Appendix D

Ecotoxicity Profiles

INTRODUCTION

Ecotoxicity profiles are provided for all chemicals of potential concern identified in Section 7 of this work plan. The profiles presented in this appendix describe the potential toxic effects associated with exposure of terrestrial and aquatic biota to these chemicals.

METALS

Cadmium

There is no evidence that cadmium, a relatively rare metal, is biologically essential or beneficial; on the contrary, it has been implicated as a cause of human mortality and various deleterious effects in fish and wildlife. In sufficient concentration, it is toxic to all forms of life, including microorganisms, higher plants, and animals (Eisler 1985). Freshwater biota are the most sensitive organisms to cadmium exposure, with toxicity inversely proportional to water hardness.

It is now conservatively estimated that adverse effects on fish or wildlife are either pronounced or probable when cadmium concentrations exceed 3 parts per billion (ppb) in fresh water or 100 ppb in the diet (Eisler 1985). Cadmium residues in vertebrate kidney or liver that exceed 10 parts per million (ppm) fresh weight or 2 ppm whole body fresh weight should be viewed as evidence of probable cadmium contamination. Residues of 200 ppm fresh weight kidney, or more than 5 ppm whole animal fresh weight, are probably life-threatening to the organism.

Exposure routes for aquatic organisms include ingestion and gill uptake. Concentrations of 0.8 to 9.9 μg Cd/L (ppb) in water were lethal to several species of aquatic insects, crustaceans, and teleosts; and concentrations of 0.7 to 570 ppb were associated with sublethal effects such as decreased growth, inhibited reproduction, and population alterations (Eisler 1985). These effects were most pronounced in waters of comparatively low alkalinity.

Mammals and birds are comparatively resistant to the biocidal properties of cadmium. The lowest oral doses producing death in rats and guinea pigs range from 150 to 250 mg Cd/kg body weight (ppm). Although mallards and chickens tolerate 200 ppm of cadmium in diets for protracted periods, kidney cadmium exceeded 130 ppm fresh weight under this regimen, a concentration considered life-threatening to some organisms. Sublethal effects of cadmium in birds, which were similar to those in other

animals, included growth retardation, anemia, and testicular damage; however, these effects are first observed in birds at higher concentrations than in aquatic biota.

Cadmium bioconcentrates in aquatic organisms, primarily in the liver and kidney (EPA 1999). Cadmium accumulated from water is slowly excreted, while cadmium accumulated from food is eliminated more rapidly. Metal-binding, proteinaceous, metallothioneins appear to protect vertebrates from deleterious effects of high metal body burdens (Eisler 1985).

Exposure routes in ecological mammalian species include ingestion and inhalation, while dermal absorption is negligible (EPA 1999). Absorption and retention of cadmium decrease with prolonged exposure. Cadmium absorption through ingestion is inversely proportional to intake of other metals, especially iron and calcium. Cadmium accumulates primarily in the liver and kidneys. Cadmium crosses the placental barrier. Cadmium does not undergo direct metabolic conversion, but the ionic (+2 valence) form binds to proteins and other molecules (EPA 1999). Absorbed cadmium is excreted very slowly, with urinary and fecal excretion being approximately equal. Freshwater aquatic species are most sensitive to the toxic effects of cadmium, followed by marine organisms, birds, and mammals.

Chromium

Chromium is not an essential element in plants (Kabata-Pendias and Pendias 1984). Although most soils contain significant amounts of chromium, its availability to plants is highly limited. The (VI) form is more soluble and available to plants than the (III) form and is considered the more toxic form (Efroymson et al. 1997). In soils within a normal Eh and pH range, chromium (VI), a strong oxidant, is likely to be reduced to the less available chromium (III) form although the (III) form may be oxidized to the (VI) form in the presence of oxidized manganese. In nutrient solution, however, both forms are about equally taken up by plants and toxic to plants. Chromium (VI), as CrO_4^{-2} , may share a root membrane carrier with SO_4^{-2}. Chromium (VI) is more mobile in plants than chromium (III), but translocation varies with plant type. After plant uptake, it generally remains in the roots because of the many binding sites in the cell wall capable of binding especially the chromium (III) ions. Within the plant, chromium (VI) may be reduced to the chromium (III) form and complexed as an anion with organic molecules. Chromium may interfere with carbon, nitrogen, phosphorus, iron, and molybdenum metabolism, and enzyme reactions (Kabata-Pendias and Pendias 1984). Symptoms of toxicity include stunted growth, poorly developed roots, and leaf curling. Phytotoxic effects of chromium have been reported at soil concentrations as low as 1.8 ppm (Efroymson et al 1997a).

Beyer and Cromartie (1987) showed that chromium has a moderate potential to bioaccumulate in earthworms (BAFs ranged from 0.1 to 5.4 depending on soil conditions).

Soil invertebrates have shown adverse effects at soil concentrations as low as 2 ppm (Efroymson et al. 1997b).

Chromium exists primarily in the Cr^{+3} and Cr^{+6} valence forms in environmental and biological media (EPA 1999). The +3 or (III) form functions as an essential element in mammals by maintaining efficient glucose, lipid, and protein metabolism. Chromium appears to play an important role in the maintenance of vascular integrity. A deficiency of this metal in wildlife results in elevated serum cholesterol levels and increased atherosclerotic aortic plaques (EPA 1999). Exposure routes for ecological mammalian species include ingestion, inhalation, and dermal absorption. Following oral exposure chromium is poorly absorbed from the gastrointestinal tract; however, fasting increases the absorption. Absorbed chromium is distributed to various organs including the liver and spleen. Following inhalation exposure, chromium is distributed to the lung, kidney, spleen, and erythrocytes (EPA 1999). Following dermal exposure, chromium is readily absorbed and distributed to the blood, spleen, bone marrow, lymph glands, urine, and kidneys. Chromium is not truly metabolized, but undergoes various changes in valence states and binding with ligands and reducing agents in vivo. Elimination of chromium is slow (EPA 1999). A large degree of accumulation by aquatic and terrestrial plants and animals in the lower trophic levels has been documented; however, the mechanism of this accumulation remains unknown.

The U.S. EPA regards all chromium compounds as toxic, although the most toxic and carcinogenic chromium compounds tend to be the strong oxidizing agents with an oxidation state of +6 (Irwin et al. 1997). The biological effects of chromium depend on chemical form, solubility, and valence. In mammals and birds hexavalent chromium causes cellular damage (mutagenic, carcinogenic, and teratogenic) via its role as a strong oxidizing agent, whereas trivalent chromium can inhibit various enzyme systems or react with organic molecules (Irwin et al. 1997; Eisler 1986a). In mammalian species, chromium is considered one of the least toxic trace elements, as normal stomach pH converts hexavalent chromium to trivalent chromium (Irwin et al. 1997). Adverse effects from chromium were predicted to occur in mammals at oral doses ranging from 22 to 120 mg/kg/day depending on body size and species based on a study of chromium (VI) toxicity to rats (Sample et al. 1996). For avian species, adverse effects were predicted to occur at an oral dose of 5.0 mg/kg, based on a study of $CrK(SO_4)^2$ to black ducks (Sample et al. 1996).

Records of acute toxicity of hexavalent and trivalent chromium salts to representative species of aquatic life make it clear that Cr^{+6} is the more toxic to freshwater biota in comparatively soft and acidic waters, that younger life stages are more sensitive than older organisms, and that 96 h is insufficient to attain stable mortality patterns (Eisler 1986a). There are at least five ionic species of hexavalent Cr, of which two—the hydrochromate ion and the chromate ion—are the predominant species and probably the agents that are toxic to freshwater life. However, water pH dramatically affects the

concentration of each. For hexavalent chromium, LC50 (96 h) values for sensitive freshwater and marine species were between 445 and 2,000 ppb. For trivalent chromium, LC50 (96 h) concentrations were 2,000 to 3,200 ppb for sensitive freshwater organisms and 3,300 to 7,500 ppb for marine biota (Eisler 1986a). Among sensitive species of freshwater fish, Cr^{+6} concentrations of 16 to 21 ppb resulted in reduced growth of rainbow trout and chinook salmon fingerlings during exposure of 14 to 16 weeks, and altered plasma cortisol metabolism in rainbow trout after 7 days. Further, locomotor activity in bluegills increased after 2 weeks in 50 ppb Cr^{+6}. Chromium uptakes and effects in fish were modified significantly by many biological and abiotic variables, including water temperature and pH, the presence of other contaminants or compounds, and sex and tissue specificity.

Copper

Copper is toxic in aquatic environments and affects fish, invertebrates, and amphibians (Eisler 1998). Toxic effects in birds include reduced growth rates, lowered egg production, and developmental abnormalities. Toxicity in mammals includes a wide range of animals and effects such as liver cirrhosis, necrosis in kidneys and the brain, gastrointestinal distress, lesions, and low blood pressure.

Copper is a micronutrient essential for plant nutrition (Efroymson et al. 1997a). It is required as a co-factor for many enzymes and is an essential part of a copper protein involved in photosynthesis. Copper occurs as part of enzymes and enzyme systems. Root absorption appears to be passive, perhaps in organo-copper complexes, and active through a specific carrier. Roots have a strong capability to hold copper in a complexed form (Kabata-Pendias and Pendias 1984). The form in which it is taken into the root affects its binding there. Copper can be transported in the xylem and phloem of plants complexed with amino acids (Efroymson et al. 1997a). The basic deleterious effect of copper is related to the root system where it interferes with enzyme functioning. It also strongly interferes with photosynthesis and fatty acid synthesis. The most common toxicity symptoms include reduced growth, poorly developed root system, and leaf chlorosis. Phytotoxicity appears at soil copper concentrations as low as 100 ppm (Kabata-Pendias and Pendias 1984; Efroymson et al. 1997a).

Beyer and Cromartie (1987) and Kabata-Pendias and Pendias (1985) showed that copper has a moderate potential to bioaccumulate in earthworms (BAFs ranging from 0.01 to 2.5 depending upon soil conditions). Soil invertebrates show adverse effects at copper concentrations as low as 60 ppm in soil (Efroymson et al. 1997b).

Copper occurs naturally in many animals and plants and is an essential micronutrient that animals incorporate into several essential enzymes. Copper may exist in two oxidation states: +1 or +2. Copper (+1) is unstable and, in aerated water over the pH range of most natural waters (6 to 8), oxidizes to the +2 state (EPA 1999). Copper is not biodegraded or

transformed and does not bioaccumulate (EPA 1999). Adverse effects from copper exposure include hematological, hepatic, developmental, immunological, and renal impairment.

Experiments with domestic poultry show that copper accumulates in livers of mallard ducklings at dietary concentrations as low as 15 mg/kg DW ration. Further, gizzard histopathology and a reduction in weight gain of chicks (*Gallus* sp.) occur at 250 to 350 mg Cu/kg DW ration, and growth of turkey poults is improved at 60 mg Cu/kg DW ration and inhibited at 120 mg/kg DW ration, with signs of gizzard histopathology at 500 mg/kg DW ration (Eisler 1998). Adverse effects from copper were predicted to occur at an oral dose of 61.7 mg/kg/day for avian species based on a study of copper oxide toxicity to chicks (Sample et al. 1996).

Copper can be lethal to mammals through a variety of routes (Eisler 1998). Dietary copper is lethal when eaten for extended periods at more than 80 mg Cu/kg ration in sheep (equivalent to 5.1-10.7 mg Cu/kg BW daily), more than 238 mg/kg ration in pigs, and more than 4,000 mg/kg ration in rats (equivalent to more than 133 mg Cu/kg BW daily). Adverse sublethal effects of copper to sensitive mammals occur in cattle at dietary levels greater than 20 mg Cu/kg BW by way of intraperitoneal injection and more than 4.2 mg Cu/kg BW via drinking water; in sheep given daily oral doses of 7.5 to 15.0 mg Cu/kg BW or fed diets containing more than 37.3 mg Cu/kg ration; in rats at greater than 100 mg Cu/kg ration (equivalent to greater than 7.9 mg Cu/kg BW daily), greater than 400 mg Cu/L drinking water, or greater than 2.0 to 2.5 mg Cu/kg BW daily via injection; and in pigs at more than 14.5 mg Cu/kg BW daily via diet. Adverse effects from copper were predicted to occur in mammals at oral doses ranging from 5.6 to 52.3 mg/kg/day depending on body size and species based on a study of copper sulfate toxicity to mink (Sample et al. 1996).

Copper exerts its toxic effects by binding to DNA or by generating free radicals (EPA 1999). Aqueous copper speciation and toxicity depend on the ionic strength of the water. Primarily it is the dissolved cupric ion (Cu+2) and possibly hydroxyl complexes that are toxic to aquatic biota; copper complexes consisting of carbonates, phosphates, nitrates, ammonia, and sulfates are weakly toxic or nontoxic (EPA 2000a). In hard, moderately polluted waters, 43 to 88 percent of the copper is associated with suspended solids and not available to biota (Eisler 1998). In general, mortality of tested aquatic species is greatest under conditions of low water hardness (as measured by $CaCO_3$), starvation, elevated water temperatures, and early developmental stages. Diet is the most important route of copper accumulation in aquatic animals, and food choice influences body loadings of copper (Eisler 1998).

A considerable number of aquatic species are sensitive to dissolved concentrations of copper in the range of 1-10 µg/L (EPA 2000a). Sensitive species of representative freshwater plants and animals die within 96 h at waterborne copper concentrations of 5.0

to 9.8 ug/L (Eisler 1998). The most sensitive freshwater species have LC50 (96 h) values between 0.23 and 0.91 ug Cu/L and include daphnids (*Daphnia* spp.), amphipods (*Gammarus pseudolimnaeus*), snails (*Physa* spp.), and chinook salmon (*Oncorhynchus tshawytscha*). In aquatic invertebrates, copper causes gill damage at high concentrations, and in fishes it interferes with osmoregulation (Eisler 1998). Adverse sublethal effects of copper on behavior, growth, migration, and metabolism occur in representative species of fishes at nominal water concentrations between 4 and 10 ug/L and lethal and reproductive effects occur in fish at concentrations of 10 to 20 ug/L (Eisler 1998).

Lead

Predicting the accumulation and toxicity of lead is difficult since its effects are influenced to a large degree, relative to other metals, by interactions among physical, chemical, and biological variables. Under controlled conditions, lead adversely affects survival, growth, reproduction, development, and metabolism of most species (Eisler 1988). In general, organolead compounds are more toxic than inorganic lead compounds; and young, immature organisms are more susceptible to lead's effects (Eisler 1988). Food chain biomagnification of lead is negligible. For vertebrates, lead is known to modify the structure and function of the kidney, bone, central nervous system, and the hematopoietic system. It produces adverse biochemical, histopathological, neuropsychological, ferotoxic, teratogenic, and reproductive effects. Inhibition of blood delta aminolevulnic acid dehydratase (ALAD), an enzyme critical in heme formation, has been observed as a result of exposure to lead in a variety of fish, invertebrates, and birds (EPA 2000a).

Lead does not play an essential role in plant metabolism (Kabata-Pendias and Pendias 1984). Lead is readily taken by roots when present in soluble forms in soil. Lead is taken up passively by roots, and translocation to shoots is limited (Efroymson et al. 1997a). It is bound to the outside of roots, in the apoplast, and in cell walls and organelles of absorbing roots. In the plant, lead may exist in a naturally chelated form, or in pyro- or orthophosphate forms. Uptake of lead by terrestrial plants is limited by the low bioavailability of lead from soils (Eisler 1988). The phytotoxicity of lead is relatively low compared with other trace elements. It affects mitochondrial respiration and photosynthesis by disturbing electron transfer reactions. The primary symptom of lead toxicosis is reduction of root and shoot growth. Phytotoxicity has been observed at lead concentrations as low as 100 ppm in soil (Kabata-Pendias and Pendias 1984; Efroymson et al. 1997a).

Beyer and Cromartie (1987) and Kabata-Pendias and Pendias (1985) showed that lead has a high potential to bioaccumulate in earthworms (BAFs ranging from 0.01 to 228 depending upon soil conditions). Soil invertebrates show adverse effects at lead concentrations of 500 ppm in soil (Efroymson et al. 1997b).

Lead is considered to be a non-essential element for wildlife with no nutritional or biochemical function (NAS 1980). Birds and mammals suffer effects from lead poisoning such as damage to the nervous system, kidneys, liver, sterility, growth inhibition, developmental retardation, and detrimental effects in blood (Eisler 1988). The bioavailability of lead is dependent on diet, growth rate, and physiological stress; of the lead that is available, approximately 90 percent accumulates in bones (NAS 1980). Lead absorption is 10 percent or less in mammals; however, young mammals have been shown to absorb lead at a much higher rate that adults. In addition to storage of lead in bone, lead also targets the kidney. Predicting the accumulation and toxicity of lead is difficult since its effects are influenced to a large degree, relative to other metals, by interactions among physical, chemical, and biological variables. The mechanism by which lead acts is believed to be indirect interference in normal metal-dependent enzyme functions at specific cellular sites. Toxicological effects may include abnormalities in hematological, neural, renal, and skeletal systems. Irreversible central nervous system damage and decreased intelligence at extremely low doses of lead has been observed in mammals (ATSDR 1997). Adverse effects from lead were predicted to occur in mammals at oral doses ranging from 22 to 209 mg/kg/day, depending on body size and species, based on a study of lead acetate toxicity to rats (Sample et al. 1996).

Among sensitive species of birds, survival was reduced at doses of 75 to 150 mg Pb2+ /kg BW or 28 mg alkyl lead/kg BW, reproduction was impaired at dietary levels of 50 mg Pb2+ /kg, and signs of poisoning were evident at doses as low as 2.8 mg alkyl lead/kg BW (Eisler 1988). Among sensitive species of mammals, survival was reduced at acute oral doses as low as 5 mg/kg BW in rats, at chronic oral doses of 0.3 mg/kg BW in dogs, and at dietary levels of 1.7 mg Pb/kg BW in horses. Sublethal effects were documented in monkeys given doses as low as 0.1 mg Pb/kg BW daily (impaired learning 2 years postadministration), or fed diets containing 0.5 mg Pb/kg (abnormal social behavior). Adverse effects from lead were predicted to occur at an oral dose concentration of 11.3 mg/kg/day for avian species based on a study of lead acetate toxicity to Japanese quail (Sample et al. 1996).

In aquatic environments, waterborne lead was the most toxic form and organolead compounds are much more toxic to aquatic organisms than are the inorganic lead compounds (EPA 2000a; Eisler 1988). The common forms of dissolved lead are lead sulfate, lead chloride, lead hydroxide, and lead carbonate, but the distribution of salts is highly dependent on the pH of the water. Most lead entering surface waters is precipitated in the sediment as carbonates or hydroxides and bioavailability from the sediment is controlled by the sediment organic content and AVS concentration (EPA 2000a). Adverse effects were noted on daphnid reproduction at 1.0 ug/L of Pb2+ and on rainbow trout survival at 3.5 ug/L of tetraethyllead. High bioconcentration factors were recorded for filter-feeding bivalve molluscs and freshwater algae at 5.0 ug Pb 2+/1. Fish exposed to high levels of lead exhibit a wide-range of effects including muscular and

neurological degeneration and destruction, growth inhibition, mortality, reproductive problems, and paralysis (Eisler 1988).

Lead is accumulated by aquatic organisms equally from water and through dietary exposure (EPA 2000a). Although methylated lead is rapidly bioaccumulated from the water by trout, for example, there is no evidence that lead biomagnifies in the aquatic environment. Log BCFs of 5.15 (cladoceran) and 3.56 (midge) were reported in the literature (EPA 2000a).

Mercury

For all organisms tested, early developmental stages were the most sensitive to mercury, and organomercury compounds—especially methylmercury—were more toxic than inorganic forms (Eisler 1987a). Numerous biological and abiotic factors modify the toxicity of Hg compounds, sometimes by an order of magnitude or more, but the mechanisms are not clear.

Mercury is not an essential plant nutrient (Kabata-Pendias and Pendias 1984). Although mercury in solution is readily taken up by plants, the relationship between levels in soils and plants is weak due to low bioavailability. Organic forms of Hg may be translocated to a greater degree than inorganic forms in some plants (Efroymson et al. 1997). Mercury appears to interfere with sulfur-containing enzymes and disrupt the metabolic processes of plants (Kabata-Pendias and Pendias 1984). Mercury also inhibits potassium uptake of plants. Symptoms of mercury toxicity include stunting of seedling and root growth, and the inhibition of photosynthesis that leads to reduced growth and yield. Phytotoxicity appears to occur at soil copper concentrations as low as 0.3 ppm (Kabata-Pendias and Pendias 1984; Efroymson et al. 1997a).

Kabata-Pendias and Pendias (1984) showed that mercury has a low potential to accumulate in earthworms (BAFs ranging from 0.33 to 0.40 depending on test conditions). Soil invertebrates show adverse effects at mercury concentration as low as 0.5 ppm (Efroymson et al. 1997b).

Mercury is a highly toxic compound with no known natural biological function as an essential element (EPA 1999; NAS 1980). Mercury exists in three valence states: mercuric (Hg^{+2}), mercurous (Hg^{+1}), and elemental (Hg^{+0}) mercury. Inorganic mercury compounds are less toxic than organomercury compounds with the compound methylmercury being of greatest toxic concern. Bacteria commonly present in the environment are able to convert inorganic forms of mercury to organic forms. Methylmercury is highly stable, lipophilic, and known to bioaccumulate and biomagnify in terrestrial food chains (EPA, 1999). The majority of mercury detected in biological tissues is present in the form of ethylmercury. Wildlife can be exposed to mercury through oral and inhalation pathways. In all receptors, the target organs are the kidney and central nervous system. The

mechanism of mercury toxicity is interference with metabolism and cell division. Mercury binds strongly with sulfhydryl groups causing inhibition or inactivation of proteins containing thiol ligands and ultimately leading to miotic disturbances (EPA 1999).

Mercury is a known mutagen, teratogen, and carcinogen (Eisler 1987a). At comparatively low concentrations in birds and mammals, it adversely affects reproduction, growth and development, behavior, blood and serum chemistry, motor coordination, vision, hearing, histology, and metabolism. In mammals, methyl mercury irreversibly destroys the neurons of the central nervous system and can also be teratogenic and mutagenic. It has a high potential for bioaccumulation and biomagnification, and is slow to depurate. Organomercury compounds were more effective in producing adverse effects than were inorganic Hg compounds; however, effects were significantly enhanced or ameliorated by numerous biotic and nonbiological modifiers. Lethal concentrations of total Hg to sensitive, representative organisms varied from 2.2 to 31.0 mg/kg body weight (acute oral) and 4.0 to 40.0 mg/kg (dietary) for birds; and from 0.1 to 0.5 mg/kg body weight (daily dose) and 1.0 to 5.0 mg/kg (dietary) for mammals (Eisler 1987a). For sensitive species of birds, sublethal harmful levels were 640 ug Hg/kg body weight daily, or 50 to 500 ug Hg/kg in the diet; for sensitive mammals, these levels were 250 ug Hg/kg body weight daily, or 1,100 ug Hg/kg diet. Adverse effects from methyl mercury were predicted to occur in mammals at oral doses ranging from 0.045 to 0.418 mg/kg/day depending on body size and species based on a study of methyl mercury chloride toxicity to rats and mink (Sample et al. 1996). For avian species, adverse effects from methylmercury were predicted to occur at an oral dose concentration of 0.064 mg/kg/day based on a study of methyl mercury dicandiamide toxicity to Japanese quail (Sample et al. 1996). As expected, predicted adverse effect concentrations from non-methylated mercury occur at higher concentrations: 3.4 mg/kg/day to 18.7 mg/kg/day for mammals and 0.90 mg/kg/day for avian species.

There is a high potential for bioaccumulation and biomagnification with mercury, with biomagnified concentrations reported in fish up to 100,000 times the ambient water concentrations (Eisler 1987a). Bacterial synthesis of methylmercury from inorganic mercury compounds present in the water or sediments is the major source of this molecule in aquatic environments (Matilainen et al. 1991). Methylmercury is highly water soluble and has a K_{ow} that varies dependent upon the pH and ionic strength of water (Major et al. 1991). Mercury, at comparatively low concentrations, adversely affects the reproduction, growth, behavior, metabolism, blood chemistry, osmoregulation, and oxygen exchange of marine and freshwater organisms. In general, the accumulation of mercury by aquatic biota is rapid, and depuration is slow. It is emphasized that organomercury compounds, especially methylmercury, were significantly more effective than inorganic mercury compounds in producing adverse effects and accumulations. Lethal concentrations of total Hg to sensitive, representative organisms varied from 0.1 to

2.0 ug/L of medium for aquatic fauna. Reproduction was inhibited among sensitive species of aquatic organisms at water concentrations of 0.03 to 0.1 ug/L.

Silver

Silver is not an essential plant nutrient (Kabata-Pendias and Pendias 1984). Silver taken up by plants remains in the root system precipitated with phosphate or chloride (Efroymson et al. 1997a). The toxicity of silver is related to the binding potential of silver ions to enzymes and other active molecules at cell surfaces. Although silver has been shown to reduce growth of plants, no other toxicity symptoms have been observed. Phytotoxicity appears to occur at soil silver concentrations as low as 2.0 ppm (Kabata-Pendias and Pendias 1984; Efroymson et al. 1997a).

No toxicity information was found describing the toxic effects of silver on soil biota.

Silver has no known biological function, but it is a normal trace constituent of many organisms (NAS 1980; Irwin et al. 1997). At low doses silver in mammals is found in highest concentrations in the gastrointestinal tract. However, at high doses the highest concentrations are found in the liver and the spleen (NAS 1980). Silver, as ionic Ag+, is one of the most toxic metals known to aquatic organisms in laboratory testing (Irwin et al. 1997). Therefore, most toxicity information available for silver focuses on its aquatic toxicity. Less is known on the toxicity of silver to wildlife. The most likely route of exposure to silver by wildlife is ingestion of food and water (Irwin et al. 1997). Signs of chronic silver intoxication in tested birds and mammals included cardiac enlargement, vascular hypertension, hepatic necrosis, anemia, lowered immunological activity, altered membrane permeability, kidney pathology, enzyme inhibition, growth retardation, and a shortened life span (Irwin et al. 1997). Silver was not mutagenic, carcinogenic, or teratogenic to tested animals by normal routes of exposure (Eisler 1996). Adverse effects of silver on poultry occur at 1.8 mg/kg FW whole egg by way of injection (reduced survival), 10 mg/kg in copper-deficient diets (reduced hemoglobin), and 200 mg/kg in copper-adequate diets (growth suppression), or when the birds are given drinking water containing 100 mg Ag/L (liver necrosis). Effects of silver on sensitive species of mammals include death at 13.9-20.0 mg/kg BW by intraperitoneal injection, histopathology of kidney and brain at 250-450 mg Ag/L drinking water, tissue accumulations at 6 mg/kg diet, and liver necrosis when fed diets containing more than 130 mg/kg. In humans, generalized argyria seems to be declining, which may be due to improved work conditions.

Silver does not appear to be a highly mobile element under typical conditions in most aquatic habitats (EPA 2000a). Free silver ion is lethal to representative species of sensitive aquatic plants, invertebrates, and teleosts at water concentrations of 1.2-4.9 mg/L (Eisler 1996). Adverse effects occur on development of trout at concentrations as low as 0.17 mg/L and on phytoplankton species composition and succession at 0.3 to 0.6 mg/L.

Aquatic organisms accumulate silver from environmental sources. In fish and amphibian toxicity tests with 22 metals and metalloids, silver was the most toxic tested element as judged by acute LC50 values. In solution, ionic silver is extremely toxic to aquatic plants and animals, and water concentrations of 1.2-4.9 mg/L killed sensitive species of aquatic organisms, including representative species of insects, daphnids, amphipods, trout, flounders, sticklebacks, guppies, and dace (Eisler 1996). At nominal water concentrations of 0.5 to 4.5 mg/L, accumulations in most species of exposed organisms were high and had adverse effects on growth in algae, clams, oysters, snails, daphnids, amphipods, and trout, molting in mayflies, and histopathology in mussels. Among all tested species, the individuals most sensitive to silver were the poorly nourished and young and those exposed to low water hardness or salinity. It is emphasized that silver-induced stress syndromes vary widely among animal classes. Among marine organisms, for example, silver ion was associated with respiratory depression in marine gastropods and cunners (*Tautagolabrus adspersus*), a teleost; however, silver ion increased oxygen consumption in six species of bivalve mollusks.

Little evidence exists to support the general occurrence of biomagnification of silver within marine or freshwater food webs. Silver uptake by aquatic organisms appears to be almost entirely from the dissolved form. When silver was bound to algal cell membranes, it could not be dislodged by either mechanical disruption or leaching at low pH; therefore, silver bound to algal cells is likely not assimilable by higher organisms (EPA 2000a).

Zinc

Zinc is an essential element for plant growth (Efroymson et al. 1997a). It has a part in many enzymes and is involved in disease protection and metabolism of carbohydrates and proteins. Zinc is actively taken up by roots in ionic form and, to a lesser extent, in organically chelated form. It is fairly uniformly distributed between roots and shoots being transported in the xylem in ionic form. Transport in the phloem appears to be as an anionic complex. Zinc acts to inhibit CO fixation, phloem transport of carbohydrates, and alter membrane permeability. Toxicity symptoms include chlorosis and depressed plant growth. Phytotoxicity appears to occur at soil zinc concentrations as low as 50 to 70 ppm (Kabata-Pendias and Pendias 1984; Efroymson et al. 1997a).

Beyer and Cromartie (1987) and Kabat-Pendias and Pendias (1985) report that zinc has a high potential to accumulate in earthworms. (Reported BAFs range from 0.1 to 26 depending upon soil conditions.) Soil invertebrates show adverse effects at zinc concentration as low as 100 ppm (Efroymson et al. 1997b). Representative soil invertebrates showed reduced growth at 300-1,000 mg Zn/kg diet and reduced survival at 470-6,400 mg Zn/kg soil (Eisler 1993).

Zinc is required for normal growth, development, and function in all animal species that have been studied (NAS 1980). It is absorbed from the intestinal tract as needed and is primarily excreted in the feces. Zinc attaches to organic molecules such as amino acids, proteins, and nucleic acids through directly binding to sulfhydryl, amino, imidazole, and phosphate groups (NAS 1980). Zinc has low toxicity to birds and mammals. Exposures to high concentrations of zinc may result in reduced weight gain, anemia, reduced bone ash, decreased tissue concentrations of iron, copper, and manganese, and decreased use of calcium and phosphorus (NAS 1980). Domestic poultry and avian wildlife had reduced growth at >2,000 mg Zn/kg diet, and reduced survival at >3,000 mg Zn/kg diet or at a single oral dose >742 mg Zn/kg BW; younger stages (i.e., chicks, ducklings) were least resistant (Eisler 1993). Sensitive species of livestock and small laboratory animals were adversely affected at >0.8 mg Zn/m^3 air, 90-300 mg Zn/kg diet, >90 mg Zn/kg BW daily, >300 mg Zn/L drinking water, and >350 mg Zn/kg BW single oral dose. Adverse effects from zinc were predicted to occur in mammals at oral dose ranging from 89.8 to 836.4 mg/kg/day depending on body size and species based on a study of zinc oxide toxicity to rats (Sample et al. 1996). For avian species, adverse effects from zinc were predicted to occur at an oral dose of 131 mg/kg/day based on a study of zinc sulfate toxicity to leghorn hens (Sample et al. 1996).

Zinc in the water column can partition to dissolved and particulate organic carbon. Bioavailability of zinc in sediments is controlled by the AVS concentration. Water hardness (i.e., calcium concentration), pH, and metal speciation are important factors in controlling the water column concentrations of zinc since the divalent zinc ion is believed to be responsible for observed biological effects (EPA 2000a). Significant adverse effects of zinc on growth, survival, and reproduction occur in representative sensitive species of aquatic plants, protozoans, sponges, molluscs, crustaceans, echinoderms, fish, and amphibians at nominal water concentrations between 10 and 25 μg Zn/L (Eisler 1993). Acute LC50 (96 h) values for freshwater invertebrates were between 32 and 40,930 μg Zn/L; in fish, this range was 66 to 40,900 μg/L. Daphnids and trout have been identified as some of the most sensitive species with adverse effects occurring at concentrations between 5 and 19 ug/L (Eisler 1993). In general, zinc is more toxic to embryos and juveniles than to adult, to starved animals, at elevated temperatures, in the presence of cadmium and mercury, in the absence of chelating agent, at reduced salinities, under conditions of marked oscillations in ambient zinc concentrations, at decreased water hardness and alkalinity, and at low dissolved oxygen concentrations. Zinc is not a highly mobile element in aquatic food webs and there appears to be little evidence to support the general occurrence of biomagnification of zinc within marine or freshwater food webs; a log biomagnification factor of 2.90 was determined for the midge Chironomus riparius (EPA 2000a). Bioconcentration factors (BCF) for zinc accumulation from the medium varied widely between and within species of aquatic organisms. For representative freshwater organisms, BCF values ranged from 107 to 1,130 for insects and from 51 to 432 for fish (Eisler 1993).

ORGANIC COMPOUNDS

Dichlorodiphenyl-trichloroethane (DDT)

Chronic effects of DDT and its metabolites on ecological receptors include changes in enzyme production, hormonal balance, and calcium metabolism, which may cause changes in behavior and reproduction. The most well documented response is eggshell thinning in birds which results in embryo mortality, and decreased hatchling survival. Because of the tendency of DDT to magnify in food chains, higher trophic level birds appear to be at greater risk for egg loss due to shell thinning. Eggshell thinning of greater than 20 percent has been associated with decreased nesting success due to eggshell breakage (Anderson and Hickey 1972; Dilworth et al, 1972).

Another well defined effect of DDT exposure is inhibition of acetylcholinesterase (AChE) activity. Inhibition of this enzyme results in the accumulation of acetylcholine in the nerve synapses, resulting in disrupted nerve function. Chronic inhibition of 50 percent of brain AChE has been associated with mortality in birds (Ludke et al. 1975).

The effects of DDT on other receptor groups are not as clearly defined. Invertebrate species are generally more susceptible than fish species to effects associated with exposure to DDT in the water column (U.S. EPA 2000a). Sediments contaminated with pesticides, including DDT, have been shown to affect benthic communities at sediment exceeding 2 μg/kg (U.S. EPA 2000a).

For fish, the primary route of uptake is via prey items, but both DDT and its metabolites can be accumulated through the skin or gills upon exposure to water. Short-term exposure to DDT concentrations of less than 1 μg/L in water and 1.1 to 2.4 mg/kg in fish embryos have been reported to elicit toxic responses in fish (U.S. EPA 2000a).

Eggshell thinning has been linked to chronic exposure to DDT and its metabolites in the diet of birds. The mode of action of DDT is to affect the activity of Ca^{2+} ATP ase systems in the shell gland are affected, thereby interfering with the deposition of calcium in the shell (Lundholm 1987). Evidence strongly indicates that DDE is responsible for most reproductive toxicity in avian species (U.S. EPA 2000a). Measurements of residues in eggs of birds are a reliable indicator of adverse effects. There is a large amount of variability in sensitivity to DDT and its metabolites among bird species, with waterfowl and raptor species showing the greatest sensitivities. Studies have shown the brown pelican to be most susceptible to adverse effects, with eggshell thinning and depressed productivity occurring at 3.0 μg/g of DDE in the egg and total reproductive failure when residues exceed 3.7 μg/g (U.S. EPA 2000a).

Several studies have been conducted in which the effects of organochlorine toxicants on mink have been investigated (Giesy et al. 1994; Jensen et al. 1977; Aulerich and Ringer

1970; Duby et al. 1971; Frank et al. 1975; Proulx et al. 1987). Generally, population sensitive endpoints of survival, growth, and reproduction have been evaluated, with reproduction being the most sensitive endpoint. Based on these studies, concentrations above 100 mg/kg may be sufficient to cause toxicity to mammals.

Dioxins/Furans

Information is lacking or scarce on the biological properties of PCDD isomers, except 2,3,7,8-TCDD (Eisler 1986b). The latter has been associated with lethal, carcinogenic, teratogenic, reproductive, mutagenic, histopathologic, and immunotoxic effects. There are substantial inter- and intraspecies differences in sensitivity and toxic responses to 2,3,7,8-TCDD. Typically, animals poisoned by 2,3,7,8-TCDD exhibit weight loss, atrophy of the thymus gland, and eventually death. The toxicological mechanisms are imperfectly understood. Dioxins are not generally acutely toxic to adult organisms, but their long term accumulation is thought to be expressed chronically, and may ultimately result in death.

Dioxins are highly lipophilic compounds that tend to bind to organic matter in the soil. They may accumulate of root surfaces of plants. However, since dioxins are high molecular weight compounds, they have a negligible potential to translocate from roots into plants via the zylem (McCrady and Maggard 1993; Bacci et al. 1992; McCrady et al. 1990). Dioxins are volatile and can sorb to the leaf cuticle when volitalized (ATSDR 1998). Wet and dry deposition of dioxin-containing particles is also an important mode of accumulation of dioxins on plants. However, dioxins sorbed onto plant surface have a negligible potential to enter the tissues (McCrady et al. 1990). Since dioxins are not taken up into plants, exposure is incomplete and toxicity has not been reported.

Reinecke and Nash (1984) show that dioxin has a moderate to high potential to bioaccumulate in earthworms (BAFs range from 0.17 to 9.4 depending on soil conditions). EPA (2003) states that a wide variety of invertebrates including amphipods, cladocerans, midges, mosquito larvae, sandworms, oligochaete worms, snails, clams, and grass shrimp are insensitive to 2,3,7,8-TCDD toxicity. Likewise, dioxin-like PCBs are generally ineffective in reducing survival, growth, and reproduction in the cladoceran Daphnia magna and the purple sea urchin. The insensitivity of invertebrates to dioxin-like toxicity is consistent with the recent finding that they lack the protein necessary to mediate the toxic effects of dioxin.

Polychlorinated dibenzodioxins (PCDD), as a group, represent 75 different positional isomers, while polychlorinated dibenzofurans (PCDF) comprise over 135 compounds (ATSDR 1998). These two chemical classes are generally referred to as dioxins and tetra-chloro dibenzodioxins (TCDD) and tetra-chloro dibenzofurans (TCDF) are a subset of these compounds. Dioxins are lipophylic and persistent in the environment. They are both naturally occurring, for example as a result of wood combustion, and unintended

manufacturing byproducts. Dioxins have no nutritional value and are known to bioaccumulate and be highly toxic to wildlife species. Wildlife are exposed to dioxins primarily through prey consumption. Dioxin toxicity is believed to be mediated intracellularly by binding with the aryl hydrocarbon receptor (Ah-R). The resulting Ah-R complex moves into the cell nucleus, where it will bind to the DNA, and may alter the expression of a number of gene sequences. Many of the observed toxic effects of dioxins (and the coplanar PCBs) are attributable to specific alterations in gene expression (EPA 2000b). Unlike most toxic chemicals, the lethality of TCDD is delayed and species specific (EPA 2000b). The characteristic signs and symptoms of lethal toxicity by TCDD are severe weight loss and thymic atrophy (EPA, 2000b). Other toxic effects include hyperplasia or atrophy of the spleen, testes, or ovaries, bone marrow depletion, and systemic hemorrhage (EPA 2000b). Dioxins are believed to cause alterations to developmental endocrine (thyroid and steroid hormones) and immune functions, as well as interference in vitamin production, which results in disruption of patterns of embryonic development at critical stages (EPA 2000b). General population level manifestations of dioxin exposure include adversely affected patterns of survival, reproduction, growth, and resistance to diseases (Eisler and Belisle 1996).

LD50 values computed 37 days after a single oral dose of 2,3,7,8-TCDD varied from 15 ug/kg body weight in Northern bobwhite (*Colinus virginianus*), with 95% confidence limits of 9.2 and 24.5 ug/kg, to more than 810 ug/kg bodyweight for the ringed turtle-dove (*Streptopelia risoria*) (Eisler 1986b). Mallards (*Anas platyrhynchos*) were intermediate in sensitivity with an acute oral LD50 value of more than 108 ug/kg body weight. For all 3 species, death occurred 13 to 37 days after treatment; remission in survivors had apparently occurred by day 30 post-treatment. Domestic chickens were relatively sensitive to PCDDS, especially 2,3,7,8-TCDD, with an estimated 2,3,7,8-TCDD oral LD-50 range of 25 to 50 ug/kg body weight (Eisler 1986b). Chickens fed 1 or 10 ug of 2,3,7,8-TCDD, 1,2,3,7,8,9-hexa CDD, or hepta-CDDs per kg body weight daily for 21 days showed signs of chick edema disease (i.e., pericardial, subcutaneous, and peritoneal edema, liver enlargement and necrosis with fatty degeneration; frequently resulting in death). Adverse effects from 2,3,7,8-TCDD were predicted to occur at an oral dose of 0.000014 mg/kg/day for avian species based on a study of 2,3,7,8-TCDD toxicity ring-necked pheasants (Sample et al. 1996).

Acute toxicity studies with 2,3,7,8-TCDD have shown marked differences—up to 8,400X—between the single oral LD50 dose for the guinea pig and the hamster (*Cricetus* sp.) (Eisler 1986b). The acute oral LD50 value of 0.6 ug/kg body weight for guinea pigs, suggests that 2,3,7,8-TCDD may be the most toxic compound ever tested on small laboratory animals. The unusual resistance of hamsters may be associated with its enhanced rate of metabolism and excretion of 2,3,7,8-TCDD relative to other PCDD isomers examined. Poisoning in mammals by 2,3,7,8-TCDD is typically characterized by loss of body weight and delayed lethality; large interspecies differences exist in lethal dosages and toxic effects. Intraspecies differences in sensitivity to 2,3,7,8-TCDD, up to 14

fold, were recently reported among 3 strains of mice (Eisler 1986b). Atrophy of the thymus is a consistent finding in mammals poisoned by 2,3,7,8-TCDD, and suppression of thymus-dependent cellular immunity, particularly in young animals, may contribute to their death. Developing mammalian fetuses are especially sensitive to 2,3,7,8-TCDD, and maternal exposure results in increased frequencies of stillbirths. The limited data available suggest that 2,3,7,8-TCDD concentrations of 10 to 12 ppt in food items should not be exceeded for birds and other wildlife. Adverse effects from 2,3,7,8-TCDD were predicted to occur in mammals at oral doses ranging from 0.0000011 to 0.000022 mg/kg/day depending on body size and species based on a study of 2,3,7,8-TCDD toxicity to rats (Sample et al. 1996).

Aquatic invertebrates are presumed to lack the Ah receptor, and, as such, are thought to be relatively insensitive to dioxins. *Daphnia magna* exposed to nominal concentrations of TCDD in water were not affected at concentrations as high as 1,030 ng/L. Similarly, dioxins have been reported to bioaccumulate in benthic invertebrates that also lack the Ah receptor to significant concentrations without adverse effects. West et al. (1997) exposed *Chironomus tentans* and *Lumbriculus variegatus* to dietary concentrations of TCDD and no toxic effects were observed in full life-cycle tests with either species at tissue residue concentrations up to 9,533 ug/kg-lipid of TCDD. Sensitive species of fish exhibited reduced growth and fin necrosis at concentrations as low as 0.1 ppt of 2,3,7,8-TCDD after exposure for 24 to 96 hours (Eisler 1986b). Concentrations of 1.0 ppt and higher were eventually fatal, and exposure to lower concentrations of 0.01 ppt for 24 hours had no measurable effect. Invertebrates, plants, and amphibians were comparatively resistant to 2,3,7,8-TCDD. Accumulation of 2,3,7,8-TCDD from the aquatic environment was evident for all species examined (Eisler 1986b). In outdoor pond studies, a major portion of the added 2,3,7,8-TCDD concentrated in aquatic plants and at the sediment-water interface; however, most (85-99%) of the 2,3,7,8-TCDD originally added to the ecosystem remained in the sediments at the end of the study. Among bony fish, body burdens of 2,3,7,8-TCDD increased with increasing concentration in the water column and with increasing duration of exposure; on removal to uncontaminated water, less than 50% was lost in 109 days.

Pentachlorphenol

The toxicity of commercial or technical grades of pentachlorophenol (PCP) significantly exceeds that of analytical or purified PCP (Eisler 1989). Some of this added toxicity is attributed to impurities such as dioxins, dibenzofurans, chlorophenols, and hexachlorobenzene. Pentachlorophenol is rapidly accumulated, rapidly excreted, and has little tendency to persist in living organisms. It acts by uncoupling oxidative phosphorylation. PCP affects energy metabolism by increasing oxygen consumption and altering the activities of several glycolytic and citric acid cycle enzymes and by increasing the consumption rate of stored lipid (EPA 2000a).

Terrestrial plants and soil invertebrates were adversely affected at 0.3 mg PCP/L (root growth), and at 1 to 5 g PCP/m^2 soil (reduction in soil biota populations) (Eisler 1989). PCP may be phytotoxic at a soil concentration as low as 3 ppm (Efroymson et al. 1997a) and may be toxic to soil biota at soil concentrations as low as 30 ppm (Efroymson et al. 1997b).

Fatal PCP doses for birds were 380 to 504 mg/kg BW (acute oral), >3,850 mg/kg in diets, and >285 mg/kg in nesting materials (Eisler 1989). Adverse sublethal effects were noted at dietary levels as low as 1.0 mg/kg ration. Residues (mg/kg fresh weight) in birds found dead from PCP poisoning were >11 in brain, >20 in kidney, >46 in liver, and 50 to 100 in egg.

Data are scarce on the toxicity of PCP to mammalian wildlife, but studies with livestock and small laboratory animals show that the chemical is rapidly excreted (Eisler 1989). However, there is great variability between species in their ability to depurate PCP, as well as in their overall sensitivity. Acute oral LD-50's in laboratory animals were 27 to 300 mg/kg BW. Tissue residues were elevated at dietary levels as low as 0.05 mg/kg feed and at air levels >0.l mg/m3. Histopathology, reproductive impairment, and retarded growth were evident at doses of 0.2 to 1.25 mg/kg BW, and when the diets fed contained >30 mg PCP/kg. Adverse effects from pentachlorophenol were predicted to occur in mammals at oral doses ranging from 0.673 to 6.273 mg/kg/day depending on body size and species based on a study of zinc oxide toxicity to rats (Sample et al. 1996).

In aquatic biota, pentachlorophenol affects energy metabolism by partly uncoupling oxidative phosphorylation and increasing oxygen consumption, by altering the activities of several glycolytic enzymes and the citric acid cycle enzymes, and by increasing the consumption rate of stored lipid (Eisler 1989). Collectively, these events could reduce the availability of energy for maintenance and growth and thereby reduce the survival of larval fish and the ability of prey to escape from a predator. Adverse effects on growth, survival, and reproduction of representative sensitive species of aquatic organisms occurred at PCP concentrations of about 8 to 80 ug/L for algae and macrophytes, about 3 to 100 ug/L for invertebrates (especially molluscs), and <1 to 68 ug/L for fishes, especially salmonids. The accumulation of PCP in fishes is rapid, and primarily by direct uptake from water rather than through the food chain or diet. Pentachlorophenol was most toxic and most rapidly metabolized in aquatic environments at elevated temperatures and reduced pH (Eisler 1989). Accumulation of PCP is pH-dependent; at pH 4, PCP is completely protonated and therefore highly lipophilic. At this pH, PCP has the greatest accumulation potential. Conversely, PCP is completely ionized at pH 9 (EPA 2000a). PCP is rapidly accumulated and rapidly excreted, and it has no tendency to persist in living organisms. However, PCP tends to accumulate in mammalian tissues unless it is efficiently conjugated into a readily excretable form (EPA 2000a).

Polycyclic Aromatic Hydrocarbons

Polycyclic aromatic hydrocarbons (PAHs) generally occur in the environment as complex mixtures. The toxicities of individual PAHs are additive and increase with increasing K_{ow}s , whereas the bioavailabilities of PAHs decrease as a function of their K_{ow}s (EPA 2000a). PAHs are highly potent carcinogens that can produce tumors in some organisms at even single doses; but other non-cancer-causing effects are not well understood (Eisler 1987b). A wide variety of PAH-caused adverse biological effects have been reported in numerous species of organisms under laboratory conditions, including effects on survival, growth, metabolism, and especially tumor formation (Eisler 1987b). Inter- and intra-species responses to carcinogenic PAHs were quite variable, and were significantly modified by many chemicals including other PAHs that are weakly carcinogenic or noncarcinogenic.

PAHs act primarily through a non-specific narcotic mode of action to cause toxicity, although more specific effects, such as adverse reproductive effects, may also occur. Nonpolar narcosis is the most commonly recognized mode of action resulting from exposure to most industrial organic chemicals. Narcotic toxicity is recognized as being rapidly and completely reversible, not dependant on chemical class, and dependent on lipophilicity, which results in a physical, but not chemical, change in cellular membranes (Schultz 1989). These physical changes in cell membranes result in altered to membrane fluidity, thickness, and surface tension. Ultimately, narcosis can cause cardiovascular-respiratory depression, medulary paralysis, and death.

Plants and vegetables can absorb PAHs from soils through their roots, and translocate them to other plant parts such as developing shoots (Eisler 1987b). Uptake rates were governed in part, by PAH concentration, PAH water solubility, soil type, and PAH physicochemical state (vapor or particulate). Lower molecular weight PAHs were absorbed by plants more readily than higher molecular weight PAHs. Under laboratory conditions, some plants concentrated selected PAHs above that of their immediate geophysical surroundings, but this has not been conclusively demonstrated in field-grown cultivated crops or other vegetation. Above-ground parts of vegetables, especially the outer shell or skin, contained more PAHs than underground parts, and this was attributed to airborne deposition and subsequent adsorption. PAH-induced phytotoxic effects were rare; however, the data base on this subject is small. Most higher plants can catabolize benzo(a)pyrene, and possibly other PAHs, but metabolic pathways have not been clearly defined. Phytotoxic effects of PAHs have been reported at soil concentrations ranging from 25 to >100 ppm (Efroymson et al. 1997a).

Few studies have been done on the toxic effect of PAHs to birds (Eisler 1987b). Mallards fed diets that contained 4,000 mg PAHs/kg (mostly as naphthalenes, naphthenes, and phenanthrene) for a period of 7 months showed no mortality or visible signs of toxicity during exposure. However, liver weight increased 25 percent and blood flow to liver

increased 30 percent, when compared to controls. In birds, evaluated concentrations of PAHs have been associated with impaired reproduction, growth retardation, morphological abnormalities, and metabolic and behavioral alterations (Eisler 2000).

Unsubstituted PAHs do not accumulate in mammalian adipose tissues despite their high lipid solubility, probably because they tend to be rapidly and extensively metabolized (Eisler 1987b). Biological half-life of PAHs is limited, as judged by rodent studies. In the case of benzo(a)pyrene and rat blood and liver, half-life values of 5 to 10 minutes were recorded; the initial rapid elimination phase was followed by a slower disappearance phase lasting 6 hours or more. In a study where mink were treated for 60 days prior to breeding and through kit weaning with either crude oil or bunker C fuel oil at a dietary concentration of 500 mg/kg adverse effects were found. Both exposures resulted in a significantly reduced number of live born kits per female and a significantly reduced number of kits surviving to weaning (Mazet et al. 2001). Adverse effects from benzo(a)pyrene were predicted to occur in mammals at oral doses ranging from 10.8 to 49.3 mg/kg/day depending on body size and species based on a study of benzo(a)pyrene toxicity to mice (Sample et al. 1996).

PAHs vary substantially in their toxicity to aquatic organisms. In general, toxicity increases as molecular weight increases (although high molecular weight PAHs have low acute toxicity, perhaps due to their low solubility in water) and with increasing alkyl substitution on the aromatic ring. Toxicity is most pronounced among crustaceans and least among teleosts (Eisler 2000). In all but a few cases, PAH concentrations that are acutely toxic to aquatic organisms are several orders of magnitude higher than concentrations found in even the most heavily polluted waters (Eisler 2000). Sediments from polluted regions, however, may contain PAH concentrations similar to those which are acutely toxic, but their limited bioavailability would probably render them substantially less toxic than PAHs in solution (Eisler 2000). Although fish efficiently metabolize PAHs, elevated concentrations in sediments have been linked to hepatic disorders, such as adenomas and carcinomas, and external lesions in such the bottom-feeding fish as common carp (*Cyprinus carpio*) and brown bullhead (*Ameirurus nebulosus*). NOAA found that PAH sediment concentrations ranging from 54 to 2,800 ng/g dry wt caused toxicopathic liver lesions in English sole and based on these results and assuming a sediment organic content of 2 percent, recommend a sediment quality guideline of 1,000 ug/kg total PAH to protect estuarine fish against degenerative liver lesions, spawning inhibition, and reduced egg viability.

Bioaccumulation of low-molecular-weight PAHs from sediments by Rhepoxynius abronius (amphipod) and Armandia brevis (polychaete) was similar; however, a large difference in tissue concentration between these two species was measured for high-molecular-weight PAHs (EPA 2000a). Conclusions drawn from this study were: 1) low-molecular-weight PAHs were available to both species from interstitial water; 2) sediment ingestion was a much more important uptake route for the high-molecular-weight PAHs;

and 3) bioavailability of the high-molecular weight-PAHs to amphipods was significantly reduced due to their partitioning to dissolved organic carbon.

Polychlorinated Biphenyls

Polychlorinated biphenyls (PCBs) can produce a wide variety of responses in organisms and have been reported as neurotoxicants, hepatotoxicants, immunotoxicants, and carcinogens (Safe 1991; Shain *et al*, 1991). While sensitivity and responses tend to be species-specific, general responses include lethality, reproductive and/or developmental toxicity, hepatic lesions, tumor promotion, suppression of the immune system, and induction of drug-metabolizing enzymes (McFarland and Clarke 1989; Safe, 1990; Eisler and Belisle 1996).

For vertebrates, PCBs induce metabolic breakdown in the liver through enzyme induction within the cytochrome P450 system (Eisler and Belisle, 1996). The degree of metabolic breakdown is primarily dependent on the degree of chlorination and their spatial arrangement. As the number of chlorine atoms in the PCB molecule increase, and the number of unsubstituted adjacent carbon atoms decrease, metabolic transformation decreases. PCB elimination is limited due to the highly lipophilic nature of these compounds. This causes PCBs to bioaccumulate in organisms and biomagnify up the food chain.

Of the 209 possible PCB congeners research has indicated that as much as 75 percent of tissue burdens of PCBs in invertebrates, fish, birds, and mammals is attributed to only 25 specific congeners (McFarland and Clarke 1989). These congeners with the greatest likelihood for bioaccumulation and toxicity are the planar non-, ortho-, or mono-ortho substituted PCBs, which chemically resemble and toxicologically behave similarly to the 2,3,7,8- substituted polychlorinated dibenzofurans (PCDFs) and dibenzo-p-dioxins (PCDDs) (Walker and Peterson 1991). Specifically, several lines of testing have implicated the planar PCB congeners 77, 81, 126, and 169 as major contributors to the toxicity of PCB mixtures (Ankley *et al*, 1991).

Examination of field and laboratory data suggest that many of the toxic effects caused by planar PCBs are mediated subcellularly by the aryl hydrocarbon receptor (Ah-R), the same receptor responsible for mediating dioxin toxicity. This receptor is involved in the translocation of PCBs into the nucleus and their subsequent binding to the PCH-Ah receptor complex on the DNA (Safe 1991). The signs of PCB 126 toxicity in lake trout early life stages are similar to those shown by TCDD, and include yolk-sac edema, multifocal hemorrhages, craniofacial malformation, in addition to mortality (Zabel *et al*, 1995). However, recent work has suggested that while the TCDD-like congeners act by a common mechanism (i.e., the Ah receptor), the combined effects of TCDD with the coplanar PCB congeners may not be strictly additive (Walker *et al*, 1996). Despite this uncertainty, the additive model continues to be acceptable for assessing risk because

deviation from additivity has been estimated to be within an accepted tenfold range (Walker *et al.* 1996).

As a means of normalizing toxicity amongst dioxin-like compounds, toxicity is expressed relative to the most toxic PCH (2,3,7,8-TCDD) by the use of toxic equivalency factors (TEFs) (Safe 1990, 1991). The term TEF is generally defined as the relative potency of a compound compared to the ability of 2,3,7,8-TCDD to cause a particular toxic or biological effect. TEFs are calculated by setting the toxic potency of 2,3,7,8-TCDD equal to 1.0, and determining the relative potencies of other PCHs as the ratio of the concentration of PCH to the concentration of 2,3,7,8-TCDD producing an equivalent response.

Multiplication of each congener concentration by its TEF generates a toxic equivalency (TEQ) for that congener. The sum of the TEQ for each congener yields the total TEQ for the mixture. This model assumes that congeners act additively through a similar mode of action to produce toxicity. The additivity of dioxin-like compounds is based on an Ah receptor-mediated response. Total TEQ levels, rather than individual congener levels, have been shown to better correlate with biological endpoints, such as lethality and deformities in fish, mammals, and birds (Ankley *et al.* 1989; Giesy *et al.* 1995, 1994a, 1994b; Tillitt *et al.* 1992; Walker and Peterson 1991; Zabel *et al.* 1995).

Invertebrates do not have an Ah receptor and are, therefore, not impacted by this receptor-mediated toxicity. Also, invertebrates have a limited cytochrome P450 detoxification system, so there is limited metabolic breakdown of these compounds. As a result, PCB toxicity to invertebrates is potentially less than that experienced by vertebrate species, and PCBs are retained in invertebrate tissues. The review of PCB toxicity by Niimi (1996) has suggested that PCB concentrations of greater than 10 μg/L cause zooplankton death within a few days, and concentrations of 1 to 10 μg/L cause death over longer periods of exposure. A sediment concentration of 0.059 mg/kg is sediment is a consensus low effect threshold for sediment invertebrates (MacDonald et al. 2000).

The effects of PCBs on Great Lakes fish and wildlife has been extensively documented. PCB-induced reproductive impairment has been demonstrated for several fish species (Mac 1988; Ankley *et al*, 1991; Walker and Peterson 1991; Walker *et al.* 1991a, 1991b; Williams and Giesy, 1992). Generally, the most sensitive endpoints for effects of PCBs in fish are early life-stage survival and recruitment where exposure has resulted from transfer of PCBs from maternal tissue to eggs (Eisler and Belisle 1996; Walker *et al.* 1996). Whole body environmental concentrations of PCBs in adult fish do not generally result in death (Eisler and Belisle, 1996). Numerous field studies evaluating PCB fish tissue concentrations and adverse effects, as summarized by Niimi (1996), also supports this. Based on several field studies, lethal body burden concentrations have been estimated at greater than 100 mg/kg for young fish and greater than 250 mg/kg for older fish (Niimi 1996).

PCB-induced reproductive impairment has been demonstrated for a number of insectivorous and piscivorous birds (Kubiak *et al.* 1989; Gilbertson *et al.* 1991; Tillitt *et al*,

1992) and mink (Aulerich *et al.* 1973, Aulerich and Ringe, 1977; Bleavins *et al.* 1980; Wren 1991; Giesy *et al.* 1994c; Heaton *et al.* 1995a, 1995b; Tillitt *et al.* 1996). Bird embryos are the most sensitive life stage for assessing the effects of contaminants (Elliott *et al.*, 1996; Kubiak and Best 1991). A study by Kubiak *et al.* (1989) showed that concentrations of 22 mg/kg in the fish eating Forster stern (*Sterna fosteri*) eggs were associated with significantly reduced hatching success.

PCB toxicity in mammals is highly variable. While some PCBs are extremely toxic, and can cause reproductive failure and produce death in very low levels, others appear to produce few, if any, toxic responses (Eisler 1986c). Toxic responses to PCBs are also highly species-specific. Mink have been shown to be highly sensitive to the effects of PCBs in their diet, and so they have been identified as an indicator species for water quality and ecosystem health in the Great Lakes. Younger mammals appear to be more susceptible to PCB poisoning than adults (Eisler 1986c). Mutagenic, carcinogenic, and teratogenic effects of PCB exposure have been observed, with mutagenic activity appearing to increase with increasing chlorination of the PCB molecule (Eisler,1986c). In a study of multigenerational effects on mink fed the same Saginaw Bay PCB-contaminated carp, Restum *et al.* (1998) determined that after 18 months of exposure to PCBs, mink with a dietary intake of 0.5 mg/kg PCB had significantly decreased kit survival.

REFERENCES

Agency for Toxic Substances and Disease Registry (ATSDR). 1997. Toxicological Profile for Lead. Division of Toxicology/Toxicology Information Branch, Atlanta, Georgia. December 1997. (available online at http://www.atsdr.cdc.gov/toxpro2.html)

ATSDR. 1998. Toxicological Profile for Chlorinated Dibenzo-p-dioxins. Division of Toxicology/Toxicology Information Branch, Atlanta, Georgia. December 1998. (available online at http://www.atsdr.cdc.gov/toxpro2.html)

Anderson, D. W. and J. J. Hickey, 1972. Eggshell changes in certain North American birds. Proc. Int. Ornith. Con. 15:514B540.

Ankley, G. T., D. Tillitt, and J. Giesy, 1989. Maternal transfer of bioactive polychlorinated aromatic hydrocarbons in spawning chinook salmon (Oncorhynchus tshawytscha). Marine Environmental Research. 28:231B234.

Ankley, G. T., D. T. Tillitt, J. P. Giesy, P. D. Jones, and D. A. Verbrugge, 1991. Bioassay derived 2,3,7,8 tetrachlorodibenzo p dioxin equivalents in PCB containing extracts from the flesh and eggs of Lake Michigan chinook salmon (Oncorhynchus tshawytscha) and possible implications for reproduction. Canadian Journal of Fisheries and Aquatic Sciences. 48:1685B1690.

Aulerich, R. J. and R. K. Ringer, 1970. Some effects of chlorinated hydrocarbon pesticides on mink. American Fur Breeder. 46:10B11.

Aulerich, R. J., R. K. Ringer, and S. Iwamoto, 1973. Reproductive failure and mortality in mink fed on Great Lakes fish. J. Reprod. Fert. Suppl. 19:365B376.

Aulerich, R. J. and R. K. Ringer, 1977. Current status of PCB toxicity to mink, and effect on their reproduction. Archives of Environmental Contamination and Toxicology. 6(2B3):279B292.

Bacci, E., M.J. Carejera, C. Gaggi, G. Chemello, D. Calamatir, and M. Vighi. 1992. Chlorinated dioxins: volatilization from soils and bioconcentration in plant leaves. Bulletin of Environmental Contamination and Toxicology 48:401-408.

Beyer, W.N., and E.J. Cromartie. 1987. A survey of Pb, Cu, Zn, Cd, Cr, As, and Se in earthworms and soil from diverse sites. Environmental Monitoring and Assessment 8:27-36.

Bleavins, M. R., R. J. Aulerich, and R. K. Ringer, 1980. Polychlorinated biphenyls (Aroclors 1016 and 1242): Effects on survival and reproduction in mink and ferrets. Archives of Environmental Contamination and Toxicology. 9(5):627B635.

Dilworth, T., G. Keith, P. Pearce, and L. Reynolds, 1972. DDE and eggshell thickness in New Brunswick woodcock. Journal of Wildlife Management. 36(4):1186B1193.

Duby, R. T., H. F. Travis, and C. E. Terrill, 1971. Uterotropic activity of DDT in rats and mink and its influence on reproduction in the rat. Toxicology and Applied Pharmacology. 18:348B355.

Efroymson, R.A., M.E. Will, G.W. Suter II, and A.C. Wooten. 1997a. Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Terrestrial Plants: 1997 Revision. Oak Ridge National Laboratory, Oak Ridge, TN. 128 pp. ES/ER/TM-85/R3.

Efroymson, R.A., M.E Will, and G.W. Suter II. 1997b. Toxicological Benchmarks for Contaminants of Potential Concern for Effects on Soil and Litter Invertebrates and Heterotrophic Processes: 1997 Revision. Oak Ridge National Laboratory, Oak Ridge TN. ES/ER/TM-126/R2.

Eisler, R., 1985. Cadmium Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. Biological Report 85(1.2). United States Fish and Wildlife Service. July.

Eisler 1986a. Chromium Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. U. S. Fish and Wildlife Service, U.S. Department of the Interior. Biological Report 6.

Eisler, R., 1986b. Dioxin Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. Biological Report 85(1.8). United States Fish and Wildlife Service.

Eisler, R., 1986c. Polychlorinated Biphenyl Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. Biological Report 85(1.7). United States Fish and Wildlife Service, Laurel, Maryland. April.

Eisler, R., 1987a. Mercury Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. Biological Report 85(1.14). United States Fish and Wildlife Service.

Eisler, R. 1987b. Polycyclic aromatic hydrocarbon hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Fish Wildl. Serv. Biol. Rep. 85(1.11).

Eisler, R., 1988. Lead Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. Biological Report 85(1.10). United States Fish and Wildlife Service.

Eisler, R., 1989. Pentachlorophemol Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. Biological Report 85(1.17). United States Fish and Wildlife Service.

Eisler, R. 1993. Zinc hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Fish Wildl. Serv. Biol. Rep. 10.

Eisler, R., 1996. Silver Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. Biological Report 32. United States Fish and Wildlife Service.

Eisler, R. 1998. Copper hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Geological Survey, Biological Resources Division, Biological Science Report USGS/BRD/BSR—1998-0002.

Eisler, R. 2000. Polycyclic Aromatic Hydrocarbons. In: Handbook of Chemical Risk Assessment. Health Hazards to Humans, Plants, and Animals. Volume 2. Organics. Lewis Publishers.

Eisler, R. and A. A. Belisle, 1996. Planar PCB hazards to fish, wildlife, and invertebrates: A synoptic review. Contaminant Hazard Reviews. 31.

Elliott, J. E., R. J. Norstrom, and G. E. Smith, 1996. Patterns, trends, and toxicological significance of chlorinated hydrocarbon and mercury contaminants in bald eagle eggs from the Pacific coast of Canada, 1990B1994. Archives of Environmental Contamination and Toxicology. 31(3):354B67.

Frank, R. and M. V. H. Holdrinet, 1975. Residue of organochlorine compounds and mercury in birds= eggs from the Niagara Peninsula, Ontario. Archives of Environmental Contamination and Toxicology. 3(2):205B218.

Giesy, J. P., D. A. Verbrugge, R. A. Othout, W. W. Bowerman, M. A. Mora, P. D. Jones, J. L. Newsted, C. Vandervoort, S. N. Heaton, R. J. Aulerich, S. J. Bursian, J. P. Ludwig, G. A. Dawson, T. J. Kubiak, D. A. Best, and D. E. Tillitt, 1994. Contaminants in fishes from Great Lakes influenced sections and above dams of three Michigan rivers, II: Implications for health of mink. Archives of Environmental Contamination and Toxicology. 27(2):213B223.

Giesy, J. P., W. W. Bowerman, M. A. Mora, D. A. Verbrugge, R. A. Othoudt, J. L. Newsted, C. L. Summer, R. J. Aulerich, S. J. Bursian, J. P. Ludwig, G. A. Dawson, T. J. Kubiak, D. A. Best, and D. E. Tillitt, 1995. Contaminants in fishes from Great Lakes influenced sections and above dams of three Michigan rivers: III. Implications for health of bald eagles. Archives of Environmental Contamination and Toxicology. 29(3):309B21.

Giesy, J., J. Ludwig, and D. Tillitt, 1994a. Deformities in birds of the Great Lakes Region: Assigning causality. Environmental Science & Technology. 28:128AB135A.

Giesy, J. P., J. P. Ludwig, and D. E. Tillitt, 1994b. Chapter 9: Dioxins, dibenzofurans, PCBs and colonial, fish eating water birds. In: Dioxins and Health. A. Schecter (Ed.) Plenum Press, New York, New York. p. 249B307.

Giesy, J. P., D. A. Verbrugge, R. A. Othout, W. W. Bowerman, M. A. Mora, P. D. Jones, J. L. Newsted, C. Vandervoort, S. N. Heaton, R. J. Aulerich, S. J. Bursian, J. P. Ludwig, M. E. Ludwig, G. A. Dawson, T. J. Kubiak, D. A. Best, and D. E. Tillitt, 1994c. Contaminants in fishes from Great Lakes influenced sections and above dams of three Michigan rivers, I: Concentrations of organochlorine insecticides, polychlorinated biphenyls, dioxin equivalents, and mercury. Archives of Environmental Contamination and Toxicology. 27(2):202B212

Gilbertson, M., T. Kubiak, J. Ludwig, and G. Fox, 1991. Great Lakes embryo mortality, edema, and deformities syndrome (GLEMEDS) in colonial fish eating birds: Similarity to chick edema disease. Journal of Toxicology and Environmental Health. 33(4):455B520.

Heaton, S. N., S. J. Bursian, J. P. Giesy, D. E. Tillitt, J. A. Render, P. D. Jones, D. A. Verbrugge, T. J. Kubiak, and R. J. Aulerich, 1995a. Dietary exposure of mink to carp from Saginaw Bay, Michigan: 1. Effects on reproduction and survival, and the potential risks to wild mink populations. Archives of Environmental Contamination and Toxicology. 28(3):334B343.

Heaton, S. N., S. J. Bursian, J. P. Giesy, D. E. Tillitt, J. A. Render, P. D. Jones, D. A. Verbrugge, T. J. Kubiak, and R. J. Aulerich, 1995b. Dietary exposure of mink to carp from Saginaw Bay, Michigan: 2. Hematology and liver pathology. Archives of Environmental Contamination and Toxicology. 29(3):411B417

Irwin, R.J., M. VanMouwerik, L. Stevens, M.D. Seese, and W. Basham. 1997. Environmental Contaminants Encyclopedia. National Park Service, Water Resources Division, Fort Collins, Colorado. Distributed within the Federal Government as an Electronic Document (Projected public availability on the internet or NTIS: 1998).

Jensen, S., J. E. Kihlstrom, N. Olsson, C. Lundberg, and J. Orberg, 1977. Effects of PCB and DDT on mink (Mustela vison) during the reproductive season. Ambio. 6:239.

Kabata-Pendias, A., and H. Pendias. 1984. Trace Elements in Soils and Plants. CRC Press.

Kubiak, T. J., H. J. Harris, L. M. Smith, T. R. Schwartz, D. L. Stalling, J. A. Trick, L. Sileo, D. E. Docherty, and T. C. Erdman, 1989. Microcontaminants and reproductive impairment of the Forster=s tern on Green Bay, Lake Michigan - 1983. Archives of Environmental Contamination and Toxicology. 18(5):706B727.

Kubiak, T. J., H. J. Harris, L. M. Smith, T. R. Schwartz, D. L. Stalling, J. A. Trick, L. Sileo, D. E. Docherty, and T. C. Erdman, 1989. Microcontaminants and reproductive impairment of the Forster=s tern on Green Bay, Lake Michigan - 1983. Archives of Environmental Contamination and Toxicology. 18(5):706B727.

Kubiak, T. J. and D. A. Best, 1991. Wildlife Risks Associated with Passage of Contaminated, Anadromous Fish at Federal Energy Regulatory Commission Licensed Dams in Michigan. Contaminants Program, Division of Ecological Services, East Lansing, Michigan. August 16.

Ludke, J. L., E. F. Hill, and M. P. Dieter, 1975. Cholinesterase response and related mortality among birds fed cholinesterase inhibitors. Archives of Environmental Contamination and Toxicology. 3:1B21.

Lundholm, E., 1987. Thinning of eggshells of birds by DDE: Mode of action on the eggshell gland. Comp. Biochem. Physiol. 88C:1B22.

Mac, M. J., 1988. Toxic substances and survival of Lake Michigan salmonids: Field and laboratory approaches. In: Toxic Contaminants and Ecosystem Health: A Great Lakes Focus. M. Evans (Ed.) John Wiley and Sons, New York, New York. p. 389B401.

MacDonald, D.D, C.G. Ingersoll, and T.A. Berger. 2000. Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems. Archives of Environmental Contamination and Toxicology 39:20-31.

Major, M.A., D.H. Rosenblatt, and K.A. Bostian. 1991. The octanol/water partition coefficient of methylmercuric chloride and methylmercuric hydroxide in pure water and salt solutions. Environmental Toxicology and Chemistry 10:5-8.

Matilainen, T., M. Verta, M. Neimi, and A. Uusi0Rauva. 1991. Specific rates of net methylmercury production in lake sediments. Water, Air, and Soil Pollution. 56:595-605.

Mazet, J.A.K., I.A. Gardner, D.A. Jessup, L.J. Lowenstine. 2001. Effects of petroleum on mink applied as a model for reproductive success in sea otters. Journal of Wildlife Diseases 37(4):686-692.

McCrady, J.K., C. McFarlane, and L.K. Gander. 1990. The transport and fate of 2,3,7,8-TCDD in soybean and corn. Chemosphere 21(3):359-376.

McCrady, J.J., and S.P. Maggard. 1993. Uptake and photodegradation of 2,3,7,8-tetrachlorodibenzo-p-dioxin sorbed to grass foliage. Environmental Science and Technology 27:343-350.

McFarland, V. A. and J. U. Clarke, 1989. Environmental occurrence, abundance, and potential toxicity of polychlorinated biphenyl congeners: Considerations for a congener specific analysis. Environmental Health Perspectives. 81:225B39.

National Academy of Sciences (NAS). 1980. Mineral Tolerance of Domestic Animals. National Academy Press. Washington, D.C.

Proulx, G., D. V. Weseloh, J. E. Elliott, S. Teeple, P. A. Anghern, and P. Mineau, 1987. Organochlorine and PCB residues in Lake Erie mink populations. Bulletin of Environmental Contamination and Toxicology. 39(6):939B44.

Niimi, A. J., 1996. Chapter 5: PCBs in aquatic organisms. In: Environmental Contaminants in Wildlife. W. N. Beyer (Ed.) Lewis Publishers, Boca Raton, Florida. p. 117B152.

Reinecke, A.J., and G. Nash. 1984. Toxicity of 2,3,7,8-TCDD and short-term bioaccumulation by earthworms (Oligochaeta). Soil Biology and Biochemistry 16:45-49.

Restum, J. C., S. J. Bursian, J. P. Giesy, J. A. Render, W. G. Helferich, E. B. Shipp, and D. A. Verbrugge, 1998. Multigenerational study of the effects of consumption of PCB contaminated carp from Saginaw Bay, Lake Huron, on mink: 1. Effects on mink reproduction, kit growth and survival, and selected biological parameters. Journal of Toxicology and Environmental Health, Part A. 54:343B375.

Safe, S., 1990. Polychlorinated biphenyls (PCBs), dibenzo p dioxins (PCDDs), dibenzofurans (PCDFs), and related compounds: Environmental and mechanistic considerations which support the development of toxic equivalency factors (TEFs). Toxicology. 21(1):51B88.

Safe, S., 1991. Polychlorinated dibenzo p dioxins and related compounds: Sources, environmental distribution and risk assessment. Environmental Carcino. & Ecotox. Revs. C9(2):261B302.

Sample, B.E., D.M. Opresko, and G.W Suter II. 1996. Toxicological Benchmarks for Wildlife: 1996 Revision. Oak Ridge National Laboratory, Oak Ridge, TN. 227 pp. ES/ER/TM-86/R3.

Schultz, T.W. 1989. Nonpolar narcosis: A review of the mechanism of action for baseline aquatic toxicity. In: Aquatic Toxicology and Hazard Assessment: 12th volume, ASTM STP 1027. American Society of Testing and Materials. Philadelphia, Pennsylvania. Pp. 104-109.

Shain, W., B. Bush, and R. Seegal, 1991. Neurotoxicity of polychlorinated biphenyls: Structure activity relationship of individual congeners. Toxicology and Applied Pharmacology. 111:33B42.

Tillitt, D. E., G. T. Ankley, and J. P. Giesy, 1992. Polychlorinated biphenyl residues and egg mortality in double crested cormorants from the Great Lakes. Environmental Toxicology and Chemistry. 11:1281B1288.

Tillitt, D. E., R. W. Gale, J. C. Meadows, J. L. Zajicek, P. H. Peterman, S. N. Heaton, P. D. Jones, S. J. Bursian, T. J. Kubiak, J. P. Giesy, and R. J. Aulerich, 1996. Dietary exposure of mink to carp from Saginaw Bay: 3. Characterization of dietary exposure to planar halogenated hydrocarbons, dioxin equivalents, and biomagnification. Environmental Science & Technology. 30(1):283B291.

U.S. Environmental Protection Agency (U.S. EPA). 1999. Screening Level Ecological Risk Assessment Protocol for hazardous Waste Combustion Facilities. Office of Solid Waste and Emergency Response. EPA530-D-99-001A. August.

U.S. EPA. 2000a. Bioaccumulation Testing and Interpretation for the Purpose of Sediment Quality Assessment. Status and Needs. EPA 823-R-00-001. U.S. Environmental Protection Agency, Office of Water and Office of Solid Waste. February.

U.S. EPA. 2000b. Draft Exposure and Human Health Reassessment of 2,3,7,8-Tetrachlorodibenzo-p-dioxin (TCDD) and Related Compounds. NCEA Office of Research and Development, Washington, D.C. September 2000. EPA/600/P-00/001 Bb,Bc,Bd,Be,Bg.

U.S. EPA. 2003. Framework for Application of the Toxicity Equivalence Methodology for Polychlorinated Dioxins, Furans, and Biphenyls in Ecological Risk Assessment. Risk Assessment Forum, Washington, D.C. June 2003. EPA/630/P-03/002A.

Walker, M. K. and R. E. Peterson, 1991. Potencies of polychlorinated dibenzo p dioxin, dibenzofuran, and biphenyl congeners, relative to 2,3,7,8 tetrachlorodibenzo p dioxin, for producing early life stage mortality in rainbow trout (Oncorhynchus mykiss). Aquatic Toxicology. 21:219B238.

Walker, M. K., J. Lawrence C. Hufnagle, M. K. Clayton, and R. E. Peterson, 1991a. An egg injection method for assessing early life stage mortality of polychlorinated dibenzo p dioxins, dibenzofurans, and biphenyls in rainbow trout, (Oncorhynchus mykiss). Aquatic Toxicology. 2:15B38.

Walker, M. K., J. M. Spitsbergen, J. R. Olson, and R. E. Peterson, 1991b. 2,3,7,8 tetrachlorodibenzo p dioxin (TCDD) toxicity during early life stage development of

lake trout (Salvelinus namaycush). Canadian Journal of Fisheries and Aquatic Sciences. 48:875B883.

Walker, M. K., P. M. Cook, B. C. Butterworth, E. W. Zabel, and R. E. Peterson, 1996. Potency of a complex mixture of polychlorinated dibenzo p dioxin, dibenzofuran, and biphenyl congeners compared to 2,3,7,8 tetrachlorodibenzo p dioxin in causing fish early life stage mortality. Fundamental and Applied Toxicology. 30(2):178B86.

West, C. W., T. Ankley, J. W. Nichols, G. E. Elonen, and D. E. Nessa, 1997. Toxicity and bioaccumulation of 2,3,7,8-tetrachlorodibenzo-p-dioxin in long term tests with the freshwater benthic invertebrates Chironomus tentans and Lumbriculus variegatus. Environmental Toxicology and Chemistry. 16(6):1287B1294.

Williams, L. L. and J. P. Giesy, 1992. Relationships among concentrations of individual polychlorinated biphenyl (PCB) congeners, 2,3,7,8 tetrachlorodibenzo P dioxin equivalents (TCDD EQ), and rearing mortality of chinook salmon (Oncorhynchus tshawytscha) eggs from Lake Michigan. Journal of Great Lakes Research. 18(1):108B124.

Wren, C. D., 1991. Cause effect linkages between chemicals and populations of mink (Mustela vison) and otter (Lutra canadensis) in the Great Lakes Basin. Journal of Toxicology and Environmental Health. 33(4):549B585.

Zabel, E. W., P. M. Cook, and R. E. Peterson, 1995. Toxic equivalency factors of polychlorinated dibenzo p dioxin, dibenzofuran and biphenyl congeners based on early life stage mortality in rainbow trout (Oncorhynchus mykiss). Aquatic Toxicology (Amsterdam). 31(4):315B328.