1,2,2-tetrachloroethane, and vinyl chloride were also detected in groundwater samples from Range K.

7.4 Site Conceptual Model

The ecological site conceptual model (SCM) is a simplified, schematic diagram of possible exposure pathways and the means by which contaminants are transported from the primary contaminant source(s) to ecological receptors (Figure 7-1). The exposure scenarios include the sources, environmental transport, partitioning of the contaminants amongst various environmental media, potential chemical/biological transformation processes, and identification of potential routes of exposure for the ecological receptors. In this chapter the SCM will be described in relation to constituent fate and transport properties, the ecotoxicity of the various

constituents, potential ecological receptors at Range K, and the complete exposure pathways expected to exist at Range K.

7.4.1 Constituent Fate and Transport

The environmental fate and transport of contaminants in surface soil and groundwater at Range K will govern the potential for exposures to ecological receptors. In general, contaminants in environmental media may be available for direct exposure (e.g., plants exposed to surface soil), and they may also have the potential to migrate to other environmental media or areas of the site. This section discusses the mechanisms by which contaminants can be transported and the chemical properties that determine their transport.

7.4.1.1 Fate and Transport in Soil

Contaminants in surface soil at Range K have the potential to be transported from their source area to other areas within Range K and to off-site locations by a number of mechanisms, including volatilization, dust entrainment, surface runoff, and infiltration to subsurface soil/groundwater.

The majority of the constituents identified in surface soil at Range K are inorganic. These inorganic constituents are generally closely related to the soil particles themselves and are not expected to be transported great distances from their source. Volatilization of inorganic constituents is expected to be insignificant. Fugitive dust generation and entrainment by the wind with subsequent dispersion by atmospheric mixing could transport particulate-associated contaminants to other parts of Range K and to off-site locations. However, because the entire site is forested, fugitive dust generation is expected to be minimal. Several nitroaromatic compounds and one VOC were identified in the surface soil at Range K. These volatile constituents have a high potential to volatilize to the atmosphere and be transported from their source area via air movement. The concentrations of nitroaromatics and VOCs detected in surface soil at Range K are low; therefore, this transport mechanism is expected to be insignificant with respect to other transport mechanisms potentially active at this site.

The transport of surface soil-associated contaminants by surface runoff is another potential transport mechanism. Although the potential exists for surface runoff, there are no permanent surface water bodies in close proximity to Range K. There is a small ephemeral drainage feature approximately 300 feet south of the study area that could potentially transport site-related contaminants downstream during periods of significant precipitation. This drainage feature is dry during significant periods of the year; therefore, contaminant transport via surface water

could only potentially occur when significant precipitation occurs. Additionally, the ephemeral drainage feature does not directly drain Range K. Therefore, surface runoff is expected to be an insignificant transport mechanism at Range K.

Contaminants in surface soil at Range K may be transported vertically to subsurface soils and groundwater via solubilization in rainwater and infiltration. Migration in this manner is dependent upon contaminant solubility and frequency of rainfall. Although the soil types (sand, stone, and gravel) in the vicinity of Range K are expected to promote relatively rapid infiltration of rainwater, the less soluble constituents (i.e., inorganic compounds) found at Range K are not likely to migrate to any great extent vertically due to their relatively low solubilities.

The transfer of contaminants in surface soil to terrestrial plants through root uptake and to terrestrial animals through ingestion and other pathways are potentially significant transfer mechanisms. Many metals are readily absorbed from soil by plants, but they are not biomagnified to a great extent through the food web. Mercury is a notable exception, as it does tend to biomagnify through the food chain (Eisler, 1987). Nitroaromatic compounds and volatile organic compounds do not bioaccumulate to any significant extent; therefore, food web transfer of these constituents is expected to be minimal.

VOCs in the surface soil at Range K are expected to volatilize and/or photolyze relatively rapidly when exposed to sunlight (half-lives of 3 hours to 5 days) (Burrows et al., 1989). The other surface soil contaminants (metals and nitroaromatics) are expected to remain in the soil relatively unchanged by physical and/or chemical processes for much longer periods of time.

7.4.1.2 Fate and Transport in Surface Water

Due to the lack of a permanent surface water body on or directly adjacent to Range K, surface water fate and transport properties are considered relatively insignificant with respect to constituents detected at Range K. Groundwater from Range K may infiltrate the ephemeral drainage feature that runs in a southwesterly direction approximately 300 feet south of the Range K study area. If groundwater discharges to this ephemeral drainage feature during periods of significant precipitation, groundwater constituents could be transported downstream. However, the increased flow in the drainage feature during storm events would greatly dilute the concentrations of constituents detected in groundwater to levels that are likely ecologically irrelevant.

7.4.1.3 Fate and Transport in Sediment

Due to the lack of a permanent surface water body on or directly adjacent to Range K, sediment fate and transport properties are considered relatively insignificant with respect to constituents detected at Range K.

7.4.1.4 Fate and Transport in Groundwater

Contaminants in groundwater at Range K could be transported off-site via local groundwater flow, which is to the southwest. Although ecological receptors are unlikely to be exposed to groundwater directly, groundwater discharge to surface water bodies could present exposure opportunities for ecological receptors. Groundwater from Range K could discharge to the ephemeral drainage feature south of Range K if the potentiometric surface rises above the creek bed during periods of high precipitation. If groundwater discharges to the drainage feature, then groundwater contaminants could be transported downstream. This transport mechanism would only be active during periods of significant precipitation when the potentiometric surface is elevated.

7.4.2 Ecotoxicity

The ecotoxicological properties of the constituents detected at concentrations that exceeded their respective ESVs and BTVs in surface soil or groundwater at Range K are discussed in the following sections.

7.4.2.1 Aluminum

Aluminum is the most abundant element in the earth's crust. Minimal evidence exists concerning the essentiality of aluminum. Aluminum is generally considered to have low mammalian toxicity (Hayes, 1994).

Plants. Aluminum appears to be essential for the growth of some plant species (Kabata-Pendias and Pendias, 1992). Higher concentrations of aluminum are usually detected in older rather than younger leaves (Bollard, 1983). Difference in the toxicity of aluminum to plants is closely linked to the different uptake and transport of calcium (Foy, 1974). Interactions of aluminum with potassium, silicon, and organic acids have also been reported (Foy, 1974). According to Foy (1974), aluminum toxicity in plants usually does not occur in soils with pH values above 5.5. Toxicity is, however, common at soil pH values below 5.0 (Foy (1974)). The addition of nitrogenous fertilizers to soil increases the toxicity of aluminum to plants by displacing exchangeable aluminum into soil solution and lowering soil pH (Foy, 1974).

Concentrations of aluminum in leaf tissue that are excessive or toxic to various plant species range from 5 to 10 mg/kg (dry weight) (Kabata-Pendias and Pendias, 1992). A soil concentration of 50 mg/kg (dry weight) has been proposed by Efroymson et al., (1997) as a benchmark screening value for aluminum phytotoxicity. Signs of aluminum toxicity in plants include overall stunting of growth, the presence of dark green leaves, purpling of stems, death of leaf tips, and coralloid and damaged root systems (Kabata-Pendias and Pendias, 1992)

Terrestrial Invertebrates. No information was found regarding the toxicity of aluminum to terrestrial invertebrates.

Mammals. Aluminum is not an essential element for animal growth and development. Limited data exist on the concentrations and effects of aluminum on wildlife. Most absorbed aluminum is eliminated through the kidney (HSDB, 1996).

Data are scarce on the effects of aluminum on wild mammals. Laboratory studies have shown inhalation of aluminum dust to induce infections and diseases of the lung (HSDB, 1996). A derived chronic no-observed-adverse-effects-level (NOAEL) of 0.043 mg/kg/day has been reported for laboratory rats exposed to aluminum (EPA, 2003b). Laboratory-derived toxicity data from studies conducted with mice fed aluminum ($AlCl_3$) in their drinking water over three generations were used to derive a NOAEL value of 1.93 mg/kg/day (Ondreicka et al., 1966). Reproduction was the endpoint for these studies.

Birds. Dietary ingestion of aluminum at concentrations of approximately 1,400 mg/kg produced declines in inorganic phosphorus levels in blood and resulted in the development of severe rickets in chickens (HSDB, 1996). No adverse effects were observed in black ducks (*Anas rubripes*) fed diets containing 1,000 mg/kg aluminum as aluminum sulfate over a period of 12 days (Sparling, 1990). Diets with low calcium and phosphorus concentrations adversely affected the response of the ducks to aluminum (Sparling, 1990). An acute LD_{50} (lethal dose that will result in 50 percent mortality in a test population) of 111 mg/kg has been reported for exposure of birds to aluminum (Schafer et al., 1983). Ringed doves fed aluminum as $Al_2(SO_4)_3$ for four months in their diet showed no adverse effects at a dose level of 1,000 ppm. This dosage was used to derive a NOAEL value of 109.7 mg/kg/day (Carreire et al., 1986). Reproduction was the critical endpoint in this study.

Aquatic Life. Bioconcentration of aluminum has been reported for several freshwater species. A bioconcentration factor for daphnids exposed to aluminum is 574 (Cowgill and Burns, 1975). Crayfish have been reported to have a bioconcentration factor for aluminum of 1,305 (Malley et al., 1987). The National Recommended Water Quality Criteria (EPA, 2002d) for aluminum are 750 and 87 µg/L for acute and chronic exposures, respectively. The lowest chronic values for aluminum toxicity reported in the literature for fish and daphnids are 3,290 and 1,900 µg/L, respectively (Suter and Tsao, 1996). The test EC₂₀ (the concentration that will result in a specified effect on 20 percent of the test population) for fish can be used as a benchmark indicative of production within a population. The EC₂₀ value for aluminum is 4,700 µg/L (Suter and Tsao, 1996). The EC₂₀ value for aluminum with respect to daphnids is 540 µg/L (Suter and Tsao, 1996).

7.4.2.2 Arsenic

Arsenic naturally occurs as sulfides and as complex sulfides of iron, nickel, and cobalt (Woolson, 1975). In one form or another, arsenic is present in rocks, soils, water, and living organisms at concentrations of parts per billion (ppb) to parts per million (ppm) (National Academy of Sciences [NAS], 1977a). Arsenic is ubiquitous in living tissue and is constantly being oxidized, reduced, or otherwise metabolized. In soils, insoluble or slightly soluble arsenic compounds are constantly being re-solubilized, and the arsenic is being presented for plant uptake or reduction by organisms and chemical processes (NAS, 1977a). In abundance of elements, arsenic ranks 20th in the Earth's crust (1.5 to 2 mg/kg), 14th in sea water, and 12th in the human body (Woolson, 1975). It occurs in various forms, including inorganic and organic compounds and trivalent and pentavalent states (Pershagen and Vahter, 1979).

Plants. NAS (1977a) reports background arsenic concentrations in terrestrial plants as ranging from 0.01 to 5 mg/kg (dry weight). Generally, roots of a plant contain higher concentrations of arsenic than leaves. The toxicity of arsenic to plants may differ due to different soil conditions. Various chemical forms of arsenic have different phytotoxicities. In general, arsenates are less toxic to plants than arsenites. Concentrations of arsenic in leaf tissue that are excessive or toxic to various plant species range from 5 to 20 mg/kg (dry weight) (Kabata-Pendias and Pendias, 1992). A soil concentration of 10 mg/kg has been proposed by Efrogymson et al. (1997) as a benchmark screening value for phytotoxicity in soils. General symptoms of arsenic toxicity in plants include the presence of red-brown necrotic spots on old leaves, yellowing or browning roots, depressed tillering, wilting of new leaves, and root discoloration (Kabata-Pendias and Pendias, 1992).

Terrestrial Invertebrates. Fischer and Koszorus (1992) tested the earthworm *Eisenia fetida* for effects on growth and reproduction by exposing five-week old worms to 68 mg/kg arsenic (as potassium arsenate) in soil. Reduced cocoon production was reported at this exposure level. Based on the results of this test, a soil benchmark value of 60 mg/kg has been proposed by Efroymsen et al. (1997).

Mammals. Arsenic is more toxic to wildlife in the trivalent form as compared to either pentavalent arsenic or organic arsenicals. Arsenic poisoning has been reported in acute episodes but chronic arsenosis is rarely encountered (NAS, 1977a). The probability of chronic arsenic poisoning from continuous ingestion of small doses is rare, because detoxification and excretion are rapid (Woolson, 1975). General signs of arsenic toxicosis include intense abdominal pain, staggering gait, extreme weakness, trembling, fast and feeble pulse, collapse, and death (Eisler, 1988a). Adverse effects in mammals were noted in single oral doses of 2.5 to 33 mg/kg body weight, and at chronic oral doses of 1 to 10 mg/kg body weight. As little as 1.7 mg/kg arsenic has been shown to produce fetal death and malformations during critical stages of hamster embryogenesis. Various species of rodents, exposed to cacodylic acid through various routes, exhibited LD₅₀s ranging from 470 to 830 mg/kg body weight (Hood, 1985).

Laboratory data for arsenic toxicity (as arsenite) through drinking water in laboratory mice were used to derive a NOAEL value of 0.126 mg/kg/day (Schroeder and Mitchener, 1971). Reproduction was the endpoint for these laboratory tests. Arsenic concentrations of greater than 10 mg/kg (wet weight) in tissue are usually indicative of arsenic poisoning (Goede, 1985). Detoxification and excretion of arsenic are relatively rapid processes, making the probability of chronic arsenic poisoning from the continuous ingestion of small amounts of arsenic a rare event (Eisler, 1988a).

Birds. As with mammals, arsenic poisoning in birds has been reported in acute episodes but chronic arsenosis is rarely encountered. Signs of inorganic trivalent arsenite poisoning in birds include muscular incoordination, debility, slowness, jerkiness, falling hyperactivity, immobility, and seizures (Eisler, 1988a). Studies reported by Hudson et al. (1984), using mallard ducks (*Anas platyrhynchos*) fed sodium arsenite, determined an LC₅₀ of 323 mg/kg body weight. Copper acetoarsenite fed to the northern bobwhite (*Colinus virginianus*) at 480 mg/kg proved fatal to 50 percent of the test organisms in 11 days (NAS, 1977a). The grey partridge (*Perdix perdix*) succumbed to 300 mg/kg body weight of lead arsenate in 52 hours (NAS, 1977a).

Brown-headed cowbirds fed arsenic in their diets for seven months experienced mortality at doses as low as 75 ppm. These data were used to derive a NOAEL value of 2.46 mg/kg/day (USFWS, 1969). Data from a study of mallard ducks fed arsenic in their diets for 128 days were used to derive a NOAEL value of 5.14 mg/kg/day (USFWS, 1964). Mortality was the endpoint for both of these studies.

Aquatic Life. Toxic and other effects of arsenicals to aquatic life are significantly modified by numerous biological and abiotic factors (Woolson, 1975; NAS, 1977a; National Research Council of Canada [NRCC], 1978; EPA, 1980a; Howard et al., 1984; Michnowicz and Weaks, 1984; Bryant et al., 1985; EPA, 1985a; Sanders, 1986). The LC₅₀ values, for example, are markedly affected by water temperature, pH, Eh, organic content, phosphate concentration, suspended solids, and presence of other substances and toxicants, as well as arsenic speciation, and duration of exposure. In general, inorganic arsenicals are more toxic than organoarsenicals to aquatic biota, and trivalent species are more toxic than pentavalent species. Early life stages are most sensitive, and large interspecies differences have been recorded, even among those closely related taxonomically. Juvenile bluegills (*Lepomis macrochirus*) exhibited reduced survival after sixteen weeks when exposed to a single treatment of trivalent arsenic at 0.69 mg/L (EPA, 1980a; EPA, 1985a). The adult bluegill population was reduced 42 percent after several monthly applications of 4 mg/L trivalent arsenic (NAS, 1977a).

As with fish, toxic and other effects of arsenicals to aquatic invertebrates are significantly modified by numerous biological and abiotic factors (Woolson, 1975; NAS, 1977a; NRCC, 1978; EPA, 1980a; Howard et al., 1984; Michnowicz and Weaks, 1984; Bryant et al., 1985; EPA, 1985a; Sanders, 1986). The cladoceran *Daphnia magna* population exposed to 4.3 mg/L trivalent arsenic exhibited a 50 percent immobilization after 96 hours, and the amphipod *Gammarus pseudolimnaeus* experienced 50 percent immobilization following 96 hours of 0.96 mg/L trivalent arsenic exposure (Lima et al., 1984).

7.4.2.3 Barium

Barium, a silvery-white metal, is used in various alloys, in paints, soap, paper and rubber and in the manufacture of ceramics and glass. Two forms of barium, barium sulfate and barium carbonate, are often found in nature as underground ore deposits. Barium is relatively abundant in nature and is found in plants and animal tissue. Plants can accumulate barium from the soil.

Most of the barium that enters an animal's body is removed within a few days and almost all of it is gone within 1 to 2 weeks. Most of the barium that stays in the body goes into the bones and

teeth. Rats exposed to barium in their diet at lower doses, but for longer time periods, showed increased blood pressure and changes in the function and chemistry of the heart (ATSDR, 1992a).

Plants. Background concentrations of barium in various food and feed plants are reported to range from 1 to 198 mg/kg (dry weight) (Kabata-Pendias and Pendias, 1992). Concentrations are often highest in the leaves of cereals and legumes and lowest in grains and fruits (Kabata-Pendias and Pendias, 1992). The availability of barium to plants is greatly influenced by the pH of the soil, with barium more available under acidic soil conditions (Kabata-Pendias and Pendias, 1992). The concentration of barium in leaf tissue that has been reported as excessive or toxic to various plant species is 500 mg/kg (Kabata-Pendias and Pendias, 1992). A soil concentration of 500 mg/kg has been proposed by Efroymson et al. (1997) as a benchmark screening value for barium phytotoxicity.

Terrestrial Invertebrates. No information was found regarding the toxicity of barium to terrestrial invertebrates.

Mammals. Barium administered to rats via their drinking water at doses of 1, 10, and 100 mg/L had no effect on food or water consumption, or growth. Because the highest dose tested (100 mg/L) did not elicit any adverse effects, it was considered the NOAEL (5.1 mg/kg/day) (EPA, 2003b). Laboratory rat toxicity data for barium chloride in drinking water was used to calculate a NOAEL value of 5.1 mg/kg/day (Perry et al., 1983). Growth and hypertension were the test endpoints. Rats administered barium through oral gavage showed no adverse effects at dosages up to 300 mg/kg/day. These data were used to derive a NOAEL value of 19.8 mg/kg/day (Borzelleca et al., 1988). Mortality was the endpoint for this study.

Birds. A study of one-day old chicks fed barium in their diet were used to determine the amount of barium to cause mortality. While barium exposures up to 2,000 ppm produced no mortality, chicks in the 4,000 to 32,000 ppm exposure groups experienced 5 to 100 percent mortality. Because 2,000 ppm was the highest non-lethal dose, this dose was considered to be the subchronic NOAEL (Johnson et al., 1960). A NOAEL value of 20.8 mg/kg/day was derived from these data.

Aquatic Life. The chronic value for daphnids is from a 21-day test on *Daphnia magna* by Biesinger and Christensen (1972) which resulted in 16 percent reproductive impairment. The Tier II secondary acute water quality value and secondary chronic water quality value for

barium, as calculated by the method described in the EPA's *Final Water Quality Guidance for the Great Lakes System* (EPA, 1995), are 110 and 4.0 µg/L, respectively.

7.4.2.4 Beryllium

In environmental media, beryllium usually exists as beryllium oxide. Beryllium has limited solubility and mobility in sediment and soil.

Plants. Beryllium uptake by plants occurs when beryllium is present in the soluble form. The highest levels of beryllium are found in the roots, with lower levels in the stems and foliage (EPA, 1985b).

Soluble forms of beryllium are easily taken up by plants, probably in a manner similar to calcium and magnesium, but it is not readily translocated from roots to shoots (Peterson and Girling, 1981). Beryllium has been reported to inhibit seed germination, enzyme activation, and uptake of calcium and magnesium by roots. Common symptoms of beryllium toxicity to plants are brown, retarded roots, and stunted foliage (Romney and Childress, 1965). The phytotoxicity benchmark value for beryllium (10 mg/kg) is based on unspecified toxic effects on plants grown in surface soil amended with 10 mg/kg beryllium (Kabata-Pendias and Pendias, 1992).

Terrestrial Invertebrates. No information was found regarding the toxicity of beryllium to terrestrial invertebrates.

Mammals. The major exposure route for mammals is inhalation. Beryllium is poorly absorbed from the gastrointestinal tract, and is not absorbed through intact skin to any significant degree. Mammals exposed via inhalation exhibit pulmonary effects which may last long after exposure ceases. Based on animal studies, beryllium is poorly absorbed through both the gastrointestinal tract and the skin. The most important route of exposure for beryllium is inhalation, although absorption by this route does not appear to be extensive. Once beryllium is absorbed, it is circulated in the blood as an orthophosphate colloid and is then distributed primarily to the bone, liver, and kidneys in both humans and animals. Beryllium and its compounds are not biotransformed, but soluble beryllium compounds are partially converted to more insoluble forms in the lungs (Reeves and Vorwald, 1967).

Following inhalation of soluble beryllium compounds in both humans and animals, the lung appears to be the main target organ for toxicity. Acute exposure may cause chemical

pneumonitis; chronic exposure to insoluble forms may lead to chronic beryllium disease (berylliosis), a fibrotic lung disease (ATSDR, 1993b).

A variety of beryllium compounds have been demonstrated to cause pulmonary tumors following inhalation in animals. However, it is thought that oral administration does not lead to carcinogenesis due to poor absorption of the constituent from the gastrointestinal tract. The NOAEL for a rat lifetime chronic exposure to beryllium in drinking water was 0.54 mg/kg-day (Health Effects Assessment Summary Tables [HEAST], 1997).

Aquatic Life. Exposure routes for aquatic organisms include ingestion and gill uptake. Beryllium does not bioconcentrate in aquatic organisms. Beryllium uptake from water is low, resulting in low bioconcentration rates. Biomagnification of beryllium in aquatic food chains does not occur (Fishbein, 1981). Beryllium can be toxic to warm water fish, especially in soft water.

The Tier II secondary acute water quality value and secondary chronic water quality value for beryllium, as calculated by the method described in EPA (1995), are 35 and 0.66 µg/L, respectively.

The EC₂₀ for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration causing less than 20 percent reduction in either weight of young fish per initial female fish in a life-cycle or partial life-cycle test, or the weight of young per egg in an early life-stage test (Suter and Tsao, 1996). The EC₂₀ value for beryllium is 148 µg/L. A similar value can be determined for daphnids, which reflects the highest tested concentration causing less than 20 percent reduction in the product of growth, fecundity, and survivorship in a chronic test with a daphnid species. The EC₂₀ for daphnids is 3.8 µg/L (Suter and Tsao, 1996).

7.4.2.5 Cadmium

Cadmium is a silver-white, blue-tinged, lustrous metal. It is insoluble in water, although its chloride and sulphate salts are freely soluble (Windholz et al., 1976). Cadmium compounds in soil are stable and are not subject to degradation (ATSDR, 1993c).

Plants. Cadmium is a non-essential element for plant growth that can be absorbed from air or soil. The availability of cadmium in soil is influenced by organic content, clay content, pH, and redox potential (Sharma, 1980). The distribution of accumulated cadmium within plants is generally roots > stem and leaves > fruits, grains, seeds, or nutrient storage organs (Eisler, 1985a).

Concentrations from 10 to 20 mg/kg in plant tissue are expected to result in 10 percent loss in crop yield. Concentrations in leaf tissue that are excessive or toxic to various plant species range from 5 to 30 mg/kg. General symptoms of cadmium poisoning in plants include growth retardation, root damage, chlorosis of leaves, and red-brown coloration of leaf margins or veins (Kabata-Pendias and Pendias, 1992).

Terrestrial Invertebrates. A number of studies have been conducted to assess the potential adverse effects on terrestrial invertebrates. Van Gestel et al (1992) evaluated the effects of cadmium, added to the soil as CdCl₂, on growth and reproduction of *Eisenia andrei* after 21 days. A concentration of 18 mg/kg cadmium was required to reduce the number of cocoons produced per week and the number of juveniles per worm. Growth and reproduction were not affected at 10 mg/kg cadmium.

The effects of cadmium added to horse manure (as cadmium acetate) on *Eisenia fetida* was investigated by Malecki et al. (1982). Two growth periods were studied, 8 and 20 weeks, and survival, weight gain, and cocoon production were measured. The most sensitive parameter was cocoon production. In the 8-week test, the lowest concentration tested, 25 mg/kg Cd, caused a 52 percent decrease in cocoon production. In the 20-week test, the lowest concentration tested, 50 mg/kg Cd, caused a 24 percent decrease in cocoon production.

Using the results of the above tests and a number of other studies, Efrogmson et al. (1997) computed an earthworm benchmark concentration of 20 mg/kg.

Mammals. Cadmium uptake by most animals is relatively low. Absorption and retention of cadmium decreases with prolonged exposure. Cadmium tends to accumulate in the viscera, especially in liver and kidney tissues. The biological half-life of cadmium in muscle, kidney, and liver tissues ranges from 10 to 30 years. About 5 percent of ingested cadmium is absorbed, whereas between 10 and 50 percent of inhaled cadmium is absorbed (Friberg et al., 1986).

Mammals are relatively resistant to cadmium toxicity. The production of metallothioneins helps to protect mammals from cadmium toxicosis. The lowest published toxic dose of cadmium for mammals is 21.5 mg/kg, which was reported to induce adverse effects on fertility of rats (Registry of Toxic Effects of Chemical Substances [RTECS], 1996). Sutou et al. (1980) studied the effects of cadmium on rats exposed through oral gavage. While no adverse effects were observed at the 1 mg/kg/day dose level, fetal implantations were reduced by 28 percent, fetal survivorship was reduced by 50 percent, and fetal resorptions increased by 400 percent amongst

the 10 mg/kg/day exposure group. Because this study considered oral exposures during reproduction, the 1 and 10 mg/kg/day doses were considered to be chronic NOAELs and lowest-observed-adverse-effects-levels (LOAEL), respectively.

Exposure to cadmium has been shown to induce teratogenic effects in laboratory animals. The teratogenic effects of cadmium can be synergized by lead and mercury salts and antagonized by selenium (Ferm and Layton, 1981).

Birds. Birds, like mammals, are relatively resistant to cadmium toxicity. Sublethal effects in birds are similar to those in other species and include growth retardation, anemia, and testicular damage (Hammons et al., 1978). In one experiment, drake mallards were fed up to 200 mg/kg cadmium chloride in their diet for 90 days, and all survived with no loss in body weight. Egg production was, however, suppressed in mallard hens fed 200 mg/kg cadmium chloride in their diet (White and Finley, 1978). As in mammals, metallothionein binds to cadmium and thus aids in the reduction of a toxic response. Japanese quail that were fed 75 mg/kg cadmium in their diets developed signs of bone marrow hyperplasia, anemia, and hypertrophy of both heart ventricles after six weeks of exposure (Richardson et al., 1974). White and Finley (1978) studied the effects on reproduction of mallard ducks fed cadmium in their diets. Based on the results of this study, a NOAEL value of 1.45 mg/kg/day has been derived.

Aquatic Life. Freshwater organisms are more sensitive to elevated cadmium concentrations than are wildlife or plant species (Eisler, 1985a). Among all species of freshwater biota examined, cadmium concentrations of 0.47 to 5.0 µg/L were associated with decreases in standing crop, decreases in growth, inhibition of reproduction, immobilization, and population alterations (Eisler, 1985a). The EPA's National Recommended Water Quality Criteria are 2.0 µg/L for acute exposure and 0.25 µg/L for chronic exposure of freshwater aquatic life to cadmium (based on a water hardness of 100 mg/L) (EPA, 2002d). The test EC₂₀ (the concentration that will result in a specified effect on 20 percent of the test population) for fish can be used as a benchmark indicative of production within a population. The EC₂₀ value for cadmium is 1.8 µg/L (Suter and Tsao, 1996). The EC₂₀ value for cadmium with respect to daphnids is 0.75 µg/L (Suter and Tsao, 1996).

Concentrations of cadmium in freshwater biota from uncontaminated areas are usually less than 1.0 mg/kg in fish and less than 5 mg/kg in molluscs, annelids, and macrophytes (Eisler, 1985a). Bioconcentration factors of 2,000 have been reported in insects, fish, and algae.

7.4.2.6 Chromium

Chromium exists in soil primarily in the form of insoluble oxides with very limited mobility. In soil, chromium (+3) is readily hydrolyzed and precipitated as chromium hydroxide. In the aquatic phase, chromium may be in the soluble state or attached to clay-like or organic suspended solids.

Plants. Chromium does not play an essential role in plant metabolism. The concentration of chromium in terrestrial plants is controlled primarily by soluble chromium in the soil (Kabata-Pendias and Pendias, 1992). Chromium concentrations in plants are usually higher in roots than in leaves or shoots. Concentrations of chromium in leaf tissue that are excessive or toxic to various plant species range from 5 to 30 mg/kg soil (Kabata-Pendias and Pendias, 1992). General symptoms of chromium toxicity in plants include chlorosis of new leaves, necrotic spots and purpling tissues, and injured root growth (Kabata-Pendias and Pendias, 1992).

Terrestrial Invertebrates. Abbasi and Soni (1983) exposed the earthworm *Octochaetus pattoni* to chromium (as $K_2Cr_2O_7$) in soil for 60 days to assess the effects on survival and reproduction. Survival was the most sensitive endpoint with a 75 percent decrease resulting from exposure to 2.0 ppm chromium, the lowest concentration tested. The number of cocoons produced was not reduced until the concentration reached 20 ppm chromium (the highest concentration tested). The number of juveniles produced was not affected.

It is difficult to establish a benchmark concentration for chromium based on earthworm toxicity because the relative toxicity of Cr^{+3} and Cr^{+6} is not clear from the available data. Cr^{+6} ions can pass through cell membranes with much greater ease than Cr^{+3} ions. However, it is thought that Cr^{+6} is reduced to Cr^{+3} inside the cell (Molnar et al., 1989). Without a better understanding of chromium transformations in the soil, transport across earthworm cell membranes, and reactions within the cell, it is difficult to separate the effects of the two different forms of chromium. These difficulties notwithstanding, a soil benchmark value of 0.4 mg/kg has been suggested by Efrogmson et al. (1997), based on the work of Abbasi and Soni (1983).

Mammals. Chromium is a required element in animal nutrition. In general, hexavalent chromium compounds are more toxic than the trivalent chromium compounds. Adverse effects on blood and serum chemistry and morphological changes in liver have been reported in rabbits and rats exposed to chromium concentrations of 1.7 mg/kg/day for six weeks. Rats exposed to hexavalent chromium concentrations of 134 mg/L in drinking water over a two to three month period were found to develop lesions in kidney and liver tissues (Eisler, 1986).

Laboratory data based on rats exposed to chromium (as Cr₂O₃) in their diets were used to derive a NOAEL value for trivalent chromium of 2,737 mg/kg/day (Ivankovic and Preussmann, 1975). Reproduction and longevity were the endpoints in this study. Laboratory data based on rats exposed to chromium (as K₂Cr₂O₄) in their drinking water were used to derive a NOAEL value for hexavalent chromium of 3.28 mg/kg/day (MacKenzie et al., 1958). Body weight and food consumption were the endpoints in this study. Mammalian laboratory studies have shown chromium to be mutagenic, carcinogenic and teratogenic (Eisler, 1986).

Birds. Data on the effects of chromium to avian species is limited. No adverse effects were found in chickens exposed to 100 mg/kg dietary hexavalent chromium in a 32-day study (Rosomer et al., 1961). Haseltine et al. (1985) did not observe changes in survival, reproduction or blood chemistry following exposure of adult black ducks (*Anas rubripes*) to diets containing between 10 and 50 mg/kg chromium III (as CrK[SO₄]₂). Ducklings from this group that were fed the same chromium diet as the parent ducks experienced alterations in growth patterns and a reduction in survival. Based on these data, a NOAEL value of 1 mg/kg/day has been derived.

Aquatic Life. In freshwater systems, hexavalent chromium appears to be more toxic than the trivalent form. The National Recommended Water Quality Criteria for trivalent chromium are 570 µg/L for acute exposures and 74 µg/L for chronic exposures. The National Recommended Water Quality Criteria for hexavalent chromium are 16 µg/L for acute exposures and 11 µg/L for chronic exposures (EPA, 2002d). These values are based on a water hardness of 100 mg/L.

The test EC₂₀ for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration causing less than 20 percent reduction in either the weight of young fish per initial female fish in a life-cycle or partial life-cycle test, or the weight of young per egg in an early life-stage test. The EC₂₀ value for trivalent chromium is 89 µg/L and for hexavalent chromium is 51 µg/L (Suter and Tsao, 1996). A similar value can be determined for daphnids, which represents the highest tested concentration causing less than 20 percent reduction in the product of growth, fecundity, and survivorship in a chronic test with a daphnid species. The EC₂₀ benchmark for daphnids has been determined to be 0.5 µg/L for hexavalent chromium (Suter and Tsao, 1996).

7.4.2.7 Cobalt

Cobalt is a natural element and is widely distributed in the earth's crust at 0.001 to 0.002 percent (Merck Index, 1983). Small amounts of cobalt are found in rocks, soil, surface and groundwater.

Natural cobalt can stay airborne for a few days, but will stay for years in the soil. In most soils, the transfer of cobalt from soils to plants is not significant, although higher transfer rates have been observed in some higher plants and in acidic soils (Boikat et al, 1985; Francis et al., 1980). Some cobalt may seep from acid soil into groundwater. It is present in trace quantities in most foods and is readily absorbed by the gut in humans (International Commission on Radiological Protection [ICRP], 1979).

Plants. Although cobalt is essential to some blue-green algae, fungi, and microorganisms, it apparently is not essential for the growth of higher plants (Kabata-Pendias and Pendias, 1992). Several abiotic factors govern the availability of cobalt to plants. Soil factors include organic matter and clay content, pH, leachability, and concentration of manganese and iron oxides. Uptake of cobalt can occur via the roots or leaves of a plant (Kabata-Pendias and Pendias, 1992).

Concentrations of cobalt in leaf tissue that are excessive or toxic to various plant species range from 15 to 50 mg/kg (dry weight) (Kabata-Pendias and Pendias, 1992). A soil concentration of 20 mg/kg (dry weight) has been proposed by Efrogmson et al., (1997) as a benchmark screening value for cobalt phytotoxicity. General symptoms of cobalt toxicity in plants include interveinal chlorosis in new leaves followed by induced iron chlorosis and white leaf margins and damaged root tips (Kabata-Pendias and Pendias, 1992).

Terrestrial Invertebrates. No information was found regarding the toxicity of cobalt to terrestrial invertebrates.

Mammals. Cobalt is a component of vitamin B₁₂ and, therefore, is an essential micro-nutrient for animal growth. No information has been located at this time on chronic toxic effects of cobalt to terrestrial wildlife; however, some acute studies have been completed. Additionally, there is little biomagnification of cobalt in animals of higher trophic levels (Jenkins, 1980).

Young rats are unable to survive repeated 30 mg doses of cobalt metal powder in their diet for a month (total dosage about 900 mg), whereas they can tolerate 1,250 mg of the metal in a single dose (Venugopal and Luckey, 1978). Cobalt was embryotoxic to rat fetuses when it was administered during the entire gestation (dose of 0.05 mg/kg). A dose of 0.005 mg/kg was non-toxic to the females; however, the progeny of treated females had a reduced rate (Shepard, 1986). At doses under 2 milligrams per kilogram of body weight per day (mg/kg-bw/day), no adverse effects to sheep were noted. However, at 6 mg/kg-bw/day, sheep exhibited loss of appetite, loss of weight, and debilitation were noted (National Research Council [NRC], 1977).

Birds. No information has been located at this time on chronic toxic effects of cobalt to birds; however, some acute studies have been completed. Additionally, there is little biomagnification of cobalt in animals of higher trophic levels (Jenkins, 1980).

Chickens were administered 50 mg/kg of diet/day with acute effects of loss of appetite, loss of weight, and debilitation. At doses under 2 mg/kg-bw/day, no adverse effects to chickens were noted (NRC, 1977).

Aquatic Life. In most surface water bodies, cobalt is primarily associated with the sediment. However, some mobilization may occur in acidic water and in the presence of chloride ions or chelating agents. Bioaccumulation factors for freshwater fish range from 40 to 1,000 (Smith and Carson, 1981).

Research by Evans et al. (1988) indicates that cobalt does not significantly bioaccumulate in benthic bottom feeders.

7.4.2.8 Iron

Iron is an essential trace element, required as a constituent of oxygen-carrying and oxidative-reductive macro-molecules such as hemoglobin, myoglobin, and cytochrome P-450. As such, most iron-related health concerns are induced by insufficient iron intake, rather than excess iron intake (NRC, 1989).

Plants. Wallihan (1966) reported unspecified reductions in plant growth in a solution culture with the addition of 10 ppm iron. Wallace et al., (1977) evaluated the effects of iron (as FeSO₄) on leaf, stem, and root weights of bush bean seedlings grown for fifteen days in nutrient solution. Iron at 28 ppm reduced all three measures 67, 52, and 67 percent, respectively, while 11.2 ppm iron had no effect. After 55 days cabbage seedling plant weight was reduced 45 percent by 50 ppm iron added as FeSO₄ to nutrient solution, while 10 ppm had no effect on growth (Hara et al., 1976).

Iron is the key metal required for energy transformations needed for cellular function. It occurs in heme and non-heme proteins and is concentrated in chloroplasts. Organic iron complexes are involved in photosynthetic electron transfer. Plant symptoms of toxicity are not specific and differ among plant species and growth stages (Foy et al., 1978).

Terrestrial Invertebrates. No information was found regarding the toxicity of iron to terrestrial invertebrates.

Mammals. Iron is an essential nutrient for most wildlife species and is necessary to maintain homeostasis; therefore, it is only toxic at very high concentrations. Bioaccumulation factors have been calculated for several small mammal species. Small herbivorous mammals were estimated to have an iron bioaccumulation factor of 0.0127, and small omnivorous mammals were estimated to have an iron bioaccumulation factor of 0.01209. These bioaccumulation factors indicate that iron is not accumulated in small mammal tissues (Sample et al., 1998). Additionally, the bioaccumulation factor for earthworms has been estimated to be 0.038, indicating that iron is not accumulated in earthworm tissues (Sample et al., 1998).

Aquatic Life. The National Recommended Water Quality Criteria for iron (1,000 µg/L) is based on field study at a site receiving acid mine drainage (EPA, 2002d). The lowest chronic value for daphnids (158 µg/L) is a threshold for reproductive effects from a 21-day test using iron chloride with *Daphnia magna* (Dave, 1984). It is considerably lower than the 4,380 µg/L concentration causing 16 percent reproductive decrement in another test using iron chloride with *Daphnia magna* (Biesinger and Christensen, 1972). The lowest chronic value for fish (1,300 µg/L) is a concentration that caused 100 percent mortality in an embryo-larval test with rainbow trout exposed to dissolved iron salts (Amelung, 1981).

The Ontario Ministry of the Environment has prepared provincial sediment quality guidelines using the screening-level concentration approach, which estimates the highest concentration of a particular contaminant in sediment that can be tolerated by approximately 95 percent of benthic fauna (Neff et al., 1988). These values are based on Ontario sediments and benthic species from a wide range of geographical areas within the province (Persaud et al., 1993). The lowest effect level (Low) is the level at which actual ecotoxic effects become apparent. The severe effect level (Severe) represents contaminant levels that could potentially eliminate most of the benthic organisms (Persaud et al., 1993). The “Low” and “Severe” levels for iron in sediment are 2 percent (20,000 ppm) and 4 percent (40,000 ppm), respectively.

7.4.2.9 Lead

Global production of lead from both smelter and mining operations has been high throughout this century. Lead is commonly used in storage batteries as well as ammunition, solder, and casting materials. In addition, tetraethyl lead was a principal additive to gasolines as an anti-knock

agent, and was commonly used as an additive in paints. In short, lead is one of the most ubiquitous pollutants in the civilized world.

Lead is strongly sorbed in sediments and the rate is strongly correlated with grain size and organic content. In the absence of soluble complexing species, lead is almost totally adsorbed to clay particles at pHs greater than 6 (Moore and Ramamoorthy, 1984).

Plants. Although lead is not an essential nutrient for plant growth, it is detected in plant tissues due to the prevalence of lead in the environment. The bioavailability of lead in soil to plants is limited. Bioavailability may be enhanced by a reduction in soil pH, a reduction in the content of organic matter and inorganic colloids in soil, a reduction in iron oxide and phosphorous content, and by increased amounts of lead in soil (NRCC, 1973). Plants can absorb lead from soil and air. Aerial deposition of lead can also contribute significantly to the concentration of lead in above-ground plant parts. Lead is believed to be the metal of least bioavailability and the most highly accumulated metal in root tissue (Kabata-Pendias and Pendias, 1992).

Mean background concentrations of lead in grasses and clovers have been reported to range from 2.1 to 2.5 mg/kg (dry weight) (Kabata-Pendias and Pendias, 1992). Adverse effects of lead on terrestrial plants occur only at total concentrations of several hundred mg/kg of soil (Eisler, 1988b). This is explained by the fact that, in most cases, lead is tightly bound to soils and substantial amounts must accumulate before it can affect the growth of higher plants (Boggess, 1977).

Terrestrial Invertebrates. A number of studies have been conducted to assess the potential toxicity of lead to terrestrial invertebrates. Bengtsson et al. (1986) examined the effects of lead on *Dendrobaena rubida* at different soil acidities. After 4 months at pH 4.5, the number of cocoons produced per worm, hatchlings per cocoon, and percent hatched cocoons were reduced 75, 100 and 100 percent, respectively, by 500 mg/kg lead, while 100 mg/kg lead had no effect. At pH 5.5 and 6.5, lead had no adverse effects at any of the lead concentrations.

Spurgeon et al. (1994) kept adult *Eisenia fetida* in contaminated soil for 8 weeks to examine the effects of lead [as $Pb(NO_3)_2$] on survival and growth of the earthworms. After 56 days, the calculated LC_{50} was 3,760 mg/kg and the EC_{50} for cocoon production was 1,940 mg/kg.

Efroymson et al. (1997) have established a benchmark value of 500 mg/kg for earthworms based on the work of Bengtsson et al. (1986) which showed inhibition of reproduction at this concentration.

Mammals. As with plants, lead is not considered an essential nutrient for mammalian life. Ingestion is the major route of exposure for wildlife. Lead tends to accumulate in bone, hair, and teeth. Biomagnification of lead is negligible (Eisler, 1988b). Reduced survival was reported at acute oral doses as low as 5 mg/kg body weight in rats, at a chronic dose of 0.3 mg/kg body weight in dogs, and at a dietary level of 1.7 mg/kg body weight in horses (Eisler, 1988b). Laboratory data from studies of rats fed lead acetate in their diets were used to estimate a NOAEL value of 8.0 mg/kg/day (Azar et al., 1973). Reproduction was the endpoint for this study. Symptoms of lead poisoning in mammals are diverse and depend on the form of lead ingested, the concentration, and the species and its age. These symptoms may include reproductive impairment, decreased body weight, vomiting, uncoordinated body movements, visual impairment, reduced life span, renal disorders, and abnormal social behavior (Eisler, 1988b).

In laboratory studies, breeding mice exposed to low doses of lead in drinking water (25 ppm) resulted in loss of the strain in two generations with many abnormalities (Schroeder and Mitchner, 1971). Exposure of rats in this same experiment resulted in many early deaths and runts. Blood δ -aminolevulinic acid dehydratase (ALAD) activity associated with exposure to lead was reduced in white-footed mice living near a metal smelter (Beyer et al., 1985). Amounts of whole-body lead content and feeding habits of roadside rodents have been correlated with highest body burdens in insectivores such as shrews, intermediate in herbivores, and lowest in granivores (Bogges, 1977; Getz et al., 1977).

Birds. Most of the information on the effects of lead to terrestrial vertebrates is concerned with the poisoning of waterfowl by lead shot. Apparent symptoms include loss of appetite and mobility, avoidance of other birds, lethargy, weakness, emaciation, tremors, dropped wings, green feces, impaired locomotion, loss of balance and depth perception, nervous system damage, inhibition of heme synthesis, damage to kidneys and liver, and death (Eisler, 1988b; Mudge, 1983). Anemia, kidney disease, testicular and liver lesions, and neurological disorders have been associated with high brain lead concentrations in mourning doves (*Zenaida macroura*) (Kendall, 1992). Hatchlings of chickens, Japanese quail, mallards and pheasants are relatively tolerant to moderate lead exposure, including no effect on growth at dietary levels of 500 ppm and no effect on survival at 2,000 ppm (Hoffman et al., 1985).

Toxicity of lead to birds is dependent upon the form of lead, the route of exposure and exposure duration, and the species and age of the bird. Laboratory toxicity data for American kestrels fed metallic lead in their diet were used to estimate a NOAEL value of 3.85 mg/kg/day (Pattee, 1984). Reproduction was the endpoint for this study.

Aquatic Life. All life stages are sensitive to the toxic effects of lead; however, embryos are more sensitive to lead than are later juvenile stages (Davies et al., 1976). Lead uptake depends on exposure time, aqueous concentration, pH, temperature, salinity, diet, and other factors. For example, gill, liver, kidney, and erythrocytes accumulate lead from aqueous sources in proportion to exposure time and concentration (Holcombe et al., 1976). Direct erythrocyte injury is considered the first and most important sign of lead poisoning in catfish (Dawson, 1935). Respiratory distress occurs in fish living in rivers receiving lead mining wastes in England (Carpenter, 1924; 1925; 1926). Fish are thought to be asphyxiated as a result of a mucous coating over the gills (NAS, 1972).

No significant biomagnification of lead occurs in aquatic ecosystems (Boggess, 1977). Background concentrations of lead in fish tend to be less than 1 mg/kg (dry weight) (Eisler, 1988b). The EPA's National Recommended Water Quality Criteria for lead in freshwater is 65 µg/L for acute exposure and 2.5 µg/L for chronic exposure (EPA, 2002d). In general, dissolved lead is more toxic than total lead, and organic forms of lead are more toxic than inorganic forms. Soluble lead in the water column becomes less bioavailable as water hardness increases. Chronic exposure of fish to lead may result in signs of lead poisoning such as spinal curvature, anemia, darkening of the dorsal tail region, destruction of spinal neurons, difficulties in swimming, growth inhibition, changes in blood chemistry, retarded sexual development, and death (Eisler, 1988b).

The majority of benthic invertebrates do not bioconcentrate lead from water or abiotic sediment particles. There is some evidence of bioaccumulation through the food web of organic forms of lead, such as tetraethyl lead. Anderson et al., (1980) reported lead LC₅₀s of 258 ppm for the chironomid and that growth of this amphipod was not reduced above this level in freshwater sediments. In addition, Suter and Tsao (1996) reported effect levels in the water flea (*Daphnia magna*) to be in the 12.26 ppb range, while Khangrot and Ray (1989) reported a *D. magna* LC₅₀ of 4.89 ppm.

7.4.2.10 Manganese

Manganese, a silver-colored metal with chemical properties similar to iron, is a naturally occurring substance found in many minerals. Manganese is usually combined with oxygen, sulfur, and/or chlorine. Manganese is present in all living organisms and manganese is an essential element for adequate nutritional needs in mammals and many other organisms. Manganese is poorly absorbed from the intestinal tract; about 3-5% of the oral dose of manganese is absorbed. Absorption efficiency is also related to dietary intake of iron and calcium. Sufficient body stores of iron decrease absorption of manganese (ATSDR, 1992b).

Plants. Manganese is an essential element for plant growth. Uptake of manganese may occur via root or foliar uptake (Kabata-Pendias and Pendias, 1992). The concentration of manganese in plants is dependent upon plant and soil characteristics. Plants grown on flooded or acid soils tend to contain higher concentrations of manganese than plants grown in other, uncontaminated soils. In addition, concentrations of manganese in plants are positively correlated with soil organic matter (Kabata-Pendias and Pendias, 1992). Concentrations of manganese in leaf tissue that are excessive or toxic to various plant species range from 400 to 1,000 mg/kg dry weight (Kabata-Pendias and Pendias, 1992). A soil concentration of 500 mg/kg (dry weight) has been proposed by Efroymson et al., (1997) as a benchmark screening value for manganese phytotoxicity. General symptoms of manganese toxicity in plants include the presence of chlorosis and necrotic lesions on old leaves, blackish-brown or red necrotic spots, dried leaf tips, and stunted root and plant growth (Kabata-Pendias and Pendias, 1992).

Terrestrial Invertebrates. No information was found regarding the potential toxicity to terrestrial invertebrates from exposure to manganese.

Mammals. Manganese is an essential nutrient that is homeostatically regulated in vertebrates (Vanderploeg et al., 1975). Liver and kidney tissues generally contain the highest concentrations of manganese in the body. Manganese in the body is primarily excreted in the feces (Gregus and Klaassen, 1986).

Divalent manganese is more toxic than the trivalent form. Exposure to manganese dust via inhalation is usually of greater toxicological concern than ingestion (Klaassen et al., 1991). Laboratory data for rats fed manganese oxide in their diet were used to estimate a NOAEL value of 88 mg/kg/day (Laskey et al., 1982). Reproduction was the endpoint for this study. Laboratory studies with rats have found no hematologic, behavioral, or histologic effects in

animals exposed to manganese dioxide at concentrations of 47 mg/m³ for five hours per day, five days a week, for 100 days (Klaassen et al., 1991).

Birds. Japanese quail were studied by Laskey and Edens (1985) to determine the toxic effects of 5,000 ppm manganese fed to one-day old chicks. Growth and aggressive behavior were studied throughout the 75-day exposure period. While no reduction in growth was observed, aggressive behavior was reduced in 25 to 50 percent of the chicks relative to controls. However, reduced aggressive behavior was not considered a significant adverse affect. A NOAEL value of 977 mg/kg/day was derived from this study.

Aquatic Life. As discussed previously, manganese is a required nutrient for plant and animal life. Manganese concentrations in most vertebrates are homeostatically controlled (Vanderploeg et al., 1975). Bioconcentration factors for freshwater macrophytes have been reported to range from 190 to approximately 25,000 (Vanderploeg et al., 1975). With regard to freshwater fish, concentrations of manganese in fish muscle are generally less than 0.5 mg/kg and range from 3 to 10 mg/kg in whole fish (Vanderploeg et al., 1975). Bioconcentration factors from water to whole fish range from 40 to 2,300. A bioconcentration factor of 10,000 was also suggested for crustaceans (Vanderploeg et al., 1975).

No Federal water quality criteria exist for the protection of freshwater biota from elevated manganese concentrations. Suter and Tsao (1996) have estimated acute and chronic advisory levels for manganese to be 1,470 and 80.3 µg/L, respectively. The EC₂₀ for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration causing less than 20 percent reduction in either weight of young fish per initial female fish in a life-cycle or partial life-cycle test, or the weight of young per egg in an early life-stage test (Suter and Tsao, 1996). The EC₂₀ value for manganese is 1,270 µg/L. A similar value can be determined for daphnids, which reflects the highest tested concentration causing less than 20 percent reduction in the product of growth, fecundity, and survivorship in a chronic test with a daphnid species. The EC₂₀ for daphnids is less than 1,100 µg/L (Suter and Tsao, 1996).

7.4.2.11 Mercury

Mercury is a toxic compound with no known natural biological function. Mercury exists in three valence states: mercuric (Hg²⁺), mercurous (Hg¹⁺), and elemental (Hg⁰⁺) mercury. It is present in the environment in inorganic and organic forms. Inorganic mercury compounds are less toxic than organomercury compounds; however, the inorganic forms are readily converted to organic

forms by bacteria commonly present in the environment. The organomercury compound of greatest concern is methylmercury (EPA, 1999).

Mercury sorbs strongly to soil and sediment. Elemental mercury is highly volatile. In aquatic and terrestrial receptors, some forms of mercury, especially organomercury compounds, bioaccumulate significantly and biomagnify in the food chain. In all receptors, the target organs are the kidney, and central nervous system. However, mercury causes numerous other effects including teratogenicity and mutagenicity (EPA, 1999).

Plants. Mercury is not required for plant growth. Background concentrations of mercury in plants usually range from 0.0026 to 0.086 mg/kg (dry weight) (Kabata-Pendias and Pendias, 1992). Pine needles have been reported to be good biomonitors of mercury-contaminated environments (Kabata-Pendias and Pendias, 1992). In general, the concentration of mercury in plants will be elevated when mercury concentrations in soils are high. Mercury concentrations in plants, however, generally do not exceed those in associated soils (Lisk, 1972). Methyl mercury is more available to plants than either phenyl- or sulfide-mercury. In addition to mercury uptake from the soil, plants can also absorb mercury vapor (Browne and Fang, 1978).

Concentrations of mercury in leaf tissue that are excessive or toxic to various plant species range from 1 to 3 mg/kg (dry weight) (Kabata-Pendias and Pendias, 1992). A soil concentration of 0.3 mg/kg has been proposed by Efroymson et al., (1997) as a benchmark screening value for mercury phytotoxicity. General symptoms of mercury toxicity in plants include severe stunting of seedlings and roots and leaf chlorosis and browning of leaf points (Kabata-Pendias and Pendias, 1992).

Terrestrial Invertebrates. Abbasi and Soni (1983) exposed earthworms (*Octochaetus pattoni*) to mercury (as HgCl) to assess the effect on reproduction and growth. Survival and cocoon production were reduced at 0.5 ppm mercury, the lowest concentration tested. The number of juveniles produced was not affected. Based on these test data, a benchmark value for mercury in soil of 0.1 mg/kg has been proposed by Efroymson et al., (1997).

Mammals. Mercury is not an essential element for animal life. Background mercury concentrations in wildlife tend to be less than 1.0 mg/kg (wet weight) (Eisler, 1987). Biomonitoring studies have shown that mercury concentrations in mammals are highest in hair, followed by kidney and liver tissues (Bull et al., 1977; Klaassen, 1991; Wren, 1986). Mercury is bioaccumulated and biomagnified in terrestrial food chains (Eisler, 1987; Talmage and Walton,

1993). Talmage (1989) has shown the insectivorous shorttail shrew (*Blarina brevicauda*) to be a better monitor of environmental mercury contamination than the granivorous white-footed mouse (*Peromyscus leucopus*). Mink (*Mustela vison*) and river otter (*Lutra canadensis*) have been shown to be good monitors of mercury contamination within river environments due to their consumption of contaminated fish (Kucera, 1983).

Organic mercury compounds, especially methyl mercury, are more toxic to mammals than inorganic forms of mercury. Selenium has been shown to have a protective effect against mercury poisoning (Ganter et al., 1972). Based on laboratory data for methylmercury fed to rats and mink in their diets, a NOAEL value of 0.015 has been derived. This NOAEL is based on mortality, weight loss, reproduction, and ataxia as endpoints (Wobeser et al., 1976).

Mercury has been shown to be teratogenic, mutagenic, and carcinogenic in animal studies (Eisler, 1987). Signs of mercury poisoning that have been observed in mink include anorexia, weight loss, ataxia and splaying of hind legs, irregular vocalization, salivation, and convulsions (Wren, 1986).

Birds. Concentrations of mercury that are acutely toxic to birds following oral exposure range from 2.2 to 31 mg/kg body weight (Eisler, 1987). Mercury concentrations in the livers of methylmercury-poisoned birds ranged from 17 to 70 mg/kg (dry weight) (Solonen and Lodenius, 1984). Methylmercury is more toxic to avian species than inorganic mercury (Hill, 1981). In addition to the form of mercury to which the bird is exposed, the species, gender, age, and health of the individual may also influence the toxic response (Fimreite, 1979). Physical signs of mercury poisoning in birds include muscular incoordination, falling, slowness, fluffed feathers, calmness, withdrawal, hyporeactivity, and eyelid drooping (Eisler, 1987).

Japanese quail were fed mercury in their diet for one year to study the effects on reproduction. Egg production increased with increasing mercury dose, while fertility and hatchability decreased. Adverse effects of mercury exposure were evident at the 8 mg/kg dose level. Based on the results of this study a NOAEL value of 0.45 mg/kg/day has been derived (Hill and Schaffner, 1976). Mallard ducks fed methyl mercury dicyandiamide in their diets produced fewer eggs and fewer ducklings at exposure levels as low as 0.5 mg/kg. A NOAEL value of 0.0064 mg/kg/day was derived from these data, with reproduction the endpoint studied (Heinz, 1979).

Aquatic Life. Concentrations of mercury in freshwater fish collected from 12 monitoring stations in the United States from 1978 to 1981 ranged from 0.1 to 1.1 mg/kg (wet weight), with an average of 0.11 mg/kg (Lowe et al., 1985). Elevated concentrations of mercury in fish have often been associated with low pH, low calcium concentrations in the water, and low water hardness (Eisler, 1987). Methylating bacteria in sediments actively convert inorganic mercury into methylmercury. This results in an increase in the bioavailability of mercury. Fish absorb methylmercury more easily than inorganic mercury from the water column (Huckabee et al., 1979). Because exposure of fish to methylmercury can occur via ingestion of contaminated prey, methylmercury concentrations are usually highest in organisms near the top of the food chain, such as carnivorous fish (Huckabee et al., 1979).

Exposure of aquatic organisms to elevated mercury concentrations can result in reduced growth and reproduction (Eisler, 1987). The National Recommended Water Quality Criteria for acute and chronic exposure to mercury in freshwater systems are 1.4 and 0.77 µg/L, respectively (EPA, 2002d). The test EC₂₀ for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration causing less than 20 percent reduction in either the weight of young fish per initial female fish in a life-cycle or partial life-cycle test, or the weight of young per egg in an early life-stage test. The EC₂₀ value for methylmercury is less than 0.03 µg/L (Suter and Tsao, 1996). A similar value can be determined for daphnids, which represents the highest tested concentration causing less than 20 percent reduction in the product of growth, fecundity, and survivorship in a chronic test with a daphnid species. The EC₂₀ benchmark for daphnids has been determined to be 0.87 µg/L (Suter and Tsao, 1996).

Physical signs of acute mercury poisoning in fish include the flaring of gills, an increase in the frequency of respiratory movements, loss of equilibrium, and sluggishness (Armstrong, 1979).

7.4.2.12 Nickel

Nickel is a naturally occurring silvery metal that is found in the Earth's crust in the form of various nickel minerals. Exposure of organisms to nickel and its compounds results from breathing air, ingesting water and food that contain nickel and compounds, and skin contact with a media contaminated with nickel.

Plants. Nickel is not believed to be an essential element for plant growth; however, beneficial effects of nickel have been reported on the growth of legumes. Background concentrations of nickel in grasses and clovers collected in the United States averaged 0.13 and 1.5 mg/kg,

respectively (Kabata-Pendias and Pendias, 1992). The concentration of nickel in plants is positively correlated with nickel concentrations in soil.

Concentrations of nickel in leaf tissue that are excessive or toxic to plant species range from 10 to 100 mg/kg (dry weight) (Kabata-Pendias and Pendias, 1992). A soil concentration of 30 mg/kg has been proposed by Efroymson et al. (1997) as a benchmark screening value for nickel phytotoxicity. General symptoms of nickel toxicity in plants include the presence of interveinal chlorosis in new leaves, gray-green leaves, and brown and stunted root and plant growth. The uptake of nutrients and minerals, especially iron, can be substantially reduced as a consequence of nickel toxicity in plants (Kabata-Pendias and Pendias, 1992).

Terrestrial Invertebrates. The effects of nickel on *Eisenia fetida* were investigated by Malecki et al. (1982). The most sensitive endpoint was cocoon production. In the 8-week test, 300 mg/kg nickel caused a 41 percent decrease in cocoon production, while 200 mg/kg had no adverse effect. In the 20-week test, 200 mg/kg nickel caused a 23 percent decrease in cocoon production, while 100 mg/kg had no effect.

Neuhauser et al. (1984) evaluated the effects of soluble forms of nickel on growth and reproduction in *Eisenia fetida*. After 6 weeks cocoon production was decreased 33 percent by 250 mg/kg nickel, the lowest concentration tested. Growth was not affected until 500 mg/kg nickel was added to the substrate.

Efroymson et al. (1997) have established a soil benchmark value for nickel of 200 mg/kg based on the work of Malecki et al. (1982), which showed inhibition of reproduction at this concentration.

Mammals. Nickel is a nonessential element for animal life. Nickel concentrations within the whole bodies of small mammals from uncontaminated sites were reported to range from 2.2 to 6.2 mg/kg (dry weight) (Talmage and Walton, 1991). Highest concentrations were measured in the deer mouse (*Peromyscus maniculatus*). Highest tissue concentrations of nickel are usually found in the liver of mammals (Schroeder et al., 1964). Because nickel is poorly absorbed by the gastrointestinal tract, ingested nickel is generally not of great toxicological concern. Inhaled nickel, however, is relatively toxic. Rats fed nickel in their diet as nickel sulfate hexahydrate over three generations were studied for effects on reproduction. They were fed three dose levels (250, 500, and 1,000 ppm Ni) in their diet, and only the highest dose level caused reduced offspring body weights. No adverse effects were observed at the other dose levels. Because this

study considered exposures over multiple generations, the 500 ppm dose was considered to be the chronic NOAEL, and the 1,000 ppm dose was considered to be the chronic LOAEL (EPA, 2002d). These data were used to derive a NOAEL value of 40 mg/kg/day (Ambrose et al, 1976). Reproduction was the endpoint studied.

Birds. Mallard ducklings were fed nickel as nickel sulfate in their diet for a duration of 90 days to study the effects on mortality, growth, and behavior. They were fed three dose levels (176, 774, and 1,069 ppm Ni), and only the highest dose reduced growth and resulted in 70 percent mortality. Because the study considered exposure over 90 days, the 774 ppm dose was considered to be the chronic NOAEL, and the 1,069 dose was considered to be the chronic LOAEL (Cain and Pafford, 1981). A NOAEL value of 77.4 mg/kg/day was derived from this study based on mortality, growth, and behavior as endpoints.

Aquatic Life. The bioavailability and toxicity of nickel to aquatic biota is influenced by the pH of the water (Schubauer-Berigan et al., 1993). The national recommended water quality criteria for the protection of aquatic life for acute and chronic exposure are 470 and 52 µg/L, respectively (EPA, 2002d). Background concentrations of nickel in adult anurans ranged between 0.9 and 2.9 mg/kg (dry weight) (Hall and Mulhern, 1984).

The test EC₂₀ for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration causing less than 20 percent reduction in either the weight of young fish per initial female fish in a life-cycle or partial life-cycle test, or the weight of young per egg in an early life-stage test. The EC₂₀ value for nickel is 62 µg/L (Suter and Tsao, 1996). A similar value can be determined for daphnids which represents the highest tested concentration causing less than 20 percent reduction in the product of growth, fecundity, and survivorship in a chronic test with a daphnid species. The EC₂₀ benchmark for daphnids has been determined to be 45 µg/L (Suter and Tsao, 1996).

7.4.2.13 Selenium

Selenium is distributed widely in nature and is found in most rocks and soils at concentrations between 0.1 and 2.0 mg/kg (Fishbein, 1981). The primary factor determining the fate of selenium in the environment is its oxidation state. Selenium is stable in four valence states (-2, 0, +4, and +6) and forms chemical compounds similar to those of sulfur. The selenides (-2) are insoluble in water, as is elemental selenium. The inorganic alkali selenites (+4) and the selenates (+6) are soluble in water and are, therefore, more bioavailable.

Conditions such as pH, Eh, and the presence of metal oxides affect the partitioning of the various compounds of selenium in the environment. In general, elemental selenium is stable in soils and is found at low levels in water because of its ability to co-precipitate with sediments. The soluble selenates are readily taken up by plants and converted to organic compounds such as selenomethionine, selenocysteine, dimethyl selenide, and dimethyl diselenide. Selenium is bioaccumulated by aquatic organisms and may also biomagnify in aquatic organisms.

Plants. The role of selenium in plant growth is not fully understood. It is generally not considered essential in plant nutrition (Kabata-Pendias and Pendias, 1992). The concentration of selenium in plants has been shown to be positively correlated with the concentration of selenium in soil. Soil parameters such as pH, oxidation-reduction potential, and moisture content determine the amount of selenium available for plant uptake. Concentrations of selenium in leaf tissues that have been shown to be toxic to various plant species range from 5 to 30 mg/kg (Kabata-Pendias and Pendias, 1992). General symptoms of selenium toxicity in plants include the signs of interveinal chlorosis or black spots in plants containing approximately 4 mg/kg selenium, complete bleaching or yellowing of younger leaves at higher concentrations, and the presence of pinkish spots on roots (Kabata-Pendias and Pendias, 1992).

Terrestrial Invertebrates. Fischer and Koszorus (1992) tested the effects of 77 ppm of selenium on growth and reproduction of *Eisenia fetida* when added to a combination of peaty marshland soil and horse manure (1:1). The number of survivors and their live mass and number of cocoons produced were measured. The number of cocoons produced per worm showed the highest sensitivity to selenium with a 69 percent reduction at a selenium concentration of 77 ppm. Efrogmson et al. (1997) have proposed a soil benchmark value of 70 ppm based on this study.

Mammals. Selenium is an essential trace element for animal life. Concentrations that are essential to animals are in the range of 0.05 to 0.1 mg/kg in the diet (Arthur et al., 1992). According to Ganther (1974), selenium concentrations in healthy, unexposed, laboratory animals and livestock range between 0.1 and 1 mg/kg. Selenium offers a protective effect against some carcinogens such as benzo(a)pyrene and benzo(a)anthracene (Hammond and Beliles, 1980). Selenium also functions as an antidote to the toxic effects of mercury, thallium, copper, arsenic and cadmium (Frost and Lish, 1975).

Acute poisoning has been reported in livestock that consumed plant material containing 400 to 800 mg/kg selenium (Eisler, 1985b). Signs of acute poisoning in livestock include abnormal

movements, lowered head, drooped ears, diarrhea, elevated temperature, rapid pulse, labored breathing, bloating with abdominal pain, increased urination, and dilated pupils (Eisler, 1985b). Chronic poisoning may occur in animals exposed to dietary selenium concentrations between 1 and 44 mg/kg (Eisler, 1985b). Laboratory data from studies using rats fed potassium selenate (SeO_4) in their drinking water were used to derive a NOAEL value of 0.2 mg/kg/day (Rosenfeld and Beath, 1954). Reproduction was the endpoint in this study.

Birds. Toxicity from selenium has also been documented in birds. The major toxic effect of selenium on avian species is on reproductive success. Both sodium selenite and selenomethionine have been reported to be embryotoxic and teratogenic (Heinz et al., 1987). Reproductive impairment is likely to occur as concentrations of selenium approach 5 mg/kg. Mortality in mallard ducklings does not occur until selenium concentrations in the diet reach 40 mg/kg. While consumption of 1, 5, or 10 ppm selenium in the diet of mallard ducks had no effect on weight or survival of adults, 100 ppm selenium reduced adult survival and 25 ppm selenium reduced duckling survival. Consumption of 10 or 25 ppm selenium in the diet resulted in significantly larger frequency of lethally deformed embryos compared to the 1 or 5 ppm selenium exposures. These data were used to derive a NOAEL value of 0.5 mg/kg/day, with reproduction as the critical endpoint (Heinz et al., 1987).

Screech owls fed selenium in their diets were studied for the effects on reproduction. While exposure of owls to 0.44 mg/kg/day of selenomethionine had no adverse effects on reproduction, exposure to 1.5 mg/kg/day reduced egg production, hatchability, and nestling survival (Sample et al., 1996). A NOAEL value of 0.44 mg/kg/day was derived from these data based on reproductive effects.

Black-crowned night herons fed selenium their diets for 94 days were studied for the effects on reproduction. Exposure to 1.8 mg/kg/day selenium had no adverse effects on reproduction. These data were used to derive a NOAEL value of 1.8 mg/kg/day based on reproduction as the critical endpoint (Sample et al., 1996).

Aquatic Life. Selenium is an essential micro-nutrient for fish. Dietary requirements of selenium for fish range from 0.07 to 0.25 mg/kg, depending on the fish species (Gatlin and Wilson, 1984). The bioconcentration of selenium from water is highly dependent on the species of selenium present. Laboratory studies have shown bioconcentration factors for selenomethionine to be greater than those for selenite and selenate. Bioconcentration factors for

aquatic biota exposed to 1 µg/L selenomethionine were approximately 16,000 for algae, 200,000 for daphnids, and 5,000 for bluegills (Besser et al., 1993).

The EPA's National Recommended Water Quality Criteria for Priority Toxic Pollutants for selenium in freshwater is 5 µg/L for chronic exposure (EPA, 2002d). The toxicity of selenium to freshwater fish appears to be correlated more closely with dietary than waterborne exposure (Coyle et al., 1993). Sulfate concentrations in water may also influence the toxicity of selenium to aquatic invertebrates (Maier et al., 1993).

The test EC₂₀ for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration causing less than 20 percent reduction in the weight of young fish per initial female fish in a life-cycle or partial life-cycle test, or the weight of young per egg in an early life-stage test. The EC₂₀ for selenium is 40 µg/L (Suter and Tsao, 1996). A similar value can be determined for daphnids, which reflects the highest tested concentration causing less than 20 percent reduction in the product of growth, fecundity, and survivorship in a chronic test with a daphnid species. The EC₂₀ benchmark for daphnids is 25 µg/L selenium (Suter and Tsao, 1996).

7.4.2.14 Vanadium

Vanadium, a white to gray metal, occurs naturally in fuel oils and coal. It is used as a catalyst in the production of various chemicals, including sulfuric acid. It is also used in the hardening of steel, the manufacture of pigments, and in photography. The general population and many ecological receptors are exposed to background levels of vanadium primarily through ingestion of food.

Plants. There is some controversy over whether vanadium is an essential element for plants (Kabata-Pendias and Pendias, 1992). It appears to be required by some algal species and may be required by nitrogen-fixing bacteria. Mean background concentrations of vanadium in plants are 1.6 mg/kg for angiosperms, 0.69 mg/kg for gymnosperms, and 0.67 mg/kg for fungi (Waters, 1977). The availability of vanadium to plants is highly dependent on soil pH. Elevated levels of vanadium in soil can reduce the uptake of manganese, copper, calcium, and phosphorus (NRCC, 1980).

Concentrations of vanadium in leaf tissue that are excessive or toxic to various plant species range from 5 to 10 mg/kg (dry weight) (Kabata-Pendias and Pendias, 1992). A soil

concentration of 2 mg/kg has been proposed by Efroymson et al. (1997) as a benchmark screening value for vanadium phytotoxicity.

Terrestrial Invertebrates. No information was found regarding the toxicity of vanadium to terrestrial invertebrates.

Mammals. Vanadium has been shown to be essential in the diets of rats (Waters, 1977). Background concentrations of vanadium in the kidneys and livers of wild mammals have been reported to range from 0 to 2.07 mg/kg, and from 0 to 0.94 mg/kg, respectively (Waters, 1977). Liver and skeletal tissues usually contain the highest concentrations of vanadium (Waters, 1977). Experimental animal investigations have suggested that the liver, adrenal, and bone marrow may be adversely affected by subacute exposure to high levels of vanadium (ATSDR, 1992c; Klaassen et al., 1991). Vanadium fed to rats prior to gestation, during gestation, and through delivery and lactation was studied for effects on reproduction. The rats were fed three dose levels (5, 10, and 20 mg NaVO₃/kg/day or 2.1 mg V/kg/day). Significant differences in reproductive parameters (e.g., number of dead young per litter, size and weight of offspring) were observed at all dose levels. Therefore, the lowest dose was considered to be a chronic LOAEL. A chronic NOAEL was estimated by applying an uncertainty factor of 0.1 (chronic NOAEL = 0.21 mg V/kg/day) (EPA, 2002d).

Based on oral intubation exposure of rats to sodium metavanadate by Domingo et al. (1986), an estimated NOAEL value of 0.21 mg/kg/day has been derived. Reproduction was the endpoint for this study. Signs of acute toxicity in animals include alterations in nervous system responses, gastrointestinal distress, hemorrhaging, paralysis, convulsions, and respiratory depression (Klaassen et al., 1991).

Birds. Mallard ducks were fed vanadium as vanadyl sulfate in their diet for 12 weeks and observed for effects on mortality, body weight, and blood chemistry. The ducks were fed three different doses (2.84, 10.36, and 110 ppm V). No effects were observed at any of the dose levels. Because this study was greater than ten weeks in duration and did not consider a critical life stage, the maximum dose was considered to be a chronic NOAEL (White and Dieter, 1978). From these data a NOAEL value of 11.4 mg/kg/day has been estimated.

Aquatic Life. Background concentrations of vanadium in freshwater fish are usually less than 2.5 mg/kg (wet weight) (Jenkins, 1980). A bioconcentration factor of 3,000 has been reported for aquatic invertebrates exposed to vanadium (Neumann, 1976). No federal ambient water

quality criteria exist for the protection of freshwater biota (EPA, 2002d). The test EC₂₀ for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration causing less than 20 percent reduction in either the weight of young fish per initial female fish in a life-cycle or partial life-cycle test, or the weight of young per egg in an early life-stage test. The EC₂₀ value for vanadium is 41 µg/L (Suter and Tsao, 1996). A similar value can be determined for daphnids that represents the highest tested concentration causing less than 20 percent reduction in the product of growth, fecundity, and survivorship in a chronic test with a daphnid species. The EC₂₀ benchmark for daphnids has been determined to be 430 µg/L (Suter and Tsao, 1996).

7.4.2.15 Zinc

Zinc is a naturally occurring element which may be found in both organic and inorganic forms and, as such, is commonly found in the environment. In general, zinc is concentrated in the sediments of water bodies. NAS (1977b) has reported that zinc will probably be detected in 75 percent of all water bodies examined for the compound at various locations. The fate of zinc in soils appears to have a pH basis. Studies have shown that a pH of less than 7 often favors zinc desorption (EPA, 1984).

Plants. Background concentrations of zinc in terrestrial plants range from 25 to 150 mg/kg (dry weight) (NAS, 1979). The deficiency content of zinc in plants is between 10 and 20 ppm (dry weight). Roots often contain the highest concentrations of zinc (Kabata-Pendias and Pendias, 1992).

Certain species of plants, particularly those from the families *Caryophyllaceae*, *Cyperaceae*, and *Plumbaginaceae*, and some tree species are extremely tolerant to elevated zinc concentrations (Kabata-Pendias and Pendias, 1992). Concentrations of zinc in these plants may reach 1 percent (dry weight) in the plant. Concentrations in leaf tissue that are excessive or toxic to various plant species range from 100 to 400 mg/kg. Concentrations of 100 to 500 mg/kg are expected to result in a 10 percent loss in crop yield (Kabata-Pendias and Pendias, 1992). General symptoms of zinc toxicity in plants include the presence of chlorotic and necrotic leaf tips, interveinal chlorosis in new leaves, retarded growth of the entire plant, and injured roots that resemble barbed wire (Kabata-Pendias and Pendias, 1992).

Terrestrial Invertebrates. Spurgeon and Hopkin (1996) exposed the earthworm *Eisenia fetida* to zinc in soils with differing organic matter content and soil pH. The EC₅₀ concentrations for cocoon production in soils with a pH of 7.0 and 5 percent, 10 percent, and 15 percent organic

matter were 136, 462, and 592 mg/kg, respectively. The EC₅₀ concentrations for cocoon production in soils of pH 6.0 and 5 percent, 10 percent, and 15 percent organic matter were 199, 343, and 548 mg/kg, respectively. The EC₅₀ concentrations for cocoon production in soils of pH 5.0 and 5 percent, 10 percent, and 15 percent organic matter were 142, 189, and 230 mg/kg, respectively. Mortality was observed at higher zinc concentrations. A decrease in pH and/or organic matter content in soil led to a lower toxic concentration of zinc.

Neuhauser et al., (1985) determined an LC₅₀ for zinc in soil using adult *Eisenia fetida* exposed for 14 days. The calculated LC₅₀ was 662 mg/kg. Data from the preceding studies were used to derive a soil benchmark value for zinc in soil of 100 mg/kg (Efroymsen et al., 1997).

Mammals. Zinc is an essential trace element for normal fetal growth and development. However, exposure to high levels of zinc in the diet has been associated with reduced fetal weights, altered concentrations of fetal iron and copper, and reduced growth in offspring (Cox et al., 1969). Poisoning has been observed in ferrets and mink from chewing corroded galvanized cages (Clark et al., 1981). Symptoms of zinc toxicity are lassitude, slower tendon reflexes, bloody enteritis, diarrhea, lowered leukocyte count, depression of the central nervous system, and paralysis of the extremities (Venugopal and Luckey, 1978). A study by Kinnamon (1963) showed a NOAEL for oral exposure to a zinc compound over a period of 73 days to be 250 mg/kg body weight, and mice given 500 mg/L of zinc, as zinc sulfate, in drinking water have shown hypertrophy of the adrenal cortex and pancreas. Young animals are much more susceptible to poisoning by zinc than are mature animals (Clark et al., 1981).

Animals are quite tolerant to high concentrations of zinc in the diet. Levels 100-times that required in the diet usually do not cause detectable symptoms of toxicosis (NAS, 1979). Laboratory data for rats exposed to zinc oxide in their diet were used to estimate a NOAEL value of 160 mg/kg/day (Schlicker and Cox, 1968). Reproduction was the endpoint studied. Symptoms of zinc poisoning in mammals include lameness, acute diarrhea, and vomiting (Eisler, 1993).

Birds. Dietary zinc concentrations of greater than 2,000 mg/kg diet are known to result in reduced growth of domestic poultry and wild birds (Eisler, 1993). Reduced survival has been documented at zinc concentrations greater than 3,000 mg/kg diet or at a single dose of greater than 742 mg/kg body weight (Eisler, 1993). Laboratory data for white leghorn hens exposed to zinc sulfate in their diet for 44 weeks were used to estimate a NOAEL value of 14.5 mg/kg/day (Stahl et al., 1990). Reproduction was the endpoint for this study. A value of 51 mg/L has been

calculated as the NOAEL for chronic exposure of birds to zinc carbonate in drinking water (Sample et al., 1996).

Aquatic Life. Zinc residues in freshwater and marine fish are generally much lower than those found in algae and invertebrates. Thus there is little evidence for accumulation (Moore and Ramamoorthy, 1984). Rainbow trout (*Oncorhynchus mykiss*) have the ability to detect and avoid areas of water containing 5.6 ppb zinc (Sprague, 1968). Cairns and Scheier (1968) reported 96-hour LC₅₀s ranging from 10.13 to 12.5 ppm in hard water for bluegills (*Lepomis macrochirus*), and 96-hour LC₅₀s ranging from 2.86 to 3.78 ppm in soft water. These results demonstrate that water hardness affects the toxicity of zinc to fish. Chronic toxicity tests have been conducted with five species of freshwater fish. Chronic values ranged from 47 micrograms per liter (µg/L) for flagfish (*Jordanella floridae*) to 852 µg/L for brook trout (*Salvenius fontinalis*) (EPA, 1980b).

Acute toxicity to freshwater invertebrates is relatively low, and as with other metals, increasing water hardness decreases the toxicity of zinc (Moore and Ramamoorthy, 1984). As reported by Baudouin and Scoppa (1974), the 48-hour LC₅₀ for the cladoceran *Daphnia hyalina* was 0.055 mg/L, and 5.5 mg/L for the copepod *Cyclops abyssorum*. Four chronic toxicity tests are reported for *Daphnia magna*, with chronic values ranging from 47 µg/L to 136 µg/L (EPA, 1980b). Chronic testing with the saltwater species *Mysidopsis bahia* resulted in a chronic value of 166 µg/L (EPA, 1980b).

7.4.2.16 Chlordane

Chlordane is a chlorinated pesticide whose commercial use has been banned in the United States since 1988 (EPA, 1988). Technical chlordane is a mixture of some 50 different compounds; the major constituents being alpha-, beta-, and gamma-chlordane (Howard, 1991). Chlordane is generally immobile in soil; however, movement to groundwater can occur (Howard, 1991). The compound is biodegraded at a very slow rate (Howard, 1991). In aquatic systems, chlordane is not expected to undergo significant hydrolysis, oxidation, or direct photolysis (Howard, 1991). Because of its widespread use in the past and its low biodegradation rate, chlordane is commonly measured in low concentrations in environmental samples (Eisler, 1990). Chlordane, with an organic carbon-sediment partition coefficient (K_{oc}) of between 15,500 and 24,600, strongly adsorbs to sediment (Howard, 1991).

Plants. Information on the toxicity of chlordane in plants is very limited. Chlordane concentrations of greater than 100 µg/L were found to inhibit the growth of blue-green algae

(Glooschenko et al., 1979), whereas concentrations of greater than 1,000 µg/L inhibited the growth of green algae (Glooschenko and Lott, 1977). A 1971 nationwide survey of chlordane concentrations in corn (*Zea mays*) and sorghum (*Sorghum halepense*) indicated 0.48 mg/kg (dry weight) in corn kernels, 1.26 mg/kg (dry weight) in corn stalk, and 0.42 mg/kg (dry weight) in sorghum (Carey et al., 1978).

Terrestrial Invertebrates. No information was found regarding the potential toxicity of chlordane to terrestrial invertebrates.

Mammals. Absorption of chlordane can occur through the skin, diet, and inhalation (Eisler, 1990). Studies have shown the metabolites of technical grade chlordane to be present in bat milk (Clark et al., 1978). Oxychlordane and heptachlor epoxide are metabolites of the components of technical grade chlordane (Eisler, 1990). Oxychlordane is more toxic and persistent than any of the parent compounds.

Acute oral LD₅₀ values for exposure of sensitive mammalian species to technical grade chlordane range from 25 to 50 mg/kg body weight. Chlordane fed to rats at 2.5 mg/kg caused slight liver damage (NRC, 1977). Growth retardation and liver damage were observed in rats fed 150 and 300 mg/kg chlordane in their diets over a two-year period (Clayton and Clayton, 1982). No measurable effects were found in *Cynomolgus* monkeys (*Macaca spp.*) exposed to technical grade chlordane concentrations of 10 µg/L in air for 90 days (Khasawinah et al., 1989). Based on a study of mice fed technical grade chlordane in their diets, a NOAEL value of 4.6 mg/kg/day has been derived (Keplinger et al., 1968). The critical endpoint for this study was reproduction.

Symptoms of acute chlordane poisoning in mammals include diarrhea, avoidance of food and water, hair loss, hunched appearance, abdominal distension, labored respiration, salivation, muscle tremors, incoordination, convulsions, and death in some cases (Eisler, 1990). Chlordane has been reported to be carcinogenic in mice (Eisler, 1990) and teratogenic in rats (NRC, 1977). Chlordane potentiates aldrin, endrin, and parathion toxicity in mice (Jones et al., 1977).

Birds. Chlordane and related metabolites are commonly detected in wild birds throughout the United States. A 1982 survey revealed 45 percent of the captured European starlings (*Sturnus vulgaris*) contained detectable concentrations of oxychlordane (Eisler, 1990). Avian studies have shown older birds and raptors to contain relatively high concentrations of chlordane and its metabolites (Eisler, 1990). Lethal concentrations of oxychlordane and heptachlor have been measured in raptors that consumed either poisoned bait or prey (Blus et al., 1983). As in

mammals, chlordane concentrations are greatest in tissues high in lipid content (Eisler, 1990). Biological half-lives of chlordane isomers in birds range from 11.2 to 35.4 years (Eisler, 1990).

Among the birds tested, California quail (*Callipepla californica*) are the most sensitive to chlordane, with an acute oral LD₅₀ of 14.1 mg/kg body weight (Hudson et al., 1984). Toxic responses to technical chlordane are primarily attributed to the metabolite oxychlordane (Stickel et al., 1983). Oxychlordane concentrations in brain tissue greater than 5 mg/kg (wet weight) are considered lethal to birds (Stickel et al., 1979). Based on exposure of red-winged blackbirds to chlordane in their diets for 84 days, a chronic NOAEL value of 2.14 mg/kg/day has been derived (Stickel et al., 1983). The critical endpoint in this study was mortality. Signs of chlordane poisoning in birds include sluggishness, drooped eyelids, fluffed feathers, altered resting patterns, reduced appetite, weight loss, quivering and panting, neck arched over back, and arched back (Stickel et al., 1983).

Aquatic Life. Because of the widespread historical use of chlordane, it is commonly detected in aquatic environments. Vieth et al. (1979) reported 36 percent of all fish samples collected in domestic watersheds throughout the United States in 1976 contained chlordane. Of the components of technical grade chlordane detected in fish, cis-chlordane was the most abundant, followed by trans-nonachlor, trans-chlordane, and cis-nonachlor (Ribick and Zajicek, 1983). Bioconcentration factors for aquatic organisms range from 108 for frogs (*Xenopus laevis*) to 38,000 for fathead minnows (*Pimephales promelas*) (Howard, 1991). There is some evidence of biomagnification of chlordane in freshwater fish (Eisler, 1990).

Variations in the response of aquatic organisms to chlordane are related to both biological factors of the biota and physical factors of chlordane and the aquatic environment. Younger life-stages are more sensitive to chlordane toxicity than older life-stages (Eisler, 1990). In addition, the health and lipid content of the organism can have an impact on the response (Eisler, 1990). Water temperature, salinity, and sediment loading are abiotic factors that influence the bioavailability of chlordane (NRCC, 1975). Cis-chlordane appears to be more toxic than trans-chlordane, and the photoisomers seem to be more toxic than the parent compounds (Eisler, 1990).

The EPA's National Ambient Water Quality Criteria for chlordane is 2.4 µg/L for acute exposure and 0.0043 µg/L for chronic exposure of aquatic life (EPA, 2003b). The water quality criteria for chlordane represent the sum of alpha-chlordane, gamma-chlordane, alpha-chlordane, gamma-chlordane, alpha-nonachlor, gamma-nonachlor, and oxy-chlordane. The lowest chronic toxicity

values for chlordane reported in the literature for fish and daphnids are 1.6 and 16 µg/L, respectively (Suter and Tsao, 1996).

The test EC₂₀ for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration of chlordane causing less than 20 percent reduction in either the weight of young fish per initial female fish in a life-cycle or partial life-cycle test, or the weight of young per egg in an early life-stage test (Suter and Tsao, 1996). The fish EC₂₀ value for fish is less than 0.25 µg/L (Suter and Tsao, 1996). A similar value can be determined for daphnids, which reflects the highest tested concentration of chlordane causing less than 20 percent reduction in the growth, fecundity, and survivorship in a chronic test with a daphnid species. The EC₂₀ benchmark for daphnids is 12.1 µg/L (Suter and Tsao, 1996). Signs of chlordane poisoning in fish include hyperexcitability, increased respiration rate, erratic swimming, loss of equilibrium, convulsions, and death (NRCC, 1975).

7.4.2.17 Heptachlor

Heptachlor is an organochlorine cyclodiene insecticide, first isolated from technical grade chlordane in 1946. During the 1960s and 1970s, it was used primarily by farmers to kill termites, ants, and soil insects in seed grains and on crops, as well as exterminators and home owners to kill termites. Heptachlor acts as a nonsystemic stomach and contact insecticide. An important metabolite of heptachlor is heptachlor epoxide, which is an oxidation product formed from heptachlor by many plant and animal species (EXTOXNET, 2003).

Plants. No information was found regarding the toxicity of heptachlor to plants.

Terrestrial Invertebrates. No information was found regarding the potential toxicity of heptachlor to terrestrial invertebrates.

Mammals. Heptachlor is a highly to moderately toxic compound via the oral route of exposure, with reported oral LD₅₀ values of 100 to 220 mg/kg in rats, 30 to 68 mg/kg in mice, 116 mg/kg in guinea pigs, and 100 mg/kg in hamsters. It is moderately toxic via the dermal pathway as well, with reported dermal LD₅₀ values of 119 to 320 mg/kg in rats, and greater than 2,000 mg/kg in rabbits (EXTOXNET, 2003). Heptachlor, like many organochlorines may interfere with nerve transmission, and may also cause an increase in activity of the enzymes involved in the breakdown of foreign chemicals. Heptachlor is readily taken up through the skin, lungs, and gastrointestinal tract. Once absorbed, it is systemically distributed and preferentially moves into body fat. In mammals, heptachlor is readily converted to its most persistent and toxic

metabolite, heptachlor epoxide, in the liver. Heptachlor epoxide is stored mainly in fatty tissue, but also in liver, kidney, and muscle tissue. The acute toxicity of heptachlor epoxide, the main and most persistent of heptachlor's metabolites, may be greater. Effects due to heptachlor exposure may include hyperexcitation of the central nervous system, liver damage, lethargy, incoordination, tremors, convulsions, stomach cramps or pain, and coma. Results of animal tests show that chronic exposure to heptachlor or heptachlor epoxide adversely affects the liver, kidney, and red blood cells (EXTOXNET, 2003).

A study conducted with mink fed heptachlor in their diet for 181 days was used to derive a NOAEL value of 0.1 mg/kg/day (Crum et al., 1993). Fertility, litter size, and kit survival were reduced in the lowest exposure group (6.25 ppm).

Birds. Heptachlor is moderately to highly toxic to bird species; the reported acute oral LD₅₀ in mallard ducks is 2,080 mg/kg. The reported 5-day dietary LC₅₀ for Japanese quail is 99 ppm. Other reported 8-day dietary LC₅₀ values for heptachlor are 450 to 700 ppm in bobwhite quail, and 250 to 275 ppm in pheasants (EXTOXNET, 2003). It is also reported to decrease the survivability of chicken eggs. Heptachlor and its more toxic metabolite, heptachlor epoxide, have been found in the fat of fish and birds. They have also been found in the liver, brain, muscle, and eggs of birds (EXTOXNET, 2003).

Aquatic Life. Both heptachlor and heptachlor epoxide are highly toxic to most fish species tested. The reported 96-hour LC₅₀ values are: 5.3 to 13 µg/L in bluegill sunfish; 7.4 to 20 µg/L in rainbow trout; 6.2 µg/L in northern pike; 23 µg/L in fathead minnow; and 10 µg/L in largemouth bass. Heptachlor is also highly toxic to freshwater aquatic invertebrates. Both heptachlor and heptachlor epoxide have been shown to bioconcentrate in aquatic organisms such as fish, mollusks, insects, plankton, and algae. It has been found in several fish, mollusks, and other aquatic species at concentrations of 200 to 37,000 times the concentration of heptachlor in the surrounding water (EXTOXNET, 2003).

The Tier II secondary acute and secondary chronic values for heptachlor are 0.125 and 0.0069 µg/L, respectively (Suter and Tsao, 1996). The test EC₂₀ for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration of heptachlor causing less than 20 percent reduction in either the weight of young fish per initial female fish in a life-cycle or partial life-cycle test, or the weight of young per egg in an early life-stage test (Suter and Tsao, 1996). The lowest fish EC₂₀ for heptachlor is 0.86 µg/L. The National Recommended Water Quality Criteria (EPA, 2002d) for the protection of freshwater organisms

for heptachlor are 0.52 and 0.0038 µg/L for the criterion maximum concentration (CMC) and the criterion continuous concentration (CCC), respectively.

7.4.2.18 Methoxychlor

Methoxychlor is an organochlorine insecticide effective against a wide range of pests encountered in agriculture, households, and ornamental plantings. Methoxychlor is one of a few organochlorine pesticides that have seen an increase in use since the ban on DDT in 1972. It is quite similar in structure to DDT, but has relatively low toxicity and relatively short persistence in biological systems. It may be found in formulations with malathion, parathion, piperonyl butoxide, and pyrethrins (EXTOXNET, 2003).

Plants. No information was located in the literature regarding the toxicity of methoxychlor to plants.

Terrestrial Invertebrates. No information was found regarding the potential toxicity of methoxychlor to terrestrial invertebrates.

Mammals. Methoxychlor is practically non-toxic via the oral route, with reported oral LD₅₀ values of 5,000 to 6,000 mg/kg in rats, 1,850 mg/kg in mice, and 2,000 mg/kg in hamsters. It is reportedly practically non-toxic dermally, with a reported dermal LD₅₀ in rabbits of greater than 2,000 mg/kg. Symptoms of acute exposure include central nervous system depression, progressive weakness, and diarrhea. Data from animal studies indicate that target organs for methoxychlor include the kidneys, liver, mammary glands, and uterus (EXTOXNET, 2003).

Existing data suggest that methoxychlor is not carcinogenic, mutagenic, teratogenic, and is unlikely to cause reproductive effects in mammals. Rats exposed to methoxychlor in their diets for 11 months were observed for adverse effects on reproduction. Fertility and litter size were significantly reduced among rats fed diets containing 100 or 200 ppm methoxychlor (Gray et al., 1988). These data were used to derive a NOAEL value of 4 mg/kg/day.

Available evidence suggests that methoxychlor does not accumulate to any significant degree in fat or other tissues of mammals. At high dietary doses in rats, storage was minimal over the 18-week exposure period, and nondetectable within two weeks after the study (EXTOXNET, 2003).

Birds. Methoxychlor is slightly toxic to bird species, with reported acute LD₅₀ values of greater than 2,000 mg/kg for the mallard duck (*Anas platyrhynchos*), sharp-tailed grouse (*Tympanuchus*

phasianellus), and California quail (*Callipepla californica*). The reported 5-day dietary LC₅₀ for Japanese quail (*Coturnix japonica*) is greater than 5,000 ppm. Reported 8-day LC₅₀ values are greater than 5,000 ppm in bobwhite quail (*Colinus virginianus*) and ring-necked pheasants (*Phasianus colchicus*). Dietary levels of as high as about 145 mg/kg/day had no effects on reproductive function of male and female chickens over 8 to 16 weeks of exposure (EXTOXNET, 2003).

Aquatic Life. Methoxychlor is very toxic to fish and aquatic invertebrates. Reported 96-hour LC₅₀ values for technical grade methoxychlor (~ 90 % pure) are less than 20 µg/L for cutthroat trout, atlantic salmon, brook trout, lake trout, northern pike, and largemouth bass. Reported LC₅₀ values are between 20 and 65 µg/L for rainbow trout, goldfish, fathead minnow, channel catfish, bluegill, and yellow perch. Aquatic invertebrates with 96- or 48-hour LC₅₀ values of less than 0.1 mg/L include Daphnia, scuds, sideswimmers, and stoneflies (EXTOXNET, 2003).

Predicted bioconcentration factors (BCF) were the highest in the mussel (12,000) and the snail (8,570). These predicted BCFs indicate that methoxychlor can accumulate in aquatic organisms that do not rapidly metabolize the compound. Practically no metabolism was seen in Daphnia or mayflies. Fish reportedly break down methoxychlor fairly rapidly and thus tend not to accumulate it appreciably. No biomagnification of residues was observed in largemouth bass fingerlings fed contaminated Daphnia, and no evidence of metabolism was seen in rainbow trout (EXTOXNET, 2002). The Tier II secondary chronic value for methoxychlor is 0.019 µg/L (Suter and Tsao, 1996).

7.4.2.19 Nitrotoluene

If released to water, 4-nitrotoluene would be susceptible to photolysis, volatilization and aerobic biodegradation provided suitable acclimatization has taken place. Oxidation, chemical hydrolysis, adsorption to suspended solids and sediments, and bioaccumulation in aquatic organisms are not expected to be significant aquatic fate processes (Spectrum, 2004).

If released to soil, 4-nitrotoluene should be resistant to oxidation and chemical hydrolysis. This compound is reported to biodegrade under anaerobic conditions. 4-Nitrotoluene is expected to be moderately to highly mobile in soil and volatilize slowly from dry soil (Spectrum, 2004).

The bioconcentration factor for 4-nitrotoluene has been measured to be less than 100 in carp (*Cyprinus carpio*). Based on the water solubility and octanol-water partitioning coefficient, 4-nitrotoluene will not bioaccumulate significantly in aquatic organisms (Spectrum, 2004).

Plants. No information was located in the literature regarding the toxicity of nitrotoluene to plants.

Terrestrial Invertebrates. No information was located in the literature regarding the toxicity of nitrotoluene to plants.

Mammals. Toxicity studies in rats and mice have indicated that the most sensitive endpoint of nitrotoluene toxicity is methaemoglobinemia, leading to anaemia, Heinz body formation, reticulocytosis, and increased haematopoiesis with secondary effects on the spleen. These effects have been seen at exposure doses of 42 mg/kg/day in rats and 131 mg/kg/day in mice. Additional studies on reproduction and developmental effects in mice and rats indicated that adverse effects on these endpoints only occurred at higher doses than those described for the effects described above (CSTEE, 2003).

Birds. No information was located in the literature regarding the toxicity of nitrotoluene to birds.

Aquatic Life. The available data indicate low risks to aquatic organisms from exposure to nitrotoluene. Synthetic condensate wastewater prepared to simulate the composition of condensate wastewater from TNT manufacture and containing 32 compounds showed acute toxic concentrations for bluegills (*Lepomis macrochirus*), scud (*Hyaella azteca*), and green algae (*Selanastrum capricornutum*) of 7.1 mg/L, 22.8 mg/L, and 10 mg/L, respectively (CSTEE, 2003).

7.4.2.20 Phthalates

Phthalates are a class of predominantly man-made compounds which do not naturally occur in nature. They are manufactured and commonly used to produce flexible plastics, wetting agents, insecticidal sprays, paints, and glues (HSDB, 1996; ATSDR, 1993d). Because of their many uses, phthalates are widespread in the environment and have been identified at low levels in the air, water and soil. In air, phthalates may be adsorbed to particulate matter, and can be transferred to water by wet or dry deposition. In water and soil phthalates are subject to microbial degradation. Both aerobic and anaerobic degradation have been reported. Inman et al. (1984) demonstrated that di-n-butyl phthalate in soil was completely degraded within 100 days. Di-n-butyl phthalate and di-n-octyl phthalate have strong ultraviolet absorption bands at 274 nanometers extending beyond 290 nanometers and are therefore strong candidates for photolysis.

However, the estimated photolysis half-life in natural waters is 144 days for both compounds (Callahan et al., 1979). There is some evidence that phthalate esters might be biosynthesized and occur naturally in some plants and organisms (Callahan et al., 1979).

Plants. No information was found regarding the toxicity of phthalates to plants.

Terrestrial Invertebrates. No information was found regarding the potential toxicity of phthalates to terrestrial invertebrates.

Mammals. No studies were located on the effects of phthalate exposure to wildlife. Effects of phthalate esters in laboratory animals were seen at only very high doses (one to two percent di-n-butyl phthalate in the diet in oral studies). The male reproductive system appears to be the most sensitive target organ for acute-duration oral exposure to di-n-butyl phthalate in animals. A LOAEL of 1,000 mg/kg-day was established for decreased testis weight in rats (Oishi and Hiraga, 1980). The mechanism of testicular damage by di-n-butyl phthalate may involve interference with zinc metabolism (Foster et al., 1980). After oral administration, butyl benzyl phthalate was rapidly excreted. Rats and mice exposed to high concentrations of butyl benzyl phthalate lost weight, had testicular atrophy, hemorrhages, and hepatomegaly. LD₅₀ values for these experiments were 2.3 g/kg for rats and 4.2 to 6.2 g/kg for mice (DIALOG, 1996).

Mice fed bis(2-ethylhexyl)phthalate in their diets for 105 days were studied for effects on reproduction. While significant reproductive effects were observed among mice on diets containing 0.1 and 0.3 percent bis(2-ethylhexyl)phthalate, no adverse effects were observed among the 0.01 percent dose group. These data were used to derive a NOAEL value of 18.3 mg/kg/day (Lamb et al., 1987).

Birds. Ringed doves were fed bis(2-ethylhexyl)phthalate in their diets for 4 weeks during a critical lifestage and studied for reproductive effects (Peakall, 1974). No significant reproductive effects were observed in the maximally exposed doves (10 ppm). These data were used to derive a NOAEL value of 1.1 mg/kg/day based on reproductive effects.

Aquatic Life. Studies by Sasaki (1978) indicate that both di-n-butyl phthalate and di-n-octyl phthalate are non- or low-bioaccumulative in fishes. Studies by Streufert et al. (1981) showed the acute 48-hour LC₅₀s of di-2-ethylhexyl phthalate and di-n-butyl phthalate to the midge larvae *Chironomus plumosus* to be 18 mg/L and 0.76 mg/L, respectively. Chronic life cycle toxicity

tests showed no effect up to 0.36 mg/L di-2-ethylhexyl phthalate on midge emergence, egg production, or egg hatchability.

7.4.2.21 1,1,2,2-Tetrachloroethane

1,1,2,2-Tetrachloroethane is a man-made, colorless, dense liquid that does not burn easily. It is volatile and has a penetrating, sweet odor similar to chloroform. Most 1,1,2,2-tetrachloroethane released into the environment eventually moves into the air or groundwater. If released on the land, it does not tend to attach to soil particles. When released to surface water, much of the chemical will evaporate while the remainder may undergo degradation through hydrolysis. Similar reactions can take place in soils and sediments. Breakdown of 1,1,2,2-tetrachloroethane in both the air and groundwater is slow. In groundwater, the major degradation processes involve anaerobic biodegradation and chemical hydrolysis. Bioaccumulation of tetrachloroethane in fish or other aquatic organisms is expected to be minimal (ATSDR, 1996).

Plants. No information was found regarding the potential toxicity of 1,1,2,2-tetrachloroethane to plants.

Terrestrial Invertebrates. No information was found regarding the potential toxicity of 1,1,2,2-tetrachloroethane to terrestrial invertebrates.

Mammals. The liver appears to be the target organ most likely to be affected from low-level 1,1,2,2-tetrachloroethane exposure. However, reliable studies which demonstrate the hepatotoxicity of 1,1,2,2-tetrachloroethane are few in number. 1,1,2,2-Tetrachloroethane is converted by the liver to reactive intermediates, and hepatic hyperplasia and necrosis are most often noted in animal exposures. Common to all volatile chlorohydrocarbons, 1,1,2,2-tetrachloroethane causes pronounced depression of the central nervous system and respiratory depression is a frequent cause of death after acute exposures to high doses (ATSDR, 1996).

In rats treated with 300 mg/kg/day by gavage for 3-4 days, body weights were depressed by 16 percent. No depression in body weight was observed at an exposure level of 150 mg/kg/day. The available data show that oral administration of 1,1,2,2-tetrachloroethane to rats for 16 weeks caused no histopathological changes in the spleen or lymph nodes (ATSDR, 1996).

There is limited information available on the effects of 1,1,2,2-tetrachloroethane on reproduction. One report indicated necrosis of the testes, disturbed spermatogenesis, and clumping of the sperm in rats after 120 days of oral exposure to 1,1,2,2-tetrachloroethane.

However, in another study, no histological changes in the reproductive organs of rats or mice were observed after receiving high doses of the chemical for 78 weeks (ATSDR, 1996).

Birds. No information was found regarding the potential toxicity of 1,1,2,2-tetrachloroethane to birds.

Aquatic Life. The Tier II Secondary Acute and Secondary Chronic Values derived using the methodologies presented in the *Final Water Quality Guidance for the Great Lakes System* (EPA, 1995) are 2,100 and 610 µg/L, respectively. The lowest chronic toxicity values for fish, daphnids, and aquatic plants are reported as 2,400, 9,900, and 136,000 µg/L, respectively (Suter and Tsao, 1996). The test EC₂₀ for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration causing less than 20 percent reduction in either the weight of young fish per initial female fish in a life-cycle or partial life-cycle test, or the weight of young per egg in an early life-stage test (Suter and Tsao, 1996). The value for 1,1,2,2-tetrachloroethane has been determined to be 1,400 µg/L (Suter and Tsao, 1996). The test EC₂₀ for daphnids is < 420 µg/L (Suter and Tsao, 1996).

The EPA Region 5 ecological screening level (ESL) for 1,1,2,2-tetrachloroethane is 380 µg/L (EPA, 2003c).

7.4.2.22 Trichloroethene

Trichloroethene, or trichloroethylene, is not known to occur as a natural product (Howard, 1990). This chlorinated organic compound is primarily used as a degreaser of metals (Howard, 1990). Photooxidation is the primary mode by which trichloroethene is removed from the atmosphere (HSDB, 1996). The compound is fairly stable in soil; however, it can leach to groundwater. The primary removal process in aquatic systems is evaporation. Biodegradation is expected to be significant only under anaerobic conditions (HSDB, 1996).

Plants. Data on measured concentrations of trichloroethene in plants are limited to concentrations reported in grain-based foods, which ranged from 0.77 to 2.7 µg/kg (Heikes and Hopper, 1986). Phytotoxicity data for trichloroethene could not be found in the literature.

Terrestrial Invertebrates. No information was found regarding the potential toxicity of trichloroethene to terrestrial invertebrates.

Mammals. Concentrations of trichloroethene in wildlife are not reported in the literature. Trichloroethene is a hepatotoxin and central nervous system toxin (HSDB, 1996). The oral LD₅₀ value for mice exposed to trichloroethene is 2,402 g/kg (RTECS, 1996). The inhalation LC₅₀ value for mice exposed to trichloroethene is 8,450 ppm for a four-hour exposure period (RTECS, 1996). The LD₅₀ for dermal exposure of rabbits to trichloroethene is greater than 20 g/kg (RTECS, 1996). Fetotoxic and teratogenic effects have been reported in the offspring of pregnant rats exposed to trichloroethene (RTECS, 1996). Symptoms of chronic poisoning in dogs from inhalation of trichloroethene include lethargy, anorexia, nausea, vomiting, and weight loss (American Conference of Governmental Industrial Hygienists [ACGIH], 1971). There is a limited amount of evidence that suggests trichloroethene is carcinogenic in mammals (RTECS, 1996).

Mice exposed to trichloroethene through oral gavage for 6 weeks at dosage levels that ranged from 100 to 3,200 mg/kg/day were used to derive a chronic NOAEL value of 0.7 mg/kg/day (Buben and O'Flaherty, 1985). The critical endpoint in this study was hepatotoxicity.

Birds. No information was found regarding the potential toxicity of trichloroethene to birds.

Aquatic Life. Very little data exist on concentrations of trichloroethene in aquatic biota. Bioconcentration factors of between 17 and 39 have been reported for bluegill sunfish and rainbow trout exposed to trichloroethene (Burrows et al., 1989). LC₅₀ values for fathead minnows (*Pimephales promelas*) and bluegill sunfish (*Lepomis macrochirus*) exposed to trichloroethene for 96 hours have been reported as 20 and 44.7 mg/L (Borthwick, 1977).

The Tier II Secondary Acute and Secondary Chronic Values derived using the methodologies presented in the *Final Water Quality Guidance for the Great Lakes System* (EPA, 1995) are 440 and 47 µg/L, respectively. The lowest chronic toxicity values for fish and daphnids are reported as 11,100 and 7,257 µg/L, respectively (Suter and Tsao, 1996). The test EC₂₀ for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration causing less than 20 percent reduction in either the weight of young fish per initial female fish in a life-cycle or partial life-cycle test, or the weight of young per egg in an early life-stage test (Suter and Tsao, 1996). The value for trichloroethene has been determined to be 5,758 µg/L (Suter and Tsao, 1996).

7.4.2.23 1,3,5-Trinitrobenzene

1,3,5-Trinitrobenzene (TNB) is an explosive that is less sensitive to impact than TNT. TNB is formed as a byproduct during the manufacture of TNT. The photolysis of TNT also results in the

formation of TNB. TNB appears to be resistant to photolysis and hydrolysis. A low vapor pressure (3.2×10^{-6} mm Hg) indicates that TNB is not likely to partition to the air. TNB is only slightly soluble in water (0.34 g/L). Calculated bioconcentration factors for TNB (6.36) indicate little propensity to bioconcentrate.

Plants. *Selenastrum capricornutum* was tested for toxicity to TNB under static conditions. Mortality or significantly reduced growth, measured on day 5 and day 14 of the test, occurred at all concentrations tested (0.1 – 17.3 mg/L) (Bailey, 1982). Although a no-effect concentration was not attained in this study, it would be < 0.1 mg/L.

Terrestrial Invertebrates. No studies were found that reported the toxicity of TNB to terrestrial invertebrates.

Mammals. The U.S. Army conducted several toxicity studies that have demonstrated adverse health effects of TNB at high doses. Chronic studies have clearly demonstrated TNB-induced toxicity to the hematopoietic system. From these studies a NOAEL value of 2.64 mg/kg/day and a LOAEL value of 13.31 mg/kg/day have been derived (Reddy et al., 1997).

Three studies are available for calculating TNB screening benchmarks. A subchronic study with Fischer-344 rats assessed dosages of 0, 3.91, 22.73, and 44.16 mg/kg/day in their food. Moderate to severe seminiferous tubular degeneration was seen in male rats receiving 22.73 and 44.16 mg/kg/day doses. No effects were observed in the two lowest doses. The final chronic NOAEL based on this study was determined to be 0.4 mg/kg/day (Reddy et al., 1994). In a subchronic study with the white-footed mouse, Pathology Associates, Inc. (1994) assessed TNB dosages of 0, 23.5, 67.44, and 113.51 mg/kg/day. Seminiferous tubular degeneration was seen in 3 of 9 males receiving the 113.51 mg/kg/day dose. No adverse effects were seen at the two lowest doses. The final chronic NOAEL based on this study was determined to be 6.74 mg/kg/day.

In the chronic study (Reddy et al., 1996), male and female rats in the high-dose group showed decreased body weights associated with decreased food consumption, changes in relative organ weights, and adverse hematological findings; however, the life span was not affected and testicular effects, evident at the shorter time periods, were reversed except in the high-dose group. These findings, coupled with generally negative findings in reproductive and developmental toxicity studies, indicate that the highest dose tested, 13.44 mg/kg/day, is a

LOAEL for ecologically relevant endpoints and the mid-dose is a NOAEL. Because this was a chronic study, the final chronic NOAEL was determined to be 2.64 mg/kg/day.

Birds. No studies were found that reported the toxicity of TNB to birds.

Aquatic Life. Acute toxicity tests were conducted under static conditions with one invertebrate and four species of fish. The 48-hour EC₅₀ values obtained in two different laboratories for *Daphnia magna* were almost identical (i.e., 2.7 and 2.98 mg/L). Fish were more sensitive to TNB than daphnids, with 96-hour LC₅₀ values of 0.38 – 1.1 mg/L. Channel catfish appeared to be the most sensitive species.

Chronic tests were conducted under flow-through conditions with one invertebrate and two species of fish. Daphnids were tested at measured concentrations of 0 to 2.68 mg/L, fathead minnows were tested at measured concentrations of 0.08 to 0.72 mg/L, and rainbow trout were tested at measured concentrations of 0.08 to 0.71 mg/L. TNB was toxic to fish, with LOECs for several parameters < 1.0 mg/L (Talmage et al., 1999).

The Tier II Secondary Acute and Secondary Chronic Values for TNB, derived using the methodologies presented in the *Final Water Quality Guidance for the Great Lakes System* (EPA, 1995), are 60 and 11 µg/L, respectively. The lowest chronic toxicity values for fish, daphnids, and aquatic plants are reported as 120, 750, and 100 µg/L, respectively (Talmage et al., 1999). The test EC₂₀ for fish can be used as a benchmark indicative of production within a population. It is the highest tested concentration causing less than 20 percent reduction in either the weight of young fish per initial female fish in a life-cycle or partial life-cycle test, or the weight of young per egg in an early life-stage test (Suter and Tsao, 1996). The value for 1,1,2,2-tetrachloroethane has been determined to be 1,400 µg/L (Suter and Tsao, 1996). The test EC₂₀ for daphnids is < 420 µg/L (Suter and Tsao, 1996).

A sediment screening benchmark has been derived by Talmage et al. (1999) using the equilibrium partitioning approach suggested by DiToro et al. (1991) and EPA (1993). Assuming an organic carbon content of one percent, the sediment screening benchmark was estimated to be 0.002 mg/kg.

7.4.2.24 Vinyl Chloride

Vinyl chloride is a low molecular weight organic compound that rapidly volatilizes after it is released to soil and surface water. Aquatic organisms may take up vinyl chloride, however, it is

rapidly depurated because it is highly water-soluble. Routes of exposure for wildlife include inhalation, ingestion, and dermal exposure. Bioaccumulation in terrestrial and aquatic organisms is not an important process in the environmental fate of vinyl chloride because of its high volatility and the rapid metabolism by higher-trophic-level receptors.

Plants. No information was found regarding the potential toxicity of vinyl chloride to plants.

Terrestrial Invertebrates. No information was found regarding the potential toxicity of vinyl chloride to terrestrial invertebrates.

Mammals. In mammals, vinyl chloride may be absorbed by the body via inhalation, ingestion, and dermal contact. It is rapidly absorbed and distributed throughout the tissues following uptake. Because of the rapid metabolism and excretion of vinyl chloride, storage within the body is minimal. Rats exposed to vinyl chloride by inhalation were found to have peak blood levels after 30 minutes of exposure to 7,000 ppm, which decreased rapidly after removal (Withey, 1976). Vinyl chloride has been found to be rapidly and almost completely absorbed from the gastrointestinal tract after oral exposure (Withey, 1976). Based upon a single study with monkeys, dermal absorption of vinyl chloride vapor is not likely to be significant (Hefner, 1975).

Renal effects, including increased relative kidney weight and an increase in severity of tubular nephrosis, have been reported in several rat studies (Feron et al., 1979). Chronic exposure of rats to vinyl chloride in their feed for 149 weeks caused an increase in the incidence of basophilic foci of cellular alteration (Til et al., 1991). Chronic oral exposure of rats fed vinyl chloride daily for 2 years also resulted in areas of hepatocellular alteration at concentrations as low as 1.8 mg/kg/day. In this study, areas of necrosis were observed in the liver of female rats fed 5.6 mg/kg/day and male rats fed 17 mg/kg/day (Feron et al., 1981). Increased incidence of hepatic cysts were found in female rats fed 1.7 mg/kg/day and clear or basophilic areas of cellular alteration were found in male rats fed 1.7 mg/kg/day.

Feron et al. (1981) studied the effects of vinyl chloride on longevity and mortality in rats. Vinyl chloride was administered orally in the diet of rats at three dose levels (1.7, 5.0, and 14.1 mg/kg/day). Significantly reduced survivorship was observed at all dose levels; therefore, the 1.7 mg/kg/day dose level was considered to be the chronic LOAEL. The NOAEL was estimated by multiplying the LOAEL by an uncertainty factor of 0.1 to derive a NOAEL of 0.17 mg/kg/day.

Birds. No information was found regarding the potential toxicity of vinyl chloride to birds.

Aquatic Life. Vinyl chloride is not expected to significantly bioconcentrate in aquatic organisms because it has a very low log K_{ow} value. Bioconcentration and accumulation in aquatic carnivores is not expected because of the high volatility of vinyl chloride and the rapid metabolism of vinyl chloride by higher trophic-level organisms.

The EPA Region 5 ESL for vinyl chloride in freshwater is 0.93 mg/L.

7.4.3 Potential Receptors

Potential ecological receptors in the vicinity of Range K fall into two general categories: terrestrial and semi-aquatic. Within these two general categories there are several major feeding guilds that could be expected to occur in the vicinity of Range K: herbivores, invertivores, omnivores, and carnivores. All of these feeding guilds have the potential to be directly exposed to soil-related constituents at Range K via various activities (e.g., feeding, grooming, bathing). Some of these feeding guilds may also be exposed to site-related chemicals in groundwater if the potentiometric surface is elevated sufficiently such that groundwater discharges to the ephemeral drainage feature south of the Range K study area. These feeding guilds may also be exposed to site-related chemicals via food web transfers.

Dermal absorption of constituents from soil is a potential pathway for all feeding guilds at Range K; however, birds and mammals are less susceptible to dermal exposures because their feathers or fur prevents skin from coming into direct contact with the soil (EPA, 1993). Dermal absorption of inorganic compounds from direct contact with soil is expected to be minimal due to the low dermal permeability of the constituents detected in surface soil at Range K. Since VOCs were only detected in very low concentrations in surface soil at Range K, inhalation of VOCs is not a significant exposure pathway. Inhalation of constituents sorbed to soil particles and inhaled as dust is a potential pathway for all of the feeding guilds at Range K; however, the fact that the entire study area is forested precludes the generation of significant amounts of fugitive dust. Therefore, inhalation of particulates is expected to be minimal at Range K.

Although terrestrial species could be exposed to constituents in groundwater if discharged to the drainage feature located south of the Range K study area, this exposure pathway would only be complete during periods of significant precipitation, when groundwater elevations are higher than the creek bed elevation. These potential exposures are expected to be relatively

insignificant due to the fact that they would most likely occur only for short durations during a normal year and the concentrations of constituents in groundwater would be greatly reduced due to the increased flow and dilution experienced during these high precipitation periods.

Semi-aquatic species (i.e., amphibians) have a greater potential for exposure to constituents in groundwater as they spend a majority of their lifetime in close proximity to water bodies. Semi-aquatic species could be exposed to constituents in groundwater via ingestion and dermal contact. Although these species are expected to be in close contact with groundwater when it is expressed as surface water during periods of high precipitation, there are significant periods of the year when the ephemeral drainage feature at Range K is dry and exposure to groundwater does not occur.

Aquatic species (e.g., fish) and the feeding guilds (i.e., piscivores) that rely on aquatic species for food are not expected to occupy the area around Range K. The small ephemeral drainage feature in the vicinity of Range K does not provide suitable habitat to support aquatic species, and by association, the species that rely on them for food.

7.4.3.1 Herbivorous Feeding Guild

The major route of exposure for herbivores is through ingestion of plants that may have accumulated contaminants from the soil. Since terrestrial herbivores by definition are grazers and browsers, they could be exposed to chemicals that have accumulated in the vegetative tissues of the plants at the site. Terrestrial herbivores may also be exposed to site-related chemicals in soil through incidental ingestion of soil while grazing, grooming, or other activities. Herbivores could be exposed to groundwater through ingestion of water in the ephemeral drainage feature during periods of high precipitation when the potentiometric surface of the groundwater exceeds the creek bed elevation. However, exposure to groundwater in this manner is only expected to be sporadic.

Typical herbivorous species that could be expected to occur at Range K and are commonly used as sentinel species in ecological risk assessment include eastern cottontail (*Sylvilagus floridanus*), eastern gray squirrel (*Sciurus carolinensis*), pine vole (*Pitymys pinetorum*), whitetail deer (*Odocoileus virginianus*), and wild turkey (*Meleagris gallopavo*).

Aquatic herbivores, such as muskrat (*Ondatra zibethica*) and mallard (*Anas platyrhynchos*) are not expected to occur at Range K due to the general lack of aquatic habitat in the area of Range K.

7.4.3.2 Invertivorous Feeding Guild

Invertivores specialize in eating insects and other invertebrates. As such, they may be exposed to site-related chemicals that have accumulated in insects and other invertebrates. Invertivores may also be exposed to site-related chemicals in soil through incidental ingestion of soil while probing for insects, grooming, or other activities. Ingestion of soil while feeding is a potential exposure pathway for invertivores since much of their food (i.e., earthworms and other invertebrates) lives on or below the soil surface. Invertivores could be exposed to groundwater through ingestion of water in the ephemeral drainage feature during periods of high precipitation when the potentiometric surface of the groundwater exceeds the creek bed elevation. However, exposure to groundwater in this manner is expected to be sporadic.

Typical invertivorous species that could be expected to occur at Range K and are commonly used as sentinel species in ecological risk assessment include American woodcock (*Philohela minor*), carolina wren (*Thryothorus ludovicianus*), shorttail shrew (*Blarina brevicauda* or *Blarina carolinensis*), and eastern mole (*Scalopus aquaticus*). Aquatic invertivores such as the wood duck (*Aix sponsa*) and blacknose dace (*Rhinichthys atratulus*) are not expected to occur at Range K due to the general lack of aquatic habitat in the area of Range K.

7.4.3.3 Omnivorous Feeding Guild

Omnivores consume both plant and animal material in their diet, depending upon availability. Therefore, they could be exposed to chemicals that have accumulated in the vegetative tissues of plants at the site and also chemicals that may have accumulated in smaller animal tissues that the omnivores prey upon. Omnivores may also be exposed to site-related chemicals in soil through incidental ingestion of soil while feeding, grooming, or other activities. Omnivores could be exposed to groundwater through ingestion of water in the ephemeral drainage feature during periods of high precipitation when the potentiometric surface of the groundwater exceeds the creek bed elevation. However, exposure to groundwater in this manner is expected to be sporadic.

Typical omnivorous species expected to occur at Range K and are commonly used as sentinel species in ecological risk assessment include red fox (*Vulpes vulpes*), white-footed mouse (*Peromyscus leucopus*), and American robin (*Turdus migratorius*). Aquatic omnivores, such as raccoon (*Procyon lotor*) and creek chub (*Semotilus atromaculatus*) are not expected to occur at Range K because of the general lack of aquatic habitat in the area.

7.4.3.4 Carnivorous Feeding Guild

Carnivores are meat-eating animals and are therefore potentially exposed to site-related chemicals through consumption of prey animals that may have accumulated contaminants in their tissues. Carnivores are quite often top predators in a local food web and are often subject to exposure to contaminants that have bioaccumulated in lower trophic-level organisms or biomagnified through the food web. Food web exposures for carnivores are based on the consumption of prey animals that have accumulated COPECs from various means. Smaller, herbivores, omnivores, invertivores, and other carnivores may consume soil, plant, and animal material as food and accumulate COPECs in their tissues. Subsequent ingestion of these prey animals by carnivorous animals would expose them to COPECs. These soil-related exposure pathways are expected to be minimal at Range K because none of the constituents detected in surface soil at elevated concentrations are accumulated in animal tissues to any great extent (Shugart, 1991 and U.S. Army Environmental Hygiene Agency [USAEHA], 1994). Therefore, food web exposures to these chemicals are expected to be minimal.

Carnivores may also be exposed to site-related chemicals in soil through incidental ingestion of soil while feeding, grooming, or other activities. Carnivores could be exposed to groundwater through ingestion of water in the ephemeral drainage feature during periods of high precipitation when the potentiometric surface of the groundwater exceeds the creek bed elevation. However, exposure to groundwater in this manner is only expected to be sporadic.

Typical carnivorous species expected to occur at Range K and are commonly used as sentinel species in ecological risk assessment include red-tailed hawk (*Buteo jamaicensis*), black vulture (*Coragyps atratus*), and bobcat (*Lynx rufus*).

7.4.4 Complete Exposure Pathways

For exposure to occur, a complete exposure pathway must exist between the contaminant and the receptor. A complete exposure pathway requires the following four components:

- A source mechanism for contaminant release
- A transport mechanism
- A point of environmental contact
- A route of uptake at the exposure point (EPA, 1989).

If any of these four components is absent, then a pathway is generally considered incomplete. Potentially complete and incomplete exposure pathways at Range K are depicted in the SCM shown on Figure 7-1.

Ecological receptors may be exposed to constituents in soils via direct and/or secondary exposure pathways. Direct exposure pathways include soil ingestion, dermal absorption, and inhalation of volatile constituents and constituents adsorbed to fugitive dust. Significant exposure via dermal contact is limited to organic constituents that are lipophilic and can penetrate epidermal barriers. Mammals are less susceptible to exposure via dermal contact with soils because their fur prevents skin from coming into direct contact with soil. However, soil ingestion may occur while grooming, preening, burrowing, or consuming plants, insects, or invertebrates resident in soil.

Exposures to site-related constituents in surface water and sediment are not likely because there are no permanent surface water bodies associated with Range K. However, exposure to constituents in groundwater may occur sporadically when groundwater may discharge to the ephemeral drainage feature located south of the Range K study area.

Exposure via inhalation of fugitive dust is limited to contaminants present in surface soils at areas that are devoid of vegetation. The amount of vegetative cover, the inherent moisture content of the soil, and the frequency of soil disturbance play important roles in the amount of fugitive dust generated at a particular site. Because Range K is entirely forested, the amount of fugitive dust generated at Range K is expected to be minimal.

Secondary exposure pathways involve constituents that are transferred through different trophic levels of the food chain and may be bioaccumulated and/or bioconcentrated. This may include constituents bioaccumulated from soil into plant tissues or into terrestrial species ingesting soils. These plants or animals may, in turn, be consumed by animals at higher trophic levels.

In general, the constituents detected in surface soil at Range K may bioaccumulate in lower trophic level organisms (i.e., terrestrial invertebrates may bioaccumulate inorganic compounds detected in soil); however, they will not bioconcentrate through the food chain. None of the constituents detected in surface soil at Range K at elevated concentrations are expected to bioconcentrate to any great extent in either plant or animal tissues.

A summary of the feeding guilds and potentially complete exposure pathways at Range K is presented in Table 7-1.

7.5 Screening-Level Risk Estimation

A screening-level estimation of potential risk can be accomplished by comparing the exposure point concentration of each detected constituent in each environmental medium to a corresponding screening-level ecological toxicity value. In order to conduct the SLERA, the following steps must be followed:

- Determine appropriate screening assessment endpoints
- Determine the ecological toxicity values that are protective of the selected assessment endpoints
- Determine the exposure point concentrations of constituents detected at the site
- Calculate screening-level hazard quotients.

These steps are summarized below.

7.5.1 Ecological Screening Assessment Endpoints

Most ecological risk assessments focus on population measures as endpoints, since population responses are better defined and more predictable than are community or ecosystem responses. For screening-level assessments such as this SLERA, assessment endpoints are any adverse effects on ecological receptors, where receptors are plant and animal populations and communities, habitats, and sensitive environments.

Adverse effects on populations can be inferred from measures related to impaired reproduction, growth, and survival. Adverse effects on communities can be inferred from changes in community structure or function. Adverse effects on habitats can be inferred from changes in composition and characteristics that reduce the ability of the habitat to support plant and animal populations and communities.

Due to the nature of the SLERA process, most of the screening assessment endpoints are generic in nature (i.e., protection of sediment benthic communities from adverse changes in structure or function).

The assessment endpoints for this SLERA were identified for surface soil and surface water and are summarized below. It is important to note that although surface water was not sampled at Range K, the potential for groundwater seepage into the nearby ephemeral drainage feature

necessitated that an assessment of groundwater be made by comparing groundwater data to surface water ESVs.

- **Soil**
 - Protection of the terrestrial invertebrate community from adverse changes in structure and function
 - Protection of the terrestrial plant community from adverse changes in structure and function.
- **Surface Water**
 - Protection of the aquatic community from adverse changes in structure and function

7.5.2 Ecological Screening Values

The ecological screening values (ESV) used in this assessment represent the most conservative values available from various literature sources and have been selected to be protective of the assessment endpoints described above. These ESVs have been developed specifically for FTMC in conjunction with EPA Region 4 and are presented in the *Final Human Health and Ecological Screening Values and PAH Background Summary Report* (IT, 2000b). The ESVs used in this assessment are based on NOAELs when available. If a NOAEL-based ESV was not available for a certain COPEC, then the most health-protective value available from the scientific literature was used in this assessment.

For each environmental medium sampled at Range K (surface soil and groundwater), a hierarchy has been developed which presents an orderly method for selection of ESVs. The hierarchy for selecting ESVs for surface soil is as follows:

- EPA Region 4 constituent-specific ESVs
- EPA Region 4 ESVs for general class of constituents
- EPA Region 5 ecological data quality levels (EDQL)
- EPA Region 3 Biological Technical Advisory Group (BTAG) values
- ESVs from Talmage et al., 1999.

The hierarchy for selecting ESVs for surface water, which were used to assess groundwater data at Range K, is as follows:

- EPA Region 4 constituent-specific ESVs;

- NOAA Screening Quick Reference Tables (SQRT), chronic freshwater ambient water quality criteria;
- EPA Region 5 EDQLs;
- Office of Solid Waste and Emergency Response (OSWER) Ecotox Threshold values;
- EPA Region 3 BTAG values;
- Lowest chronic value from Suter and Tsao, 1996; and
- ESVs from Talmage et al., 1999.

7.5.3 Determination of Exposure Point Concentrations

Exposure point concentrations represent the chemical concentrations in environmental media that a receptor may contact. Since the exposure point concentration is a value that represents the most likely concentration to which receptors could be exposed, a value that reflects the central tendency of the data set is most appropriate to use for free-ranging animals that would be expected to use the site indiscriminately. Smaller, more sessile organisms with smaller home ranges may be exposed to only a portion of a site. Additionally, habitat preferences or preferential avoidance behavior may result in exposure to only portions of a given site. Therefore, a subset of the data would be most appropriate to estimate an exposure point concentration for these species. The most conservative approach is the use of the maximum detected constituent concentrations as exposure point concentrations. At the screening-level stage, the data sets are generally not robust enough for statistical analysis and the level of conservatism in the exposure estimates is high to account for uncertainties. Therefore, in the screening-level stage, the maximum detected constituent concentration in each environmental medium is used as the exposure point concentration. The use of the maximum detected constituent concentration as the exposure point concentration ensures that the exposures will not be underestimated, and therefore, constituents will not be inadvertently eliminated from further assessment.

The statistical summaries (including the exposure point concentrations) for surface soil and groundwater at Range K are presented in Tables 7-2 and 7-3.

7.5.4 Screening-Level Hazard Quotients

In order to estimate whether constituents detected in environmental media at the site have the potential to pose adverse ecological risks, screening-level hazard quotients were developed. The screening-level hazard quotients were developed via a three-step process as follows:

- Comparison to ESVs
- Identification of essential macronutrients
- Comparison to naturally occurring background concentrations.

Constituents that were detected in environmental media at Range K were evaluated against the ESVs by calculating a screening-level hazard quotient (HQ_{screen}) for each constituent in each environmental medium. An HQ_{screen} was calculated by dividing the maximum detected constituent concentration in each environmental medium by its corresponding ESV as follows:

$$HQ_{screen} = \frac{MDCC}{ESV}$$

where:

HQ_{screen} = screening-level hazard quotient;
 $MDCC$ = maximum detected constituent concentration; and
 ESV = ecological screening value.

A calculated HQ_{screen} value of one indicated that the MDCC was equal to the chemical's conservative ESV and was interpreted in this assessment as a constituent that does not pose the potential for adverse ecological risk. An HQ_{screen} value less than one indicated that the MDCC was less than the conservative ESV and that the chemical is not likely to pose adverse ecological hazards to most receptors. Conversely, an HQ_{screen} value greater than one indicated that the MDCC was greater than the ESV and that the chemical might pose adverse ecological hazards to one or more receptors and requires further assessment.

In order to better understand the potential risks posed by chemical constituents at Range K, a mean hazard quotient was also calculated by comparing the arithmetic mean constituent concentration in surface soil and groundwater to the corresponding ESV. The calculated screening-level hazard quotients for surface soil and groundwater at Range K are presented in Tables 7-2 and 7-3.

EPA recognizes several constituents in abiotic media that are necessary to maintain normal function in many organisms. These essential macronutrients are iron, magnesium, calcium, potassium, and sodium (EPA, 1989). Most organisms have mechanisms designed to regulate nutrient fluxes within their systems; therefore, these nutrients are generally only toxic at very high concentrations. Although iron is an essential nutrient and is regulated within many organisms, it may become increasingly bioavailable at lower soil pH values, thus increasing its potential to elicit adverse effects. Therefore, iron was not evaluated as an essential nutrient in this SLERA. Essential macronutrients were only considered COPECs if they were present in site samples at concentrations ten times the naturally occurring background concentration.

The comparison of detected constituent concentrations with naturally occurring constituent concentrations was conducted via a three-tier process outlined in a Shaw technical memorandum (Shaw, 2003c). The first tier of the background comparison process was a comparison of the maximum detected constituent concentration to the BTV. A study of the natural geochemical composition associated with FTMC (SAIC, 1998) determined the mean concentrations of 24 metals in surface soil, surface water, sediment, and groundwater samples collected from presumably un-impacted areas. Per agreement with EPA Region 4, the BTV for each metal was calculated as two times the mean background concentration for that metal. The BTV for each metal was used to represent the upper boundary of the range of natural background concentrations expected at FTMC, and was used as the basis for evaluating metal concentrations measured in site samples. Site sample metal concentrations less than or equal to the corresponding BTV represent the natural geochemical composition of media at FTMC, and not contamination associated with site activity. Site sample metal concentrations greater than the corresponding BTV require further background assessment.

If maximum constituent concentrations were greater than the BTV, then the second tier of the background comparison was employed. Tier two of the background comparison consists of statistical comparisons of the site data to background data using the Slippage Test and the Wilcoxon Rank Sum (WRS) Test. If the site data failed either the Slippage Test or the WRS Test, then the site data were subjected to a geochemical evaluation to determine whether concentrations of inorganic compounds are naturally occurring or are elevated due to contamination (Tier 3). The three-tier background comparison process is described in detail in Appendix I of this report.

Thus, the first step in determining screening-level hazard quotients was a comparison of maximum detected constituent concentrations to appropriate ESVs. Constituents with HQ_{screen} values less than one were considered to pose insignificant ecological risk and were eliminated from further consideration. Constituents with HQ_{screen} values greater than one were eliminated from further consideration if they were macronutrients and their detected concentrations were less than ten-times naturally occurring levels. Those constituents that had HQ_{screen} values greater one and were not considered macronutrients were then compared to background using the three-tier background screening process. If constituent concentrations were determined to be less than their naturally occurring background concentrations, then a risk management decision could result in eliminating these constituents from further assessment.

7.6 Identification of Constituents of Potential Ecological Concern

A constituent was identified as a COPEC if the following conditions were met:

- The maximum detected constituent concentration exceeded the ESV;
- The maximum detected constituent concentration was 10 times the BTV if the constituent was identified as a macronutrient; and
- Constituent concentrations were determined to be greater than naturally occurring levels based on the three-tier background screening process.

If a constituent in a given environmental medium did not meet these conditions, then it was not identified as a COPEC at Range K and was not considered for further assessment. If a constituent met these conditions, then it was identified as a COPEC. Identification of a constituent as a COPEC indicates that further assessment of that particular constituent in a given environmental medium may be appropriate. It does not imply that a particular constituent poses risk to ecological receptors.

The COPECs that have been initially identified for surface soil at Range K are presented in Table 7-2 and the COPECs that have initially been identified in groundwater are presented in Table 7-3. Surface water and sediment were not sampled at Range K due to the ephemeral nature of the drainage feature associated with this site.

In order to focus future ecological assessment efforts (if necessary) on the constituents that are the most prevalent at Range K and have the greatest potential to pose ecological risk, additional lines of evidence were assessed. Additional lines of evidence are sometimes useful in determining whether a certain constituent is in fact site-related and a COPEC. Some of the

additional lines of evidence used in the process of identifying COPECs include: 1) frequency of detection, 2) magnitude of the HQ_{screen} value, 3) spatial distribution, 4) alternative ESVs; and 5) association of a chemical with known Army activities. These additional lines-of-evidence were used to further define the COPECs at Range K and are discussed below.

7.6.1 Surface Soil

A number of inorganic constituents in surface soil exceeded their respective ESVs, BTVs, and are not essential macronutrients. These constituents are discussed further below.

Aluminum. Aluminum was detected in one surface soil sample (RNG-203-MW02) out of 13 samples collected at a concentration greater than its BTV. Aluminum passed the Slippage Test but failed the WRS Test; therefore, a geochemical evaluation was conducted. All of the detected aluminum in surface soil from Range K was determined to be naturally occurring through geochemical evaluation. Per EPA (2000) guidance, aluminum toxicity is associated with soluble aluminum only. Numeric screening values for aluminum are considered inappropriate. Alternatively, potential ecological risks associated with exposure to aluminum are associated with soil pH. Aluminum is identified as a COPEC only if the soil pH is less than 5.5 (EPA, 2000). Since the pH of soils at Range K is greater than 5.5, aluminum was considered to be non-toxic to ecological receptors. Based on the relative infrequency of elevated aluminum detections, the magnitude of the detected concentrations relative to background, and the fact that aluminum is only toxic at soil pH values less than 5.5, aluminum was not considered a COPEC in surface soil at Range K.

Arsenic. Arsenic was detected in two surface soil samples (RNG-203-MW04 and RNG-203-MW05) out of 13 samples collected at concentrations that exceeded the BTV. The calculated HQ_{screen} value for arsenic was 2.6. Arsenic passed the Slippage Test but failed the WRS Test; therefore, a geochemical evaluation was conducted. All of the detected arsenic in surface soil from Range K was determined to be naturally occurring through geochemical evaluation. Since elevated arsenic was detected only sporadically, the HQ_{screen} value was relatively low, and geochemical evaluation indicated that arsenic was naturally occurring, arsenic was not considered a COPEC in surface soil at Range K.

Barium. Barium was detected in one surface soil sample (RNG-203-MW08) out of 13 samples collected at a concentration that exceeded the ESV and BTV. The calculated HQ_{screen} value for barium was 2.1. Both the Slippage Test and the WRS Test concluded that the detected barium concentrations in surface soil at Range K were consistent with naturally occurring barium at

FTMC. Based on the fact that barium was infrequently detected at elevated concentrations, the low magnitude of the HQ_{screen} value, and the fact that statistical analysis indicated that barium was naturally occurring, barium was not considered a COPEC in surface soil at Range K.

Beryllium. Beryllium was detected in two surface soil samples (RNG-203-GP02 and RNG-203-MW08) out of 11 samples (excluding “B”-flagged data) at concentrations that exceeded the ESV. The calculated HQ_{screen} value for beryllium was 3.3. Beryllium passed the Slippage Test but failed the WRS Test; therefore, a geochemical evaluation was conducted. Geochemical evaluation determined that sample RNG-203-GP02 may contain a component of beryllium contamination, but all other beryllium concentrations in surface soil were naturally occurring. Efroymsen et al. (1997) have proposed a soil screening level for beryllium of 10 mg/kg based on phytotoxicity. None of the detected concentrations of beryllium were greater than this alternative ESV. Based on the fact that elevated concentrations of beryllium were infrequently detected, the calculated HQ_{screen} value was relatively low, and none of the detected concentrations were greater than the alternative ESV, beryllium was not considered a COPEC in surface soil at Range K.

Cadmium. Cadmium was detected in one surface soil sample (RNG-203-GP02) out of 13 samples at a concentration that exceeded the ESV. The calculated HQ_{screen} value for cadmium was 1.4. Statistical evaluation of the data for cadmium could not be conducted because cadmium was only detected in one surface soil sample at Range K. Geochemical evaluation determined that sample RNG-203-GP02 may contain a component of cadmium contamination, but all other cadmium concentrations in surface soil were naturally occurring. The EPA (2000) has proposed ecological soil screening levels for cadmium of 29 mg/kg (based on phytotoxicity) and 110 mg/kg (based on toxicity to soil invertebrates). None of the detected concentrations of cadmium in surface soil exceed these alternative ESVs. Based on the fact that cadmium was only detected in one surface soil sample, the calculated HQ_{screen} value was relatively low, and none of the detected concentrations of cadmium exceeded the alternative ESVs, cadmium was not considered a COPEC in surface soil at Range K.

Chromium. Chromium was detected in one surface soil sample (RNG-203-MW05) out of 13 samples at a concentration greater than the BTV. Chromium passed the Slippage Test but failed the WRS Test; therefore, a geochemical evaluation was conducted. All of the detected chromium in surface soil from Range K was determined to be naturally occurring through geochemical evaluation. Based on the infrequency of detection at elevated concentrations and

the fact that geochemical evaluation determined that the chromium in surface soil was naturally occurring, chromium was not identified as a COPEC in surface soil at Range K.

Cobalt. Cobalt was detected in two surface soil samples (RNG-203-GP02 and RNG-203-MW08) out of 11 samples at concentrations that exceeded the ESV. The calculated HQ_{screen} value for cobalt was 1.98. Both the Slippage Test and the WRS Test concluded that the detected cobalt concentrations in surface soil at Range K were consistent with naturally occurring cobalt at FTMC. Based on the fact that cobalt was infrequently detected at elevated concentrations, the low magnitude of the HQ_{screen} value, and the fact that statistical analysis indicated that cobalt was naturally occurring, cobalt was not considered a COPEC in surface soil at Range K.

Iron. Iron was detected in two surface soil samples (RNG-203-MW02 and RNG-203-MW08) out of 13 samples at concentrations that exceeded the BTV. Iron passed the Slippage Test but failed the WRS Test; therefore, a geochemical evaluation was conducted. All of the detected iron in surface soil from Range K was determined to be naturally occurring through geochemical evaluation. Based on the infrequency of detection at elevated concentrations and the fact that geochemical evaluation determined that the iron in surface soil was naturally occurring, iron was not identified as a COPEC in surface soil at Range K.

Nickel. Nickel was detected in two surface soil samples (RNG-203-GP02 and RNG-203-MW08) out of 13 samples at concentrations that exceeded the ESV. The calculated HQ_{screen} value for nickel was 2.3. Nickel failed both the Slippage Test and the WRS Test; therefore, a geochemical evaluation was conducted. Geochemical evaluation determined that sample RNG-203-GP02 may contain a component of nickel contamination, but all other nickel concentrations in surface soil were naturally occurring. Efroymson et al. (1997) have proposed a soil screening level for nickel of 200 mg/kg based on toxicity to earthworms. None of the detected concentrations of nickel were greater than this alternative ESV. Based on the fact that elevated concentrations of nickel were infrequently detected, the calculated HQ_{screen} value was relatively low, and none of the detected concentrations were greater than the alternative ESV, nickel was not considered a COPEC in surface soil at Range K.

Selenium. Selenium was detected in 6 out of 13 surface soil samples at concentrations greater than the ESV. The calculated HQ_{screen} value for selenium was 2.7. Selenium failed both the Slippage Test and the WRS Test; therefore, a geochemical evaluation was conducted. Geochemical evaluation determined that sample RNG-203-GP02 may contain a component of selenium contamination, but all other selenium concentrations in surface soil were naturally

occurring. Efroymsen et al. (1997) have proposed a soil screening level for selenium of 70 mg/kg based on toxicity to earthworms. None of the detected concentrations of selenium were greater than this alternative ESV. Based on the fact that the calculated HQ_{screen} value was relatively low and none of the detected concentrations were greater than the alternative ESV, selenium was not considered a COPEC in surface soil at Range K.

Vanadium. Vanadium was detected in two surface soil samples (RNG-203-MW05 and RNG-203-MW01) out of 13 samples at concentrations that exceeded the BTV. Vanadium passed the Slippage Test but failed the WRS Test; therefore, a geochemical evaluation was conducted. All of the detected vanadium in surface soil from Range K was determined to be naturally occurring through geochemical evaluation. Based on the infrequency of detection at elevated concentrations and the fact that geochemical evaluation determined that the vanadium in surface soil was naturally occurring, vanadium was not identified as a COPEC in surface soil at Range K.

Zinc. Zinc was detected in three surface soil samples (RNG-203-GP02, RNG-203-MW01, and RNG-203-MW08) out of 13 samples at concentrations that exceeded the ESV. The calculated HQ_{screen} value for zinc was 5.5. Zinc passed the Slippage Test but failed the WRS Test; therefore, a geochemical evaluation was conducted. Geochemical evaluation determined that sample RNG-203-GP02 may contain a component of zinc contamination, but all other zinc concentrations in surface soil were naturally occurring. Based on the relative infrequency of detection at elevated concentrations and the relatively low magnitude of the HQ_{screen} value, zinc was not identified as a COPEC in surface soil at Range K.

1,3,5-Trinitrobenzene. 1,3,5-Trinitrobenzene was detected in one surface soil sample (RNG-203-MW06) out of 13 samples at a concentration that exceeded the ESV. The calculated HQ_{screen} value for 1,3,5-trinitrobenzene was 2.1. Based on the infrequency of detection and the low level of the detected concentrations in relation to the ESV, 1,3,5-trinitrobenzene was not identified as a COPEC in surface soil at Range K.

Nitrotoluene. 2-Nitrotoluene, 3-nitrotoluene, and p-nitrotoluene were each detected in one surface soil sample out of 13 samples. There are no ESVs for these constituents in surface soil; therefore, no definitive statement can be made regarding the potential for toxicity posed by these constituents. Nitrotoluenes are considered only slightly toxic, especially when compared to nitrobenzene (Spectrum, 2004), which has a soil ESV of 1.3 mg/kg. All of the detected nitrotoluene concentrations are less than the ESV for nitrobenzene; therefore, the detected

concentrations of nitrotoluenes are expected to pose negligible risk to ecological receptors. Based on the infrequency of detection and the low detected concentrations relative to more toxic compounds (e.g., nitrobenzene), none of the nitrotoluene compounds were considered COPECs in surface soil at Range K.

Trichloroethene. Trichloroethene was detected in one surface soil sample (RNG-203-MW06) out of 13 samples at a concentration that exceeded the ESV. The calculated HQ_{screen} value for trichloroethene was 16. An alternative ESV for trichloroethene of 12.4 mg/kg has been proposed by EPA Region 5 (2003c), based on toxicity to masked shrew. The single detection of trichloroethene is orders of magnitude less than this alternative ESV. Based on the infrequency of detection and the fact that the only detected value is significantly less than the alternative ESV, trichloroethene was not identified as a COPEC in surface soil at Range K.

7.6.2 Groundwater

A number of inorganic constituents in groundwater exceeded their respective ESVs, BTVs, and are not essential macronutrients. Additionally, several pesticides, SVOCs, and VOCs were also detected in groundwater samples from Range K. These constituents are discussed further below.

Aluminum. Aluminum was detected in two groundwater samples (RNG-203-MW01 and RNG-203-MW04) out of 11 samples at concentrations that exceeded the BTV. Aluminum passed the Slippage Test but failed the Hot Measurement Test; therefore, a geochemical evaluation was conducted. All of the detected aluminum in groundwater from Range K was determined to be naturally occurring through geochemical evaluation. Based on the relative infrequency of detection at elevated concentrations and the fact that geochemical analysis indicated that all of the detected aluminum in groundwater is naturally occurring, aluminum was not identified as a COPEC in groundwater at Range K.

Barium. Barium was detected in seven groundwater samples at concentrations that exceeded the BTV. Barium passed the Slippage Test but failed the WRS Test; therefore, a geochemical evaluation was conducted. All of the detected barium in groundwater from Range K was determined to be naturally occurring through geochemical evaluation. Based on the fact that all of the detected barium in groundwater was determined to be naturally occurring, barium was not identified as a COPEC in groundwater at Range K.

Cadmium. Cadmium was detected in one groundwater sample (RNG-203-MW04) out of 18 samples at a concentration that exceeded the BTV. Due to the low frequency of detection, the

Slippage Test and WRS Test could not be conducted. Cadmium failed the Hot Measurement Test; therefore, a geochemical evaluation was conducted. Cadmium in groundwater from Range K was determined to be naturally occurring through geochemical evaluation. Based on the infrequency of detection and the fact that all of the detected cadmium in groundwater was determined to be naturally occurring, cadmium was not identified as a COPEC in groundwater at Range K.

Chromium. Chromium was detected in one groundwater sample (RNG-203-MW04) out of 18 samples at a concentration that exceeded the ESV. The calculated HQ_{screen} value for chromium was 3.7. Due to the low frequency of detection, the Slippage Test and WRS Test could not be conducted. Chromium failed the Hot Measurement Test; therefore, a geochemical evaluation was conducted. All of the detected chromium in groundwater from Range K was determined to be naturally occurring through geochemical evaluation. The National Recommended Water Quality Criteria for trivalent chromium, assuming chronic exposures, is 0.074 mg/L. All of the detected concentrations of chromium were less than this alternative ESV. Based on the infrequency of detection, the fact that geochemical evaluation of the data indicates that the chromium in groundwater is naturally occurring, and all of the detected concentrations of chromium are less than the alternative ESV, chromium was not identified as a COPEC in groundwater at Range K.

Cobalt. Cobalt was detected in one groundwater sample (RNG-203-MW01) out of 18 samples at a concentration that exceeded the BTV. Cobalt passed the Slippage Test but failed the WRS Test; therefore, a geochemical evaluation was conducted. Cobalt in groundwater from Range K was determined to be naturally occurring through geochemical evaluation. Based on the infrequency of detection at elevated concentrations and the fact that all of the detected cobalt was determined to be naturally occurring, cobalt was not identified as a COPEC in groundwater at Range K.

Iron. Iron was detected in two groundwater samples (RNG-203-MW01 and RNG-203-MW04) out of 16 samples at concentrations that exceeded the BTV. Both the Slippage Test and the WRS Test concluded that the detected iron concentrations in groundwater at Range K were consistent with naturally occurring iron in groundwater at FTMC. Based on the infrequency of detection at elevated concentrations and the fact that the geochemical evaluation indicated that all of the detected iron in groundwater was naturally occurring, iron was not identified as a COPEC in groundwater at Range K.

Lead. Lead was detected in one groundwater sample (RNG-203-MW04) out of 17 samples at a concentration that exceeded the BTV. Due to the low frequency of detection, the Slippage Test and WRS Test could not be conducted. The Hot Measurement Test indicated all of the detected lead in groundwater at Range K was naturally occurring. Based on the low frequency of detection and the fact that statistical analysis of the data indicated that lead in groundwater was naturally occurring, lead was not identified as a COPEC in groundwater at Range K.

Manganese. Manganese was detected in two groundwater samples (RNG-203-MW01 and RNG-203-MW10) out of 18 samples at concentrations that exceeded the BTV. Manganese passed the Slippage Test but failed the WRS Test; therefore, a geochemical evaluation was conducted. All of the detected manganese in groundwater from Range K was determined to be naturally occurring through geochemical evaluation. Based on the low frequency of detection and the fact that geochemical evaluation of the data indicated that all of the detected manganese in groundwater was naturally occurring, manganese was not identified as a COPEC in groundwater at Range K.

Mercury. Mercury was detected in two groundwater samples (RNG-203-MW01 and RNG-203-MW04) out of 18 samples at concentrations that exceeded the ESV. Due to the low frequency of detection, the Slippage Test and WRS Test could not be conducted. The Hot Measurement Test indicated all of the detected mercury in groundwater at Range K was naturally occurring. The National Recommended Water Quality Criterion (EPA, 2002d) for mercury is 0.77 µg/L. All of the detected concentrations of mercury in groundwater were less than this alternative ESV. Based on the infrequency of detection, the fact that the detected mercury in groundwater was determined to be consistent with naturally occurring levels, and the detected concentrations of mercury were all less than the alternative ESV, mercury was not identified as a COPEC in groundwater at Range K.

Vanadium. Vanadium was detected in one groundwater sample (RNG-203-MW04) out of 17 samples at a concentration that exceeded the ESV. The calculated HQ_{screen} value for vanadium was 2.4. Due to the low frequency of detection, the Slippage Test and WRS Test could not be conducted. Vanadium failed the Hot Measurement Test; therefore, a geochemical evaluation was conducted. All of the detected vanadium in groundwater from Range K was determined to be naturally occurring through geochemical evaluation. Based on the infrequency of detection, the relatively low HQ_{screen} value, and the fact that geochemical evaluation of the data indicates that the detected vanadium in groundwater is naturally occurring, vanadium was not identified as a COPEC in groundwater at Range K.

alpha-Chlordane. alpha-Chlordane was detected in one groundwater sample (RNG-203-MW15) out of 14 samples at a concentration that exceeded the ESV. alpha-Chlordane does not have an ESV; therefore, no definitive statement can be made regarding its toxicity. The only detected concentration of alpha-chlordane was an estimated concentration and was less than the ESV for other similar compounds. Based on the infrequency of detection and the low level detected, alpha-chlordane was not identified as a COPEC in groundwater at Range K.

Heptachlor. Heptachlor was detected in one groundwater sample (RNG-203-MW15) out of 14 samples collected at a concentration that exceeded the ESV. The calculated HQ_{screen} value for heptachlor was 8.4. The surface water ESV for heptachlor is the chronic value referenced from the *National Recommended Water Quality Criteria for Priority Toxic Pollutants* (EPA, 2002d). The acute value from the same reference is 0.52 $\mu\text{g/L}$. The only detected heptachlor concentration in groundwater was less than this acute value. It may be more appropriate to compare groundwater constituent concentrations to acute surface water screening values since potential ecological exposures to groundwater only occur at the groundwater-surface water interface. Therefore, long-term (chronic) exposures to groundwater by surface water receptors are unlikely. Since heptachlor was infrequently detected, the HQ_{screen} value was relatively low, and the only detected concentration was less than the acute ambient water quality criterion, heptachlor was not identified as a COPEC in groundwater at Range K.

Methoxychlor. Methoxychlor was detected in one groundwater sample (RNG-203-MW06) out of 14 samples at an estimated (“J”-flagged) concentration that exceeded the ESV. The calculated HQ_{screen} value for methoxychlor was 4.7. Based on the infrequency of detection and the relatively low detected concentration, methoxychlor was not identified as a COPEC in groundwater at Range K.

Bis(2-Ethylhexyl)phthalate. Bis(2-ethylhexyl)phthalate was detected in one groundwater sample (RNG-203-MW13) out of 18 samples at an estimated concentration that exceeded the ESV. The Tier II Secondary Acute value for bis(2-ethylhexyl)phthalate as calculated using the methodologies presented in the *Final Water Quality Guidance for the Great Lakes System* is 0.027 mg/L (EPA, 1995). The only detected concentration of bis(2-ethylhexyl)phthalate is less than this acute value. It may be more appropriate to compare groundwater constituent concentrations to acute surface water screening values since potential ecological exposures to groundwater only occur at the groundwater-surface water interface. Therefore, long-term (chronic) exposures to groundwater by surface water receptors are unlikely. Since bis(2-

ethylhexyl)phthalate was infrequently detected and the only detected concentration was less than the acute ambient water quality criterion, bis(2-ethylhexyl)phthalate was not identified as a COPEC in groundwater at Range K.

1,1,2,2-Tetrachloroethane. 1,1,2,2-Tetrachloroethane was detected in two groundwater samples (RNG-203-MW20 and RNG-203-MW18) out of 74 samples at concentrations that exceeded the ESV. The calculated HQ_{screen} value for 1,1,2,2-tetrachloroethane was 1.3. The EPA Region 5 ESL for 1,1,2,2-tetrachloroethane is 0.38 mg/L. All of the detected concentrations of 1,1,2,2-tetrachloroethane in groundwater at Range K are less than this value. Based on the fact that 1,1,2,2-tetrachloroethane was infrequently detected, the HQ_{screen} value is relatively low, and all of the detected concentrations are less than the alternative ESV, 1,1,2,2-tetrachloroethane was not identified as a COPEC in groundwater at Range K.

Vinyl Chloride. Vinyl chloride was detected in one groundwater sample (RNG-203-MW01) out of 74 samples at a concentration that exceeded the ESV. The calculated HQ_{screen} value for vinyl chloride was 4.7. The EPA Region 5 ESL for vinyl chloride is 0.93 mg/L. All of the detected concentrations of vinyl chloride in groundwater at Range K are less than this value. Based on the fact that vinyl chloride was infrequently detected, the HQ_{screen} value is relatively low, and all of the detected concentrations are less than the alternative ESV, vinyl chloride was not identified as a COPEC in groundwater at Range K.

Based on the lines of evidence presented above, none of the constituents detected in surface soil or groundwater at Range K have been identified as COPECs.

7.7 Uncertainty Analysis

Uncertainties are inherent in any risk assessment, and even more so in a SLERA due to the nature of the assessment process and the assumptions used in the process. A number of the major areas of uncertainty in this assessment are presented below.

An area of uncertainty that is inherent in a SLERA is the use of the maximum detected constituent concentration as the exposure point concentration for all receptors in a given medium. Most receptors have a home range large enough that precludes individuals from being exposed to the maximum constituent concentration for their entire lifetimes. Therefore, the actual exposure point concentration of a given constituent for most receptor species would be less than the maximum detected concentration. The use of the maximum detected constituent

concentrations as the exposure point concentrations for all receptors results in an overestimation of exposure for many receptors.

Additionally, there is no consideration given to the bioavailability of COPECs to different organisms. In this SLERA it is assumed that all constituents are 100 percent bioavailable to all receptor organisms. It is known that many constituents (particularly inorganic compounds) have significantly lower bioavailabilities (i.e., 1 to 10 percent for some inorganics in soil) than the 100 percent that was assumed in this assessment. This assumption has the potential to greatly overestimate exposures to certain COPECs.

Several COPECs (e.g., nitrotoluene and alpha-chlordane) do not have ESVs. The lack of toxicity data for certain COPECs makes it impossible to determine the potential for ecological risk posed by those constituents. Risks may be under- or over-estimated due to this uncertainty.

The ESVs used in this assessment are all the most conservative values from the scientific literature and many are based on the most sensitive endpoint (NOAEL values) for the most sensitive species tested. A less sensitive endpoint that is still protective of the ecological populations or communities of interest may be the LOAEL or some other endpoint. The use of NOAEL-based ESVs may overestimate potential for risks from certain COPECs. Additionally, certain ESVs may not be applicable to conditions at Range K. For instance, the soil ESVs do not take into account site-specific conditions at Range K and, thus, introduce a potentially significant level of uncertainty into the assessment.

It is important to note that the chromium ESV is based on toxicity studies using Cr^{+6} , which is more mobile and generally more toxic than Cr^{+3} . Measured chromium concentrations in surface soil are total chromium values. It is unknown what portion, if any, of the measured chromium in surface soil at Range K is in the hexavalent form. Because it is likely that only a portion of the measured chromium in surface soil is in the hexavalent form, the ESV and calculated $\text{HQ}_{\text{screen}}$ value are highly conservative.

Another area of uncertainty is the inherent limitations of the hazard quotient method for estimating risks. Hazard quotients (HQ) are not explicit expressions of risk (i.e., they are not probabilities of toxicological effects occurring in an ecological population). Additionally, because HQs are ratios, after unity has been exceeded, the magnitude of the HQ has little bearing on the potential severity of adverse effects that may be anticipated. An HQ of five does not

indicate the potential ecological risk is greater than an HQ of three. HQs are not population measures, but rather measures based on sensitive individuals from a test population.

Another area of uncertainty is the lack of consideration of synergisms and/or antagonisms between COPECs. Although it is widely accepted that synergisms and antagonisms occur between certain constituents under certain conditions, the SLERA process does not provide methods for assessing these potential synergisms/antagonisms.

One significant source of uncertainty that may impart a non-conservative bias on the SLERA results is the exclusion of metals determined to be present at concentrations comparable to naturally occurring background concentrations from consideration as COPECs. The constituents excluded as COPECs based solely on their comparison to background concentrations are discussed below.

As noted above, the exclusion of constituents as COPECs based on comparison to naturally occurring levels is performed via a three-tiered protocol (Shaw, 2003c). Tier 1 – comparison of the maximum detected constituent concentration to the BTV – is generally considered to be sufficiently conservative so that the uncertainty associated with chemicals eliminated in this tier of the protocol is minimal. Therefore, only chemicals excluded as COPECs via Tiers 2 or 3 of the background screening protocol are discussed herein.

Surface Soil. Aluminum, arsenic, barium, chromium, cobalt, iron, and vanadium were eliminated from consideration as COPECs in surface soil based solely on statistical and geochemical comparisons to naturally occurring levels in soil. The calculated HQ_{screen} values for arsenic, barium, and cobalt were 2.6, 2.1, and 1.98, respectively. Additionally, only one sample exhibited an elevated concentration of barium and two samples exhibited elevated concentrations of arsenic and cobalt. Because the maximum concentrations of arsenic, barium, and cobalt only slightly exceeded their highly conservative ESVs, and because they were only sporadically detected at these elevated concentrations, elimination of these three metals from the list of COPECs imparted minimal uncertainty on the results of this SLERA.

Per EPA (2000) guidance, aluminum toxicity is associated with soluble aluminum only. Numeric screening values for aluminum are considered inappropriate. Alternatively, potential ecological risks associated with exposure to aluminum are associated with soil pH. Aluminum is identified as a COPEC only if the soil pH is less than 5.5 (EPA, 2000). Since the pH of soils at

Range K is greater than 5.5, the exclusion of aluminum from the list of COPECs introduced minimal uncertainty into this SLERA.

Iron was detected in two of 13 surface soil samples at concentrations greater than the BTV. Iron is an essential macronutrient; thus, most organisms have mechanisms to regulate the levels of iron within their systems. As such, iron is only toxic at very high concentrations. Since iron was infrequently detected at elevated concentrations relative to background and because it is a macronutrient that is easily regulated, the level of uncertainty introduced into the results of the SLERA by eliminating iron from the list of COPECs is expected to be minimal.

Chromium was only detected in 1 of 13 samples at an elevated concentration and vanadium was detected in 2 of 13 samples at elevated concentrations, so the frequency of detection of these two constituents at elevated concentrations is low. Because the incidence of elevated concentrations of chromium and vanadium is low, the uncertainty introduced to the results of the SLERA by eliminating these two constituents is most likely minimal; however, the elimination of these two constituents remains a source of uncertainty in this SLERA.

Groundwater. Aluminum, barium, cadmium, chromium, cobalt, iron, lead, manganese, mercury, and vanadium were eliminated from consideration as COPECs in groundwater based solely on statistical and geochemical comparisons to naturally occurring levels in groundwater. The HQ_{screen} values for chromium and vanadium were calculated to be 3.7 and 2.4, respectively. Additionally, only one sample exhibited elevated concentrations of chromium and vanadium with respect to their ESVs. Because the maximum concentrations of chromium and vanadium only slightly exceeded their highly conservative ESVs and because these metals were only sporadically detected at elevated concentrations, elimination of these two metals from the list of COPECs imparted minimal uncertainty on the results of this SLERA.

Iron was detected in 2 of 16 groundwater samples at concentrations greater than the BTV. Iron is an essential macronutrient; thus, most organisms have mechanisms to regulate the levels of iron within their systems. As such, iron is only toxic at very high concentrations. Since iron was infrequently detected at elevated concentrations relative to background and it also is a macronutrient that is easily regulated, the level of uncertainty introduced into the results of the SLERA by eliminating iron from the list of COPECs is expected to be minimal.

Cadmium was only detected in 1 of 18 groundwater samples at a concentration that exceeded the BTV. The only detected concentration of cadmium in groundwater (0.011 mg/L) was less than

the National Recommended Water Quality Criterion (EPA, 2002d) for cadmium (0.25 mg/L), based on chronic exposures. Because cadmium was infrequently detected and the only detected concentration was less than the National Recommended Water Quality Criterion for cadmium, the level of uncertainty introduced into the results of this SLERA by eliminating cadmium from the list of groundwater COPECs is considered insignificant.

Mercury was only detected in 2 of 18 groundwater samples at concentrations that exceeded the ESV. The maximum detected concentration of mercury in groundwater (0.00017 mg/L) was less than the National Recommended Water Quality Criterion (EPA, 2002d) for mercury (0.00077 mg/L), based on chronic exposures. Because mercury was infrequently detected and the maximum detected concentration was less than the National Recommended Water Quality Criterion for mercury, the level of uncertainty introduced into the results of this SLERA by eliminating mercury from the list of groundwater COPECs is considered insignificant.

Although aluminum, barium, cobalt, lead, and manganese were detected infrequently at elevated concentrations in groundwater, the exclusion of these constituents from the list of groundwater COPECs remains a source of uncertainty in this SLERA

7.8 Summary and Conclusions

The potential for ecological risks at Range K was determined through a SLERA. This ecological screening process consisted of a characterization of the ecological setting at Range K, development of a SCM, a description of the fate and transport of constituents detected in various environmental media, a description of the ecotoxicity of the various constituents detected at Range K, a description of the ecological receptors, a description of the complete exposure pathways, calculation of screening-level hazard quotients, and a description of the uncertainties within the process.

7.8.1 Summary

Range K is entirely forested with a relatively immature mixed deciduous/coniferous forest. There are no surface water bodies within the Range K study area; however, a small ephemeral drainage feature is present approximately 200 feet south-southeast of the study area. Water is present in this drainage feature only during periods of significant precipitation. Range K is small (2 acres) in size; therefore, its ecological relevance is limited with respect to potential ecological exposures.

Several constituents were detected in surface soil at Range K that exhibited maximum concentrations that exceeded their respective ESVs and BTVs. These constituents were the following: aluminum, arsenic, barium, beryllium, cadmium, chromium, cobalt, iron, nickel, selenium, vanadium, zinc, 1,3,5-trinitrobenzene, and trichloroethene. These fourteen constituents exhibited maximum hazard quotients (HQ_{screen}) ranging from 1.4 to 344. Hazard quotients for 2-nitrotoluene, 3-nitrotoluene, p-nitrotoluene, cumene, and p-cymene could not be calculated because ESVs were not available for these constituents.

A more rigorous comparison of site and background data was performed for inorganic constituents in soil. This integrated statistical and geochemical evaluation indicated that detected concentrations of aluminum, arsenic, barium, chromium, cobalt, iron, and vanadium are naturally occurring and are not present as a result of site-related activity. Additional lines of evidence were also utilized to further assess the constituents detected in surface soil. Based on the statistical and geochemical background evaluation and other lines-of-evidence, none of the detected constituents in surface soil were identified as COPECs at Range K.

A number of constituents were detected in groundwater at Range K that exhibited maximum concentrations exceeding their respective ESVs and BTVs. These constituents were the following: aluminum, barium, cadmium, chromium, cobalt, iron, lead, manganese, mercury, vanadium, heptachlor, methoxychlor, bis(2-ethylhexyl)phthalate, 1,1,2,2-tetrachloroethane, and vinyl chloride. These fifteen constituents exhibited maximum hazard quotients (HQ_{screen}) ranging from 1.3 to 154. Hazard quotients for alpha-chlordane, 2-nitrotoluene, p-nitrotoluene, 1,4-dithiane, 1,4-oxathiane, 1,2,4-trimethylbenzene, 1,3,5-trimethylbenzene, and p-cymene could not be calculated because ESVs were not available for these constituents.

A more rigorous comparison of site and background data was performed for inorganic constituents in groundwater. This integrated statistical and geochemical evaluation indicated that detected concentrations of aluminum, arsenic, barium, cadmium, chromium, cobalt, iron, lead, manganese, mercury, and vanadium are naturally occurring and are not present as a result of site-related activity. Additional lines of evidence were also used to further assess the constituents detected in groundwater. Based on the statistical and geochemical background evaluation and other lines-of-evidence, none of the detected constituents in groundwater were identified as COPECs in groundwater at Range K.

7.8.2 Conclusions

The SLERA for Range K determined that none of the constituents detected in surface soil or groundwater were COPECs. Based on the conservative assessment techniques used in this SLERA and considering additional lines-of-evidence, none of the detected constituents were deemed to pose significant ecological risks. Additionally, the small size of Range K limits its ecological relevance with respect to potential ecological exposures. Therefore, no further ecological assessment is warranted at Range K.

8.0 Summary and Conclusions

This chapter summarizes the results of the RI at Range K and presents the major conclusions.

8.1 Summary

8.1.1 Geological and Hydrogeological Data

Based on data collected during the drilling of 25 monitoring wells, the geology at Range K is characterized by 7 to 37 feet of residuum overlying fractured shale, limestone, and interbedded shale and limestone of the upper Mississippian Parkwood and Floyd Shale Undifferentiated and the upper Mississippian Tusculumbia Limestone. The residuum consists of yellowish-orange to light brown clay and reddish-brown clayey silt. The majority of Range K was composed of moderately to intensely fractured shale bedrock. The shale was interbedded with limestone in some areas. A transitional zone of highly weathered shale with interbedded sandstone and limestone separates the residuum and the bedrock in the eastern section of Range K.

Slickensides were noted in numerous rock cores, which probably indicates some degree of faulting and/or movement of subsurface materials (namely shales). Fossiliferous, fractured limestone was present in the southern and southeastern section of Range K.

Groundwater elevation data collected from periods of high precipitation (February) and low precipitation (October) show an overall southerly groundwater flow direction. Horizontal hydraulic gradients are relatively low, with an approximate 7.5-foot elevation difference across the study area. Hydraulic conductivity values calculated from slug tests yielded geometric mean values of 3.97 ft/day for residuum and 4.05 ft/day for bedrock. Average linear groundwater flow velocities were calculated to be 0.44 ft/day for residuum and 1.05 ft/day for bedrock.

8.1.2 Surface and Subsurface Soil Contaminant Distribution

Thirteen surface soil samples and 13 subsurface soil samples were collected during RI field activities at Range K. Metals, VOCs, SVOCs, pesticides, and explosives were detected in the soil samples. Statistical and geochemical evaluation of site metals data concluded that the metals detected in soils were naturally occurring except for four metals (beryllium, cadmium, nickel, and zinc) in one surface soil sample and five metals (beryllium, cadmium, nickel, selenium, and zinc) in a limited number of subsurface soil samples (i.e., 1 to 3 samples). The detected concentrations of organic compounds in soils were all below SSSLs. However, one VOC

(trichloroethene) and one explosive compound (1,3,5-trinitrobenzene) exceeded their respective ESVs in one surface soil sample each.

8.1.3 Groundwater Contaminant Distribution

Seventy-four groundwater samples were collected from 25 monitoring wells installed at Range K. Metals were detected in groundwater samples at concentrations exceeding SSSLs and background. However, all of the metals detected were determined to be naturally occurring based on an integrated statistical and geochemical evaluation.

Organic compounds detected in groundwater were VOCs, SVOCs, pesticides, explosives, and two CWM breakdown products. One SVOC (bis[2-ethylhexyl]phthalate), two pesticides (heptachlor and beta-BHC), one explosive compound (nitrobenzene), and one CWM breakdown product (1,4-oxathiane) exceeded their respective SSSLs in one sample each. However, the most significant groundwater contamination was chlorinated VOCs. Specifically, the concentrations of 12 VOCs exceeded SSSLs: 1,1,2,2-tetrachloroethane, 1,1,2-trichloroethane, 1,1-dichloroethane, 1,2-dichloroethane, benzene, chloroform, cis-1,2-dichloroethene, trans-1,2-dichloroethene, methylene chloride, tetrachloroethene, trichloroethene, and vinyl chloride.

Isoconcentration maps of total chlorinated VOCs, trichloroethene, and 1,1,2,2-tetrachloroethene reflect the same overall pattern of horizontal distribution and plume migration with the flow of groundwater. Contaminant flow distribution is to the southwest, which coincides with the general direction of groundwater flow. The total chlorinated VOC plume ranges from approximately 75 feet wide (near the southeastern corner of the site) to approximately 300 feet wide several hundred feet south of Range K. The plume is approximately 800 feet long. The vertical extent of contamination ranges from approximately 10 to 20 feet deep (near the southeastern corner of the site) to greater than 200 feet deep (approximately 700 feet south of Range K). The horizontal and vertical extent of these chlorinated VOCs in groundwater has been defined. Although the source area for the groundwater contamination has not been fully established, the most likely source of contamination is the discharge of decontamination agents directly onto the ground surface. The maximum concentration of chlorinated VOCs is centered near monitoring well RNG-203-MW06 in residuum and further downgradient in bedrock around RNG-203-MW18 and RNG-203-MW20. Although chlorinated VOCs were not detected in the sludge sample collected from the partially exposed drum near RNG-203-MW01, this area cannot be precluded as a source area.

Based on the soil and groundwater data collected at the site, the most likely fate and transport pathway is the leaching of organic contaminants within subsurface soils and movement through the residuum into the bedrock aquifer system.

8.1.4 Streamlined Human Health Risk Assessment

An SRA was performed to determine the potential threat to human health from exposure to environmental media at Parcel 203(7). Five receptor scenarios were evaluated in the SRA: national guardsperson, groundskeeper, construction worker, recreational site user, and resident. Soils at Range K are lightly contaminated with metals, organochlorine pesticides, nitroaromatic compounds, and VOCs, but concentrations were sufficiently low so that no chemicals were selected as COPCs. Groundwater, however, was more heavily contaminated with organochlorine pesticides, nitroaromatic explosives, orthosulfur compounds, SVOCs, and VOCs. COPCs in groundwater included low levels of two organochlorine pesticides, a nitroaromatic compound, DEHP (probably present as a laboratory contaminant), and several VOCs largely limited to chlorinated solvents.

The SRA concluded that Range K in its current condition does not pose an unacceptable health risk to any of the receptor scenarios evaluated unless the groundwater is developed as a potable source. Exposure to groundwater developed as a potable source would pose an unacceptable cancer risk to the national guardsperson, groundskeeper, and resident, and would pose a noncancer hazard to the resident. The primary risk drivers are 1,1,2,2-tetrachloroethane, trichloroethene, and vinyl chloride.

8.1.5 Screening-Level Ecological Risk Assessment

A SLERA was performed to evaluate potential risks to ecological receptors from exposure to surface soil and groundwater at Range K. The SLERA identified several constituents in surface soil at Range K that exceeded their respective ESVs and background (where available). These constituents included twelve metals (aluminum, arsenic, barium, beryllium, cadmium, chromium, cobalt, iron, nickel, selenium, vanadium, and zinc), one VOC (trichloroethene), and one explosive (1,3,5-trinitrobenzene).

In groundwater, a number of constituents exceeded their ESVs and background (where available), including ten metals (aluminum, barium, cadmium, chromium, cobalt, iron, lead, manganese, mercury, and vanadium), two pesticides (heptachlor and methoxychlor), one SVOC (bis[2-ethylhexyl]phthalate), and two VOCs (1,1,2,2-tetrachloroethane and vinyl chloride). However, based on the results of the statistical and geochemical evaluation and upon

consideration of additional lines of evidence, the SLERA determined that none of the detected constituents pose significant ecological risks. Therefore, the SLERA concluded that no further ecological assessment is warranted at Range K.

8.2 Conclusions

Based on the summary presented above, no further actions are warranted with respect to defining the extent of contamination in soils at Range K. Furthermore, the horizontal and vertical extent of groundwater contamination has been defined.