

### **3.0 Ecotoxicity**

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The ecotoxicological properties of the COPECs identified at the IMR ranges dictate which receptors have the greatest potential ecological risk and the pathways by which those receptors have the greatest potential for exposure. Factors such as the propensity to bioaccumulate or biomagnify, as well as their acute and/or chronic toxicity to immature or adult receptors are important factors in the consideration of a constituent's ecotoxicity and also in the development of assessment and measurement endpoints.

In order for a constituent to exhibit toxicity or to bioaccumulate, it first must be bioavailable. In general, there are three microbial processes affecting the bioavailability of metals (Connell and Miller, 1984). The first is biodegradation of organic matter into lower molecular weight compounds, which are more capable of complexing metal ions than the higher molecular weight organic molecules. The second is alterations to physico-chemical properties of metals by microbial metabolic activities (i.e., oxidation-reduction potential and pH conditions). Finally, the process of bacterial methylation, specifically of lead, may greatly enhance the bioavailability of certain inorganic compounds.

The actual uptake of bioavailable metals by terrestrial and aquatic organisms is through three main routes: 1) uptake across respiratory surfaces (lungs or gills), 2) adsorption from soil, sediment or water onto body surfaces, and 3) ingestion of food, water or incidental particles. Given the state of science relative to bio-uptake dynamics, the ingestion route is the most quantifiable uptake route at this time. Metal uptake from dietary sources, in comparison to direct adsorption, is also considered the primary uptake route in small terrestrial and aquatic receptors.

Although ecological receptors can readily absorb metals from food/water ingestion, their ability to regulate elevated concentrations of metals dictates their tolerance and is a critical factor in survival. Once the upper limit, or threshold, of metal sequestration and excretion is reached, sub-lethal effects such as inhibited reproduction and growth potentials may be exhibited, followed by lethality. Temporary metal storage is generally by binding to proteins, such as metallothioneins, polysaccharides, and amino acids (Connell and Miller, 1984). Storage within liver and kidney tissues as well as bone, feathers, and fur also provide a useful means for sequestering metals such as lead.

Considerable inter- and intra-species differences exist in bioaccumulation potential of individual metals. In addition, according to Phillips (1980), different chemical forms of any one metal may

be absorbed and excreted at widely differing rates. Many studies support the premise that inorganic metals do not have a high propensity to biomagnify up through food chains.

The sections that follow highlight key toxicological properties of the COPECs that have been identified at the IMR ranges (antimony, arsenic, barium, copper, lead, manganese, thallium, and zinc).

### **3.1 Antimony**

Antimony binds to soil and particulates (especially those containing iron, manganese, or aluminum) and is oxidized by bacteria in soil. Exposure routes for mammals include ingestion and inhalation. Antimony does not tend to biomagnify in terrestrial food chains (Ainsworth, 1988), and is not significantly metabolized and excreted in the urine and feces. Antimony at elevated levels has the potential to cause reproductive, pulmonary, and hepatic effects in mammals (EPA, 1999).

**Plants.** Antimony is considered a non-essential element and is easily taken up by plants if available in the soil in soluble forms (Kabata-Pendias and Pendias, 1992). A screening level of 5.0 mg/kg has been proposed by Kabata-Pendias and Pendias (1992) based on a report of unspecified phytotoxic responses by plants grown in soil amended with antimony.

**Mammals.** Female mice exposed to 5.0 milligrams per liter (mg/L) antimony as antimony potassium tartrate in their drinking water showed a reduction in their lifespan. This dose was equivalent to a lowest-observed-adverse-effects-level (LOAEL) of 1.25 mg/kg/ per day [mg/kg/day], which can be converted to a no-observed-adverse-effects-level (NOAEL) of 0.125 mg/kg/day (Integrated Risk Information Service, 2001).

Laboratory data on antimony toxicity (as antimony potassium tartrate) in laboratory mice through drinking water ingestion were used to estimate a chronic NOAEL value of 0.125 mg/kg/day (Schroeder, et al., 1968). Lifespan and longevity were the endpoints tested.

**Aquatic Life.** The available data for antimony indicate that acute and chronic toxicity to freshwater aquatic life occur at concentrations as low as 9.0 and 1.6 mg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested. Toxicity to algae can occur at concentrations as low as 0.61 mg/L.

Effects from antimony exposure on benthic community composition have been detected at levels between 3.2 and 150 mg/kg (Long and Morgan, 1990). Data on antimony suggest an effects range-low (ER-L) of 2 mg/kg and an effects range-medium (ER-M) of 25 mg/kg.

### **3.2 Arsenic**

Arsenic occurs naturally 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], 1977). 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 resolubilized, and the arsenic is being presented for plant uptake or reduction by organisms and chemical processes (NAS, 1977). Among elements, arsenic ranks 20th in abundance 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 in trivalent and pentavalent states (Pershagen and Vahter, 1979).

**Plants.** The NAS (1977) reports background arsenic concentrations in terrestrial plants as ranging from 0.01 to 5 mg/kg (dry weight). Generally, the 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 Efrogmson, 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.** In general, soil microorganisms are capable of tolerating and metabolizing relatively high concentrations of arsenic (Wang, et al., 1984). Tolerant soil microbiota can withstand concentrations up to 1,600 mg/kg; however, growth and metabolism were reduced in sensitive species at 375 mg/kg and, at 150 to 165 mg/kg, soils were devoid of earthworms and showed diminished quantities of bacteria and protozoans (NRCC, 1978). An ecological benchmark value of 60 mg/kg has been proposed by Efrogmson, et al., (1997) based on survival and reproduction in *Eisenia fetida* exposed to potassium arsenate.

**Mammals.** Arsenic is more toxic to wildlife in its 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, 1977). 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 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 lethal dose for 50 percent population tested (LD<sub>50</sub>) 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 LC<sub>50</sub> 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, 1977). The grey partridge (*Perdix perdix*) succumbed to 300 mg/kg body weight of lead arsenate in 52 hours (NAS, 1977).

**Aquatic Life.** Toxic and other effects of arsenicals to aquatic life are significantly modified by numerous biological and abiotic factors (Woolson, 1975; NAS, 1977; National Research Council of Canada (NRCC), 1978; EPA, 1980a; Howard, et al., 1984; Michnowicz and Weak, 1984; Bryant et al., 1985; EPA, 1985a; Sanders, 1986). The LC<sub>50</sub> values, for example, are markedly affected by water temperature, pH, Eh, organic content, phosphate concentration, suspended solids, and the presence of other substances and toxicants, as well as arsenic speciation and duration of exposure. In general, inorganic arsenicals are more toxic to aquatic

biota than organoarsenicals, 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, 1985a). An adult bluegill population was reduced 42 percent after several monthly applications of 4 mg/L trivalent arsenic (NAS, 1977).

As with fish, toxic and other effects of arsenicals to aquatic invertebrates are significantly modified by numerous biological and abiotic factors (Woolson, 1975; NAS, 1977; 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).

### **3.3 Barium**

Barium, a silvery-white metal, is used in various alloys, in paints, soap, paper, 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, 1992).

**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 Efroymsen, et al. (1997) as a benchmark screening value for barium phytotoxicity.

**Terrestrial Invertebrates.** The toxicity of barium to terrestrial invertebrates has not been studied extensively. A reduction in soil microbial activity was measured at a concentration of 3,433 mg/kg. Based on this study, an ecological benchmark value of 3,000 mg/kg has been suggested by Efrogmson, et al., (1997). The Netherlands has developed methods for deriving critical loads of metals in soils, which is defined as the amount of material that can be present without causing harm to terrestrial organisms within the context of the specific soil chemistry (e.g., pH, organic matter, and cation exchange capacity). The EPA Region IV soil screening value of 165 mg/kg is based on the critical load derived by the Netherlands (RIVM, 1997).

**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) (IRIS, 2001). Laboratory rat toxicity data for barium chloride in drinking water was used to calculate a NOAEL value of 5.1 mg/kg/day. Growth and hypertension were the test endpoints.

**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 *Proposed Water Quality Guidance for the Great Lakes System* (EPA, 1985b), are 110 and 4.0 micrograms per liter ( $\mu\text{g/L}$ ), respectively

### 3.4 Copper

Copper is ubiquitously distributed in nature in the free state and in sulfides, arsenides, chlorides, and carbonates. Several copper containing proteins have been identified in biological systems as oxygen binding hemocyanin, cytochrome oxidase, tyrosinase, and lactase. Copper has also been identified with the development of metalloproteins employed in the sequestering and cellular detoxification of metals. Most organisms are able to regulate copper levels within their systems. Copper may accumulate in the tissues of certain organisms, but it does not tend to accumulate or magnify in higher trophic levels.

Copper has been known to sorb rapidly to sediment. The rate of sorption is of course dependent upon factors such as the sediment grain size, organic fraction, pH, competing cations, and the presence of ligands. In industrialized freshwater environments around the world total copper levels within sediments can range from 7 to 2,350 parts per million (ppm) (Moore and Ramamoorthy, 1984).

**Plants.** Copper is an essential nutrient for the growth of plants. Background concentrations of copper in grasses and clovers collected in the United States averaged 9.6 mg/kg and 16.2 mg/kg (dry weight) (Kabata-Pendias and Pendias, 1992). Copper is one of the least mobile heavy metals in soil, and its availability to plants is highly dependent on the molecular weight of soluble copper complexes (Kabata-Pendias and Pendias, 1992).

According to Rhodes, et al. (1989), copper concentrations in plant tissues do not serve as conclusive evidence of copper toxicity in species of plants such as tomatoes, because some species are able to tolerate higher concentrations of copper than others. The pH of soil may also influence the availability and toxicity of copper in soils to plants (Rhodes, et al., 1989). In a study with tomato plants, Rhodes, et al. (1989) found a reduction in plant growth when plants were grown in soils containing greater than 150 mg/kg of copper at a pH of less than 6.5. At pH values greater than 6.5, soil copper concentrations of greater than 330 mg/kg were required to reduce plant growth.

Concentrations of copper in leaf tissue that are excessive or toxic to various plant species range from 20 to 100 mg/kg (dry weight) (Kabata-Pendias and Pendias, 1992). A soil concentration of 100 mg/kg has been proposed by Efraymson, et al., (1997) as a benchmark screening value for copper phytotoxicity in soil. General symptoms of copper toxicity in plants include the presence of dark green leaves followed by induced iron chlorosis; thick, short, or barbed wire roots; and depressed tillering (Kabata-Pendias and Pendias, 1992).

**Terrestrial Invertebrates.** Beyer (1982) and others have reported that copper concentrations in earthworms have been observed to be correlated with copper concentrations in soil. Further studies by Beyer (1990) indicate that copper can be more toxic to bioturbative earthworms than most metals. Research by Phillips (1980) suggests that copper and other metal accumulation within terrestrial invertebrates may vary significantly depending on soil conditions and other physical/chemical properties, and bioconcentration factors can approach  $10^4$ . The USEPA (2000) has derived a soil screening level (SSL) for copper of 61 mg/kg. This invertebrate SSL was based on reproductive and growth data from studies conducted with natural soils under conditions of high or very high bioavailability. The tests were conducted with highly soluble salts and neither aging nor weathering, which would lower bioavailability, was included in the experimental designs.

**Mammals.** Copper is an essential trace element to animals as well as plants (Callahan, et al., 1979), but becomes toxic at concentrations only slightly higher than essential levels (EPA, 1985). Copper is an essential element for hemoglobin synthesis and oxidative enzymes in

animals, and is absorbed by mammals following ingestion, inhalation, and dermal exposure. Once absorbed, copper is distributed to the liver, and is not metabolized (Marceau, et al., 1970). No evidence of bioaccumulation was obtained in a study of pollutant concentrations in the muscles and livers of 10 species of herbivorous, omnivorous, and carnivorous animals in Donana National Park in Spain (Hernandez, et al., 1985). Copper concentrations in small mammals collected from various uncontaminated sites ranged from 8.3 to 13.4 mg/kg (whole-body concentrations) (Talmage and Walton, 1991). Highest concentrations of copper tend to be in hair, followed in decreasing concentration by liver, kidney, and whole body (Hunter and Johnson, 1982). Among the small mammals collected, Hunter and Johnson (1982) found shrews (*Sorex araneus*) to contain the highest concentrations of copper. Mice were found to contain the lowest copper concentrations. Increased fetal mortality was observed in fetuses of mice fed more than 104 mg/kg-day of copper as copper sulfate (Lecyk, 1980). Increased mortality rates in mink offspring have been observed at levels above 3.21 mg/kg-day (Aulerich, et al., 1985).

Laboratory toxicity data for mink exposed to copper sulfate in their diet were used to estimate a NOAEL value of 11.7 mg/kg/day (Aulerich, et al., 1982). Reproduction was the endpoint studied. Symptoms of acute copper poisoning in mammals include vomiting, hypotension, melena, coma, jaundice, and death (Klaassen, et al., 1991). Selenium can act as an antidote for copper poisoning.

**Birds.** Laboratory toxicity data for one-day old chicks exposed to copper oxide in their diets were used to estimate a NOAEL value of 47 mg/kg/day (Mehring, et al., 1960). Growth and mortality were the endpoints studied.

**Aquatic Life.** Invertebrates inhabiting "polluted" freshwaters worldwide have been known to have tissue residues of copper ranging from 5 to 200 ppm (Moore and Ramamoorthy, 1984). Field studies have shown that there is virtually no accumulation of this metal through the food chain (Fuller and Averett, 1975). Studies by Kosalwat and Knight (1987) indicated that copper present in the substrate or sediment was significantly less toxic to chironomid species than overlying water column levels. The substrate copper concentration at which chironomid larval growth was reduced 50 percent (EC<sub>50</sub>) was 1,602 mg/kg. These researchers found that deformities in larval mouth parts were observed in elevated concentrations, and adult emergence was inhibited when the sediment concentration exceeded 1,800 mg/kg. Cairns, et al. (1984) reported copper toxicity in sediment for several chironomus midges and cladocerans with LC<sub>50</sub>s ranging from 681 to 2296 mg/kg.

Moore and Ramamoorthy (1984) reported that copper can be highly toxic to many aquatic plants and algae. Inhibition of growth can occur at levels as low as 0.1 mg/L. In some algal species, copper may inhibit electron transport during photosynthesis. In general, since low pH increases the proportion of free ions in solution, acidic waters may exhibit greater copper toxicity. However, Stokes (1975) reported the observance of algal adaptation to copper-tainted waters with certain species able to tolerate and flourish within highly copper-contaminated waters.

Moore and Ramamoorthy (1984) reported  $LC_{50}$  in fresh water fish ranging from 0.017 to 1.0 mg/L. Copper is similar to other metals in that its toxicity to fin fish is often greater within fresh water environments versus marine environments because of the lack of complexing agents within fresh water.

### **3.5 Lead**

Global production of lead from both smelter and mining operations has been high throughout the past century. Lead is commonly used in storage batteries as well as in 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 to plants of lead in soil 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 increased amounts of lead in soil (Natural Resources Conservation Commission, 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). Lead tends not to accumulate into plants from soil unless concentrations are very high (i.e., percentage levels). The tips of some trees, such as pine and fir, can accumulate lead from contaminated soil when contamination levels are high. Such conditions often occur at mining sites (National Library of Medicine, 1996). Lead inhibits plant growth, reduces photosynthesis, and reduces mitosis and water absorption

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, 1988). 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 (Bogges, 1977).

**Terrestrial Invertebrates.** Lead has been shown to accumulate in the tissues of lower trophic level organisms, including terrestrial invertebrates, but is not effectively transferred to higher trophic level organisms through the food web. Centipedes (*Lithobius variegatus*) that ate woodlice hepatopancreas did not assimilate lead even though the food contained concentrations that were many times greater than normally encountered. However, survival and reproduction were reduced in woodlice (*Porcellio scaber*) fed soil litter treated with 12,800 mg/kg lead (Beyer and Anderson, 1985). This concentration of lead is similar to the amount of lead reportedly associated with reductions in natural populations of decomposers, such as fungi, earthworms, and arthropods.

**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, 1988). Jenkins (1981) also reported that soil conditions of low alkalinity and low pH can enhance the potential for bioconcentration of lead in mammals, birds, mosses, lichens, lower animals, and higher plants. 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, 1988). 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, 1988).

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  $\delta$ -aminolevulinic acid dehydratase 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 (Boggess, 1977; Getz et al., 1977).

**Birds.** Most of the information on the effects of lead to terrestrial vertebrates is concerned with acute 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, 1988; 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 more 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. Hatchlings of chickens, Japanese quail, mallards, and pheasants are relatively tolerant to moderate lead exposure (Eisler, 1988). 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 (National Academy of Sciences [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, 1988). The EPA's National Recommended Water Quality Criteria for lead in freshwater are 65 micrograms per liter ( $\mu\text{g/L}$ ) for acute exposure and 2.5  $\mu\text{g/L}$  for chronic exposure (EPA, 1999). 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, 1988).

Physicochemical conditions within the water may also affect lead uptake and toxicity. Under conditions of low alkalinity (less than 50 microequivalents per liter) and low pH, lead can accumulate in fish, algae, mollusks, and benthic invertebrates (Wiener and Stokes, 1990). Irwin (1988) reported significant accumulations of lead in the Trinity River within mosquitofish, turtles, bullhead minnows, and crayfish. Nevertheless, lead concentrations were not higher in top-of-the-food-chain predators like gar than they were in mosquitofish, suggesting minimal biomagnification of lead.

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 a lead LC<sub>50</sub> 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 parts per billion (ppb) range, while Khangrot and Ray (1989) reported a *D. magna* LC<sub>50</sub> of 4.89 ppm.

### **3.6 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 to 5 percent 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 leaves (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 Efrogmson, 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).

**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/m<sup>3</sup> for five hours per day, five days a week, for 100 days (Klaassen, et al., 1991).

**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 EC<sub>20</sub> 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 EC<sub>20</sub> 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 EC<sub>20</sub> for daphnids is less than 1,100 µg/L (Suter and Tsao, 1996).

### **3.7 Thallium**

Thallium is widely distributed in trace amounts in the Earth's crust and is one of the more toxic metals. In the environment, thallium exists in either the monovalent (thallous) or trivalent (thallic) form. Thallium is chemically reactive with air and moisture, undergoing oxidation. Thallium is relatively insoluble in water. Thallium adsorbs to soil and sediment and is not transformed or biodegraded (Callahan, et al., 1979).

**Plants.** Thallium is not essential for plant growth. When soluble forms are available, thallium is readily taken up by plants and translocated to aerial parts, probably because of its similarity to potassium. Toxic effects on plants include impairment of chlorophyll synthesis and seed germination, reduced transpiration due to interference in stomatal processes, growth reduction, stunting of roots, and leaf chlorosis (Adriano, 1986). The phytotoxicity benchmark value of 1.0 mg/kg is based on unspecified toxic effects on plants grown in surface soil amended with 1.0 mg/kg thallium (Kabata-Pendias and Pendias, 1992).

**Mammals.** Birds and mammals are exposed to thallium via ingestion of soil, water, and plant material. In mammals, thallium is absorbed primarily from ingestion and is distributed to several organs and tissues, with the highest levels reported in the kidneys (Manzo, et al., 1982). Thallium exposure in mammals causes cardiac, neurologic, reproductive, and dermatological effects. Various effects and toxic responses have been reported, including paralysis and pathological changes in the liver, kidneys, and stomach mucosa in rabbits exposed to thallium (Tikhonova, 1967). Testicular toxicity in rats has also been reported (Formigli, et al., 1986).

Laboratory toxicity data for rats exposed to thallium sulfate in their drinking water were used to estimate a NOAEL value of 0.0074 mg/kg/day (Formigli, et al., 1986). Reproduction was the endpoint for this study.

**Aquatic Life.** In aquatic organisms, thallium is absorbed primarily from ingestion and thereafter bioconcentrates in the organism. Toxic effects have been observed in numerous aquatic organisms, including daphnia, fat-head minnow, bluegill sunfish, and others (EPA, 1980b). The Tier II secondary acute water quality value and secondary chronic water quality value for beryllium, as calculated by the method described in the EPA's *Proposed Water Quality Guidance for the Great Lakes System* (EPA, 1985b), are 110 and 12 µg/L, respectively.

The test EC<sub>20</sub> 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 EC<sub>20</sub> value for thallium is 81 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 EC<sub>20</sub> benchmark for daphnids has been determined to be less than 64 µg/L (Suter and Tsao, 1996).

### **3.8 Zinc**

Zinc is a naturally occurring element that 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. The NAS (1977) 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.** The USEPA (2000) has developed an ecological soil screening level (SSL) for zinc in soil of 120 mg/kg. This SSL was based on reproduction and population effects in experiments conducted with natural soils under conditions of high or very high zinc bioavailability. It is also important to note that in studies conducted with mixtures of cadmium, copper, and zinc, it was concluded that the three metals acted antagonistically. It has also been

shown that a decrease in pH and/or organic matter in the soil tends to decrease the concentration of zinc in soil at which toxic effects are observed (Spurgeon and Hopkin, 1996). Zinc has been shown to accumulate in earthworm species (Beyer, et al., 1982), but generally is not biomagnified through the food web.

**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 of 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 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 bioaccumulation (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 LC<sub>50</sub>s ranging from 10.13 to 12.5 ppm in hard water for bluegills (*Lepomis macrochirus*), and 96-hour LC<sub>50</sub>s 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 µg/L for flagfish (*Jordanella floridae*) to 852 µg/L for brook trout (*Salvenius fontinalis*) (EPA, 1980c).

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 LC<sub>50</sub> 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, 1980c). Chronic testing with the saltwater species *Mysidopsis bahia* resulted in a chronic value of 166 µg/L (EPA, 1980c).

## **4.0 Fate and Transport**

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The environmental fate and transport of the COPECs in the various media at the IMR ranges will govern the potential for exposures to wildlife. In general, COPECs 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 Chapter addresses the mechanisms by which COPECs can be transported and the chemical properties that determine their transport.

### **4.1 Fate and Transport in Soil**

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

Several volatile organic compounds (VOC) were identified in the upper soil horizons at the IMR ranges. 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 VOCs detected in surface soil at the IMR ranges are low; therefore, this transport mechanism is expected to be insignificant with respect to other transport mechanisms active at this site. Most of the metals and semivolatile organic compounds (SVOC) in the surface soil at the IMR ranges are not expected to volatilize to any great extent, with the exception of mercury, which would be expected to volatilize relatively rapidly. Most of the metals and semivolatile organic compounds in the surface soil at the IMR ranges are generally closely associated with particulate matter and would be transported from their source areas by fugitive dust generation and entrainment by the wind. Subsequent dispersion by atmospheric mixing could transport particulate-associated contaminants to other parts of the IMR ranges and to off-site locations. The generation of fugitive dust and subsequent transport by the wind is potentially a significant transport mechanism at the IMR ranges, based on the presence of unvegetated areas and areas of sparse vegetation within certain areas of these ranges (e.g., impact areas and soil berms).

The transport of surface soil-associated contaminants by surface runoff is another potentially significant transport mechanism. Surface soil contaminants may be solubilized by rainwater and subsequently transported to drainage ditches, low-lying areas, and Remount Creek via surface runoff. The solubility of inorganics in rainwater is largely dependent upon the pH of the rainwater. Because the rainwater in this region is most likely slightly acidic, the inorganic

constituents in surface soil are likely to solubilize to some degree in the rainwater and be subject to transport via runoff. Most of the semivolatile compounds are strongly associated with soil particles and would not solubilize to a large extent. Contaminants that may be more strongly bound to particulate matter in surface soil (e.g., SVOCs and some of the inorganics) may be entrained in surface water runoff and transported to drainage ditches, low-lying areas, and Remount Creek via surface runoff. Many of the metals and semivolatiles are strongly sorbed to soil particles and could be transported from their source areas via this mechanism.

Contaminants in surface soil may be transported vertically to subsurface soils and groundwater via solubilization in rainwater and infiltration. Subsequent groundwater transport to surface water in Remount Creek could result in exposure of aquatic receptors to soil contaminants. Migration in this manner is dependent upon contaminant solubility and frequency of rainfall. Although the soil types in the vicinity of the IMR ranges (sand, stone, and gravel) are expected to promote relatively rapid infiltration of rainwater, the less soluble constituents (e.g., SVOCs) found at the IMR ranges are not likely to migrate to any great extent vertically, due to their relatively low solubilities. Inorganics in soil at the IMR ranges may migrate vertically due to the acidic nature of the rainwater in this area and the increased solubility of metals that it produces. However, surface water and groundwater monitoring data indicate that this transport mechanism is insignificant at the IMR ranges as only lead was detected in surface water at elevated concentrations and lead was not detected in any groundwater samples at elevated concentrations. Furthermore, other constituents detected in groundwater were not detected in surface water at the IMR ranges.

The transfer of contaminants in surface soil to terrestrial plants through root uptake and transfer 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. There are several exceptions to this, namely, arsenic and nickel, which may bioconcentrate and/or biomagnify (Agency for Toxic Substances and Disease Registry [ATSDR], 1989 and 1995). Many of the SVOCs have the potential to bioaccumulate in lower trophic level organisms (e.g., terrestrial invertebrates), but most higher trophic level animals have the ability to metabolize these compounds rapidly, precluding the potential for bioconcentration (Eisler, 1987).

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

## **4.2 Fate and Transport in Surface Water**

In general, contaminants present in the surface water associated with the IMR ranges (Remount Creek and tributaries) are the result of erosion and runoff from the ranges. Contaminants in surface water at the IMR ranges may be transported from their sources to other areas at the ranges or to off-site locations by the following mechanisms: 1) volatilization, 2) transfer to groundwater, 3) transfer to sediment, and 4) flow downstream. Volatile organic contaminants in surface water would be expected to rapidly volatilize from the water-air interface and be dispersed in the atmosphere. Therefore, transport of volatile constituents in surface water is not expected to occur for any significant distance.

Water in Remount Creek originates mainly from discharge from Yahou Lake and overland flow from the surrounding watershed. There also appears to be sporadic and localized contributions to creek flow from groundwater where the potentiometric surface exceeds the creek bed surface. The flow contribution in Remount Creek from groundwater varies according to the amount of precipitation, with an increase of groundwater contribution when precipitation raises the potentiometric surface.

Thus constituents in groundwater could migrate to surface water in Remount Creek and its tributaries. This transport mechanism appears to be relatively insignificant based on the fact that only lead has been detected in surface water at concentrations that are elevated with respect to ecological screening values. Other constituents detected in groundwater have not been detected in surface water at elevated concentrations. Additionally, elevated lead concentrations in surface water are restricted to Remount Creek and small tributaries at the Skeet Range. Contaminant transfer to sediments represents another significant transfer mechanism, especially where contaminants are in the form of suspended solids, or are hydrophobic substances (e.g., PAH) that can become adsorbed to organic matter in the sediments. The metals detected in surface water have the potential to associate with suspended particulate matter.

Contaminants in surface water can be transported to other ranges along Iron Mountain Road or off-site via Remount Creek. Transfer of contaminants in surface water to aquatic organisms is also a potentially significant transfer pathway. Some of the inorganic constituents detected in surface water may bioaccumulate in lower trophic level organisms. Most of the inorganics detected in surface water are not highly bioconcentratable; therefore, transfer through the food web is expected to be minimal for these compounds.

### **4.3 Fate and Transport in Sediment**

Contaminant transfer between sediment and surface water potentially represents a significant transfer mechanism, especially when contaminants are in the form of suspended solids. Sediment/surface water transfer is reversible; sediments often act as temporary repositories for contaminants and gradually release contaminants to surface waters. This is especially true in surface water systems that are acidic, as is the case with Remount Creek in the vicinity of the IMR ranges. Sorbed or settled contaminants can be transported with the sediment to downstream locations. Much of the substrate of Remount Creek and its tributaries in the vicinity of the IMR ranges is best characterized as gravel or cobbles. Very few areas of high organic content sediment or muck are present. The very low organic content of gravel and cobble create a substrate with very low binding capacity; therefore, constituents released to Remount Creek and its tributaries via surface runoff or other transport mechanisms would most likely remain suspended in the surface water, be transported downstream, and would not be sequestered in the stream substrate directly adjacent to the IMR ranges.

Directly downstream of the Skeet Range, Remount Creek passes the former location of Motor Pool 3100 and then flows through a concrete culvert beneath the former parade grounds and then it flows through the Cane Creek golf course before it's confluence with Cane Creek. The portion of Remount Creek directly downstream of the Skeet Range exhibits small reaches with characteristics of both high-energy (scouring) and low-energy (depositional) environments. Short reaches of Remount Creek directly downstream of the Skeet Range in the vicinity of Motor Pool 3100 exhibit a low gradient and a narrow and shallow channel, which are characteristic of a low energy stream environment. If sediment-associated contamination was being transported by Remount Creek downstream of the IMR ranges, it could be expected to be deposited in these low energy sections of Remount Creek in the vicinity of the former Motor Pool 3100. There are also sections of Remount Creek in the vicinity of Motor Pool 3100 that exhibit characteristics of a high energy environment. These creek reaches exhibit deep erosional channels and cobble and boulder substrate. Deposition of sediment-associated COPECs is not expected in these high energy portions of Remount Creek. All of the sediment samples downstream of the IMR ranges were collected from the low-energy depositional zones of Remount Creek as these areas were the only areas that sediment was present.

Although transfer of sediment-associated contaminants to bottom-dwelling biota also represents a potentially significant transfer mechanism, it is not expected to be a major mechanism at the IMR ranges. Lower trophic level organisms may accumulate metals and PAHs; however, higher trophic level organisms have the ability to metabolize PAHs and therefore reduce their

accumulative properties. Most of the inorganics detected in sediment are not bioaccumulative. Mercury and copper may bioaccumulate to some extent due to exposures to sediment.

#### **4.4 Constituent-Specific Fate and Transport Properties**

The following sub-sections describe the fate and transport properties of each of the COPECs identified at the IMR ranges.

##### **4.4.1 Antimony**

Antimony binds to soil, particularly to particles containing iron, manganese or aluminum. It is also oxidized by bacteria in the soil. In water, antimony is oxidized when exposed to atmospheric oxygen. Antimony is not significantly metabolized and is excreted in the urine and feces. It does not biomagnify in terrestrial food chains, but can bioconcentrate in aquatic organisms. Antimony may be taken up by plants, the rate of which is dependent upon the solubility of the antimony in the soil. It should also be noted that antimony is associated with ammunition, being present in lead alloys in bullets and in materials used as primers. Antimony can be present in both the 3+ and 5+ valence states, depending on pH, oxidation-reduction potential, and several other chemical properties of the environmental medium in which it is found. Antimony can methylate via chemical and/or biological reactions into an organic form under reducing conditions such as those commonly found within highly organic fine sediments.

##### **4.4.2 Arsenic**

Most arsenic in the environment exists in soil or rock. Because many arsenic compounds tend to adsorb to soils or sediments, leaching usually results in transportation over only short distances in soil (Moore, et al., 1988).

Transport and partitioning of arsenic in water depends upon the chemical form (oxidation state and counter ion) of the arsenic and on interactions with other materials present. Soluble forms move with the water, and may be carried long distances by rivers and streams (Callahan, et al., 1979). However, arsenic may be adsorbed from water onto sediment or soil, especially clays, iron oxides, aluminum hydroxides, manganese compounds, and organic material (Callahan, et al., 1979). Sediment-bound arsenic may be released back into the water by chemical or biological interconversions of arsenic species.

Bioconcentration of arsenic occurs in aquatic organisms, primarily in algae and lower invertebrates. Bioconcentration factors measured in freshwater invertebrates and fish for several arsenic compounds ranged from 0 to 17 (USEPA, 1980). Biomagnification in aquatic food chains does not appear to be significant (Callahan, et al., 1979). Terrestrial plants may

accumulate arsenic by root uptake from the soil or by absorption of airborne arsenic deposited on the leaves (USEPA, 1982).

Arsenic in water can undergo a complex series of transformations, including oxidation-reduction reactions, ligand exchange, and biotransformation (Callahan, et al., 1979). The factors most strongly influencing the fate processes in water include the oxidation-reduction potential, pH, metal sulfide and sulfide ion concentrations, iron concentrations, temperature, salinity, and distribution and composition of the biota (Callahan, et al., 1979). The predominant form of arsenic in surface water is usually arsenate (USEPA, 1982), but aquatic microorganisms may reduce the arsenate to arsenite and a variety of methylated arsenicals (Benson, 1989). Arsenate also predominates in groundwater, but arsenite may be an important component, depending upon the characteristics of the water and surrounding geology (Robertson, 1989).

Transformations of arsenic in soil are similar to those occurring in aquatic systems, with As (+5) predominating in aerobic soils, As (+3) in slightly reduced soils (e.g., temporarily flooded), and arsine methylated arsenic, and elemental arsenic in very reduced conditions (e.g., swamps and bogs). Organoarsenical pesticides (e.g., MMA, DMA) applied to soil are metabolized by soil bacteria to alkylarsines, arsenate, and MMA (Hood, 1985). The half-life of DMA in soil is about 20 days (Hood, 1985).

#### **4.4.3 Barium**

In aquatic media, barium is likely to precipitate out of solution as an insoluble salt (i.e., as  $\text{BaSO}_4$  or  $\text{BaCO}_3$ ). Waterborne barium may also adsorb to suspended particulate matter (Bodek, et al., 1988). Sedimentation of suspended solids removes a large portion of the barium content from surface waters (Benes, et al., 1983). Barium in sediments is found largely in the form of barium sulfate (barite). The uptake of barium by fish is also an important removal mechanism (Schroeder, 1970).

Barium is not very mobile in most soil systems. The rate of transportation of barium in soil is dependent on the characteristics of the soil material. Soil properties that influence the transportation of barium to groundwater are cation exchange capacity and calcium carbonate content. In soil with a high cation exchange capacity (e.g., fine textured mineral soils or soils with high organic matter content), barium mobility will be limited by adsorption (Kabata-Pendias and Pendias, 1984). High  $\text{CaCO}_3$  content limits mobility by precipitation of the element as  $\text{BaCO}_3$  (Lagas, et al., 1984). Barium will also precipitate as barium sulfate in the presence of sulfate ions (Lagas, et al., 1984). Humic and fulvic acids have not been found to increase the mobility of barium (USEPA, 1984).

Under natural conditions barium will form compounds in the +2 oxidation state. Barium does not hydrolyze appreciably except in highly alkaline environments (i.e., at pH levels greater than or equal to 10) (Bodek, et al., 1984). Appreciable levels of barium sulfate occur because natural water often contains high sulfate concentrations. Since the solubility of barium sulfate is low, only trace amounts of barium dissolve in surface water (Bodek, et al., 1988). Barium forms salts of low solubility with arsenate, chromate, fluoride, oxalate, and phosphate ions (Bodek, et al., 1988). The chloride, hydroxide, and nitrate of barium are water soluble and are frequently detected in aqueous environments (Rai, et al., 1984).

Barium reacts with metal oxides and hydroxides in soil and is subsequently adsorbed onto soil particulates (Rai, et al., 1984). Adsorption onto metal oxides in soils and sediments probably acts as a control over the concentration of barium in natural waters (Bodek, et al., 1988). Barium is also adsorbed onto soil through electrostatic interactions. The cation exchange capacity of the sorbent largely controls the retention of barium in soils. Barium is strongly adsorbed by clay minerals (Kabata-Pendias and Pendias, 1984).

Barium can also form salts with acetate, nitrate, chloride, and hydroxide ions in soil. The mobility of barium in soils increases upon formation of these water soluble salts (Bodek, et al., 1988). In general, the solubility of barium compounds increases with decreasing pH.

#### **4.4.4 Copper**

Copper's movement in soil is determined by a host of physical and chemical interactions with the soil components. In general, copper will adsorb to organic matter, carbonate minerals, clay minerals, or hydrous iron and manganese oxides. Sandy soils with low pHs have the greatest potential for leaching. When the amount of organic matter is low, the mineral content of Fe, Mn, and Al oxides become important in determining the adsorption of copper. Copper binds to soil much more strongly than other divalent cations, and the distribution of copper in the soil solution is less affected by pH than other metals (ATSDR, 1990).

Copper binds primarily to organic matter in sediment, unless the sediment is organically poor. It also binds to iron oxides.

The bioconcentration factor (BCF) of copper in fish obtained in field studies ranges from 10 to 100, indicating a low potential for bioconcentration. The BCF is higher in mollusks, where it may reach 30,000. This may be due to the fact that many mollusks are filter feeders, and copper concentrations are higher in particulates than in water. There is abundant evidence, however,

that there is no biomagnification of copper in the food chain. No evidence of bioaccumulation in herbivorous, omnivorous, and carnivorous mammals was obtained during a study of 10 mammal species in Donana National Park in Spain. A study of metals in cottontail rabbits showed that while the concentration of copper in surface soil was 130 percent higher than in control areas, the concentration of copper in foliar samples was insignificant. No significant increase in copper was observed in rabbit muscle, femur, kidney, or liver, indicating that copper was not bioaccumulating in the food chain. Even at the lowest levels of the food chain, there is little evidence of copper bioaccumulation. In a study of earthworms and soil from 20 different sites, copper concentrations in earthworms poorly correlated with copper in soil (ATSDR, 1990).

At the pH values and carbonate concentrations characteristic of natural waters, most dissolved copper exists as carbonate complexes rather than as free (hydrated) cupric ions. The concentration of dissolved copper depends on factors such as pH, oxidation-reduction potential, and the presence of competing cations ( $\text{Ca}^{2+}$ ,  $\text{Fe}^{2+}$ ,  $\text{Mg}^{2+}$ , etc.), anions of insoluble cupric salts ( $\text{OH}^-$ ,  $\text{S}^{2-}$ ,  $\text{PO}_4^{3-}$ , etc.), and organic and inorganic complexing agents. Allard (1995) reported that copper can exist in the form of freely-dissolved divalent copper cation at a pH of less than 6. Complexation of copper with humic acids can increase the mobility of copper in groundwater and/or surface water but will also reduce the bioavailability to biota. The most significant precipitate formed in natural waters is malachite [ $\text{Cu}^2(\text{OH})^2\text{CO}_3$ ]. The combined processes of complexation, adsorption, and precipitation control the level of free copper. The chemical conditions in most natural waters are such that, even at relatively large copper concentrations, these processes will reduce the free copper concentration to extremely low values (ATSDR, 1990).

Between pH 5 and 6, adsorption is the principal process for removing copper from water; above pH 6, precipitation becomes more dominant. Copper binding in soil is correlated with pH, cation exchange capacity, organic content of the soil, and presence of iron oxides. Copper may also be incorporated into mineral lattices where it is unlikely to have ecological significance. In soils with high organic carbon content, copper will be tightly bound to organic matter (ATSDR, 1990).

#### **4.4.5 Lead**

The chemistry of lead in aqueous solution is highly complex because this element can be found in a multiplicity of forms. The form of lead at any given site is very important since its bioavailability and uptake dynamics are generally dictated by its form. For example, lead fumes, as from a smelter or gasses generated from the discharge of artillery or bullets, are more bioavailable than mining wastes or intact pieces of lead fragments. The difference is therefore

not only the size of the particles, but it's chemical form. It should also be noted that lead in soil can slowly undergo speciation to more insoluble sulfate, sulfide, oxide, and phosphate salts (National Library of Medicine, 1996). Lead has a tendency to form compounds of low solubility with the major anions of natural water. In the natural environment, the divalent form is the stable ionic species of lead. Hydroxide, carbonate, sulfide and sulfate may act as solubility controls in precipitating lead from water. The amount of lead that remains in solution depends upon the pH of the water and the dissolved salt content. Lead is more soluble in softer water and low pH water (ATSDR, 1988). Complexation of lead with humic acids can increase the mobility of lead in groundwater and/or surface water but will also reduce the bioavailability to biota.

A significant fraction of lead carried by surface water is expected to be in an un-dissolved form, which can consist of colloidal particles or lead compounds incorporated in other components of surface particulate matter from runoff. Lead may occur as sorbed ions or surface coatings on sediment mineral particles, or it may be carried as a part of suspended living or nonliving organic matter in water. The ratio of lead in suspended solids to lead in dissolved form ranges from 4:1 to 27:1 (ATSDR, 1988).

Most lead in soil is retained there and very little is transported into surface water or groundwater (ATSDR, 1988). Low alkalinity and low pH conditions in soils can enhance the potential for bioconcentration of lead in mammals, birds, mosses, lichens, lower trophic level animals, and plants (Jenkins, 1981).

Most lead does not appear to significantly bioaccumulate in most fish. However, bioaccumulation of tetraethyl lead can occur in aquatic organisms (ATSDR, 1988). Plants commonly take up lead from soil and, therefore, may return it upon decomposition. Because the bioavailability of lead is dependent upon site-specific conditions, the accuracy of the ecological assessment of lead depends heavily on site-specific tests of bioavailability and subsequent toxicity and accumulation.

#### **4.4.6 Manganese**

The transport and partitioning of manganese in water is controlled by the solubility of the specific chemical form present, which in turn is determined by Eh, pH, and the characteristics of the available anions. The metal may exist in water in any of four oxidation states (2+, 3+, 4+, or 7+). Divalent manganese (Mn<sup>+2</sup>) predominates in most waters (pH 4 – 7), but may become oxidized at pH greater than 8 or 9 (EPA, 1984). The principal anion associated with Mn (+2) in water is usually carbonate (CO<sub>3</sub><sup>-2</sup>), and the concentration of manganese is limited by the relatively low solubility of Mn CO<sub>2</sub> (Schaanning, et al., 1988). In relatively oxidized water, the

solubility of Mn (+2) may be controlled by manganese oxide equilibria, with manganese being converted to the (+3) or (+4) valence states. In extremely reduced water, the fate of manganese tends to be controlled by formation of the poorly soluble sulfide (USEPA, 1984c).

Manganese is often transported in rivers and streams as suspended sediments. Manganese in water may be significantly bioconcentrated at lower trophic levels. In general, the data indicate that lower trophic level organisms such as algae have larger BCFs than higher trophic level organisms. Thus, biomagnification of manganese in the food chain does not appear to be significant (USEPA, 1984c).

The tendency of soluble manganese compounds to adsorb to soils and sediments depends mainly on the cation exchange capacity and the organic composition of the soil (Curtin, et al., 1980). At low concentrations, manganese may be "fixed" by clays, and will not be released into solution readily. At higher concentrations, manganese may be desorbed by ion exchange mechanisms with other ions in solution (Rai, et al., 1986).

Manganese in water may undergo oxidation at high pH or Eh, and is also subject to microbial activity. Likewise, the oxidation state of manganese in soils and sediments may be altered by microbial activity.

#### **4.4.7 Thallium**

Thallium exists in water primarily as a monovalent ion (thallium<sup>+</sup>); thallium may be trivalent (Tl<sup>3+</sup>) in very oxidizing water (Callahan, et al., 1979). Thallium may precipitate from water as solid mineral phases. However, thallium chloride, sulfate, carbonate, bromide, and hydroxide are very soluble in water. In extremely reducing water, thallium may precipitate as a sulfide (Tl<sub>2</sub>S), and in oxidizing water, Tl<sup>3+</sup> may be removed from solution by the formation of Tl(OH)<sub>3</sub> (Lee, 1971).

Thallium may be bioconcentrated by organisms from water. The experimentally-derived BCF value for thallium in bluegill sunfish was reported to be 34 (Barrows, et al., 1978). Thallium is absorbed by plants from soil and thereby enters the terrestrial food chain. Thallium can be absorbed by roots of higher plants from the rhizosphere (Cataldo and Wildung, 1983). There is no evidence to suggest that thallium is biotransformed in the environment.

#### **4.4.8 Zinc**

Zinc occurs in the environment mainly in the +2 oxidation state. Sorption is the dominant reaction, resulting in the enrichment of zinc in suspended and bed sediments. Zinc in aerobic

waters is partitioned into sediment through sorption onto hydrous iron and manganese oxides, clay minerals, and organic material. The efficiency of these materials in removing zinc from solution varies according to their concentrations, pH, redox potential, nature and concentration of complexing ligands, cation exchange capacity, and the concentration of zinc (ATSDR, 1994). Similar to copper, zinc is complexed at high pHs and can exist as freely-dissolved divalent cations at lower pHs, thus enhancing its bioavailability.

Zinc is an essential nutrient that is present in all organisms. Although biota appears to be a minor reservoir of zinc relative to soils and sediments, microbial decomposition of biota in water can produce ligands, such as humic acids, that can affect the mobility of zinc in the aquatic environment through zinc precipitation and adsorption (ATSDR, 1994). Complexation with humic acids will increase the mobility of zinc in aquatic systems; however, the humic acid-zinc complexes tend to have low bioavailabilities. Zinc can accumulate in freshwater animals at 51 to 130 times the concentration present in water. In general, zinc does not biomagnify through food chains. Furthermore, although zinc bioaccumulates to some degree in aquatic systems, biota appears to represent a relatively minor sink compared to sediments. Steady-state zinc BCF for 12 aquatic species range from 4 to 24,000, with most being less than 100. With respect to bioconcentration from soil by terrestrial plants, invertebrates, and mammals, BCFs of 0.4, 8, and 0.6, have been reported, respectively. In general, plants do not concentrate zinc above levels present in the soil (ATSDR, 1994).

## 5.0 Ecosystems Potentially at Risk

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The terrestrial habitat occurring at the Iron Mountain Road ranges falls into two general categories: “cleared” areas and forested areas. It is important to note that the ecosystem richness and overall quality of the open or “cleared” areas is significantly less than the relatively unaltered forested areas. It is also important to note that the areal extent of contamination is centered within the “cleared” range areas where small arms firing took place. The “cleared” areas are those areas that were formerly maintained as lawns or mowed fields. Since maintenance activities have ceased in these areas, pioneer species are now colonizing these ranges. Typically, the species most likely to colonize these areas are the “weed” species that tend to be vigorous pioneer plants that grow and spread rapidly. The first of the pioneer species to invade these abandoned areas are the grasses and herbaceous species. These formerly maintained grassy areas are classified as being in an early old field successional state. Over time, these grass and herbaceous species will be followed by shrubs and small trees. The early old field successional areas at the IMR ranges are dominated by various grasses and herbs including *Rumex spp.* (dock), *Trifolium spp.* (clover), *Astragalus spp.* (vetch), *Asclepias spp.* (milkweed), *Galium spp.* (bed straw), *Chrysanthemum leucanthemum* (ox-eye daisy), and *Sorghum halepense* (johnson grass). Other old field herbaceous species occurring at the IMR ranges are *Rubus occidentalis* (black raspberry), *Toxicodendron radicans* (poison ivy), *Rhus glabra* (smooth sumac), *Smilax rotundiflora* (green brier), *Lonicera japonica* (Japanese honeysuckle), *Vitis labrusca* (fox grape), and *Rosa multiflora* (multiflora rose).

The forested areas outside of the “cleared” areas are best characterized as mixed deciduous/coniferous forest. With the exception of the forest stand around the skeet range, these rich and relatively unaltered forested regions represent the large safety fans across the main post. The cover species typically found in the forested areas surrounding the IMR ranges include *Pinus virginiana* (scrub pine), *Pinus taeda* (loblolly pine), *Quercus alba* (white oak), *Quercus stellata* (post oak), *Quercus prinus* (chestnut oak), *Quercus falcata* (southern red oak), *Prunus serotina* (wild black cherry), *Celtis occidentalis* (hackberry), *Juglans nigra* (black walnut), and *Cornus florida* (flowering dogwood). These mixed deciduous/coniferous forests exhibit sparse, shade-tolerant undergrowth species such as *Parthenocissus quinquefolia* (Virginia creeper), *Polystichum acrotichoides* (Christmas fern), and *Toxicodendron radicans* (poison ivy). Descriptions of the habitats at each of the small arms ranges at Iron Mountain Road are presented in the following sections.

## **5.1 Skeet Range Habitat**

The Skeet Range is comprised of two main habitat types. The “cleared” area encompasses approximately 2 acres at and adjacent to the firing lines. The habitat within this area is dominated by grasses and early old field successional species. In the past, this area was maintained lawn with concrete walkways throughout. Since maintenance activities have ceased, the grasses have grown uncontrolled and early successional species have intruded. Various grasses and herbaceous species dominate this habitat type. Scrub pine (*Pinus virginiana*) saplings have also begun to encroach on this cleared area. The forested areas surrounding the “cleared” area at the Skeet Range can be characterized as mixed deciduous/coniferous forest. Scrub pine and southern red oak dominate this habitat. There are minimal understory or herbaceous layers in this forest type as fallen leaves and pine needles form a thick mat that precludes the germination of smaller plants. White-tailed deer, wild turkey, gray squirrel, and various songbirds were observed on-site. Details regarding site history, physical characteristics, and other background information are presented in the *Engineering Evaluation and Cost Analysis for the Small Arms Ranges at Iron Mountain Road* (IT, 2001).

Remount Creek, which runs along the western boundary of the Skeet Range, and its tributaries, which generally run east-to-west across the Skeet Range, exhibit mostly gravel and cobble substrate with very little organic matter. Remount Creek is narrow (less than 3 feet across) and shallow (less than 6 inches deep) when there is water present. This ephemeral creek and its tributaries are dry during significant portions of the year, and the presence or absence of water is highly dependent upon the volume of precipitation. During dry periods, the creek may be entirely dry for significant portions of the year (6 to 8 months). The ephemeral nature of Remount Creek and its tributaries in this area limit their ability to support many aquatic organisms (e.g., large fish) and other organisms that rely on aquatic species for food (e.g., piscivores). Remount Creek and its tributaries may support semi-aquatic species (e.g., amphibians) and provide a breeding ground for some small fish species during the periods when water is present.

Site walks conducted on May 10, 2002 revealed potential impacts to the habitat of Remount Creek and the surrounding creek corridor from activities associated with construction of the “Eastern Bypass”. Moreover, it is possible that the bypass construction will require significant alterations to Remount Creek in the vicinity of the IMR ranges.

Remount Creek in the vicinity of the Skeet Range has historically been identified as a moderate to low quality foraging area for the Federally endangered gray bat (*Myotis grisescens*) (Garland, 1996). This section of Remount Creek has been identified as a gray bat foraging area because it

allegedly provides habitat for aquatic insects, which are fed upon by the gray bat. However, Remount Creek is dry during significant portions of the year, which precludes the presence of aquatic insects during those dry periods. Additionally, construction of the Eastern Bypass directly adjacent to the Remount Creek corridor has eliminated a significant portion of the tree and shrub canopy that formerly covered Remount Creek. Because the gray bat requires continuous cover while traveling to and from its foraging habitats and while foraging, the elimination of significant portions of the forest in this area would negatively affect their foraging habits. Based on its ephemeral nature in this area and the elimination of significant portions of the forest canopy, Remount Creek may no longer provide adequate foraging habitat for the gray bat.

## **5.2 Range 19 Habitat**

The total site, including the extensive forested range fan, encompasses 1,529 acres. The main study area is limited to approximately 5 to 7 acres. Details regarding site history, physical characteristics, and other background information are presented in the *Engineering Evaluation and Cost Analysis for the Small Arms Ranges at Iron Mountain Road* (IT, 2001). The study area of Range 19 is almost entirely comprised of maintained lawn, mowed fields, unvegetated soil, and roadways. Since maintenance activities have ceased, the grasses have grown uncontrolled and early successional species have intruded. Various grasses and herbaceous species dominate this habitat type. Scrub pine saplings (*Pinus virginiana*) have also begun to encroach into these previously maintained areas. The embankment on the eastern side of the site is almost completely void of vegetation, which is due to the fact that soil was historically scraped and graded along this embankment for maintenance purposes when the ranges were active. The area surrounding the cleared areas of Range 19 can be characterized as mixed deciduous/coniferous forest. Scrub pine and southern red oak dominate this habitat. There are minimal understory or herbaceous layers in this forest type as fallen leaves and pine needles form a thick mat precluding the germination of smaller plants. White-tailed deer, wild turkey, gray squirrel, and various songbirds were observed on-site.

Remount Creek is dry for significant periods of time over the vast majority of the distance that it passes along the western boundary of Range 19 (approximately 1,000 feet). The presence of small pools of water (one to two feet in length) in this portion of Remount Creek is most likely due to groundwater discharge. The creek is narrow (less than 3 feet across) and shallow (less than 6 inches deep), when water is present, and has a variable substrate of mud and leaf litter interspersed with areas of sand and gravel. Again, the presence of water in Remount Creek adjacent to Range 19 is highly dependent upon significant precipitation.

The portion of Remount Creek adjacent to Range 19, has historically been classified as low quality foraging habitat for the Federally endangered gray bat (*Myotis grisescens*) (Garland, 1996). This classification has not taken into account the impacts on Remount Creek habitat resulting from construction of the "Eastern Bypass".

### **5.3 Range 13 Habitat**

The total area of Range 13, including the range safety fan, encompasses 549 acres. The main study area is limited to approximately 5 acres, which are described herein as the "cleared" areas. The "cleared" area of Range 13 is dominated by grasses and early successional species. In the past, this area consisted of maintained lawn, mowed field, unvegetated soil, and roadways. Since maintenance activities have ceased, the grasses have grown uncontrolled and early successional species have intruded. Various grasses and herbaceous species dominate this habitat type. Scrub pine saplings (*Pinus virginiana*) have also begun to encroach into these previously maintained areas. The forested areas surrounding the "cleared" area at Range 13 can be characterized as mixed deciduous/coniferous forest. Scrub pine and southern red oak dominate this habitat. There are minimal understory or herbaceous layers in this forest type as fallen leaves and pine needles form a thick mat that precludes the germination of smaller plants. White-tailed deer, wild turkey, gray squirrel, and various songbirds were observed on-site. Details regarding site history, physical characteristics, and other background information are presented in the *Engineering Evaluation and Cost Analysis for the Small Arms Ranges at Iron Mountain Road* (IT, 2001).

Remount Creek, along the 800 foot length that runs adjacent to the western boundary of Range 13, exhibits sections of very slow moving water (zero to 6 inches deep) and areas which are completely dry. Similar to Range 19, the presence of small, intermittent areas of water (less than one foot to several feet in length) is most likely due to groundwater discharge. The creek is narrow (less than 3 feet across) and shallow (less than 6 inches deep), when water is present, and has a variable substrate of mud and leaf litter interspersed with areas of sand and gravel. Again, the presence of water in Remount Creek adjacent to Range 13 is highly dependent upon significant precipitation.

The portion of Remount Creek adjacent to Range 13, has historically been classified as low quality foraging habitat for the Federally endangered gray bat (*Myotis grisescens*) (Garland, 1996). This classification has not taken into account the impacts on Remount Creek habitat resulting from construction of the "Eastern Bypass".

#### **5.4 Range 12 Habitat**

The total area of Range 12, including the range safety fan, encompasses 311 acres. The main study area is limited to approximately 5 acres, which are described herein as the “cleared” areas. The “cleared” area of Range 12 is dominated by grasses and early successional species. In the past, this area consisted of maintained lawn, mowed field, and roadways. Since maintenance activities have ceased, the grasses have grown uncontrolled and early successional species have intruded. Various grasses and herbaceous species dominate this habitat type. Scrub pine saplings (*Pinus virginiana*) have also begun to encroach into these previously maintained areas. The forested areas surrounding the “cleared” area at Range 12 can be characterized as mixed deciduous/ coniferous forest. Scrub pine and southern red oak dominate this habitat. There are minimal understory or herbaceous layers in this forest type as fallen leaves and pine needles form a thick mat that precludes the germination of smaller plants. White-tailed deer, wild turkey, gray squirrel, and various songbirds were observed on-site. Details regarding site history, physical characteristics, and other background information are presented in the *Engineering Evaluation and Cost Analysis for the Small Arms Ranges at Iron Mountain Road* (IT, 2001).

Along the 400-foot length that runs adjacent to the western boundary of Range 12, Remount Creek exhibits sections of very slow moving water (zero to 6 inches deep) and areas which are completely dry. Similar to Range 19, the presence of small, intermittent areas of water (less than one foot to several feet in length) is most likely due to groundwater discharge. The creek is narrow (less than 3 feet across) and shallow (less than 6 inches deep), when water is present, and has a variable substrate of mud and leaf litter interspersed with areas of sand and gravel. Again, the presence of water in Remount Creek adjacent to Range 12 is highly dependent upon significant precipitation.

The portion of Remount Creek adjacent to Range 12, has historically been classified as low quality foraging habitat for the Federally endangered gray bat (*Myotis grisescens*) (Garland, 1996). This classification has not taken into account the impacts on Remount Creek habitat resulting from construction of the “Eastern Bypass”.

#### **5.5 Remount Creek Habitat**

Again, Remount Creek in the vicinity of the IMR ranges is a small, ephemeral stream that flows (when water is present) from south to north. The physical characteristics of Remount Creek and the surrounding land-use vary along its length, from its headwaters at Yahou Lake to its confluence with Cane Creek near the west-northwest boundary of the Main Post. The headwaters of Remount Creek are formed by the discharge from Yahou Lake and its tributaries, which is approximately 0.75 miles south of Range 12. Remount Creek runs in a northerly

direction along the topographic low formed by gently sloping hills to the east and west of the creek. The vast majority of the length of Remount Creek between Yahu Lake and the IMR ranges runs through the "Eastern Bypass" corridor. Again, virtually all of the trees in the by-pass corridor have been clear-cut and all of the vegetation removed. The entire area has been covered with mulch that was created by "chipping" the vegetation that was cut down. The land surrounding Remount Creek adjacent to Ranges 12 and 13 is characteristic of the clear-cut areas associated with the "Eastern Bypass" corridor. It is likely that portions of the creek adjacent to the IMR ranges will be significantly altered (i.e., re-routed, culverted) as a result of construction of the Eastern Bypass.

Immediately north of the Skeet Range, Remount Creek flows through a culvert under the old parade grounds/athletic fields and then through the grounds of the Cane Creek Golf Course until its confluence with Cane Creek in the west-northwestern corner of the Main Post.

The ecological value of Remount Creek is greatest as it flows through the Cane Creek Golf Course and intersects Cane Creek. It is within this stretch (downstream of the IMR ranges) that the creek will support foraging of insectivorous mammals and a functional aquatic ecosystem. Remount Creek and its tributaries in the vicinity of the IMR ranges may support semi-aquatic species (e.g., amphibians) and provide a breeding ground for some small fish species during the periods when water is present.