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PHASE 1A ECOLOGICAL RISK ASSESSMENT
VOLUME 3

TASK 4, TASK 5, AND TASK 6 SUMMARY REPORTS

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HUNTERS POINT ANNEX
PHASE 1A ECOLOGICAL RISK ASSESSMENT

TASK 4 SUMMARY REPORT
TOXICOLOGICAL EFFECTS COMPILATION AND DATA EVALUATION

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PHASE 1A ECOLOGICAL RISK ASSESSMENT (ERA)

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TASK 4.0 TOXICOLOGICAL EFFECTS COMPILATION AND DATA EVALUATION HUNTERS POINT ANNEX PHASE 1A ECOLOGICAL RISK ASSESSMENT

The purpose of Task 4.0 is to compile and evaluate the available toxicological information for contaminants of potential concern (COPC) detected at Hunters Point Annex (HPA). This information will be used to (1) identify potentially toxic concentrations of chemicals present at HPA, (2) provide information about the appropriate biological endpoints for activities during subsequent phases of the ecological risk assessment, (3) provide a basis for an additional quantitative evaluation of chemical impacts in subsequent phases, and (4) identify data gaps to be filled in subsequent phases.

COPCs are often identified by comparing site concentrations with screening values, which are generally regulatory criteria or benchmark values. However, ecologically based regulatory criteria are only available for priority pollutants in surface water; sediment criteria or benchmark values are available for a smaller set of contaminants. No state or federal criteria or benchmark values are available for the protection of terrestrial receptors. This lack of criteria for terrestrial receptors results from the diversity in terms of receptor species, routes of exposure and sensitivity to toxins. Therefore, it is necessary to simultaneously evaluate many aspects of transport and toxicity of contaminants to fully evaluate ecological effects. For this ecological risk assessment, effects will be evaluated by comparing site concentrations to criteria and benchmark values for surface water and sediment, by evaluating transport and bioaccumulation through the food web, and by directly measuring toxicity using bioassays.

Task 4.0 has been divided into three subtasks: reviewing the transport, fate, and toxicological properties of contaminants found at HPA; reviewing state and federal ecological benchmark values for aquatic and sediment-dwelling organisms; and evaluating the available data on toxicological effects of HPA sediment and surface water on bioassay test organisms.

4.1 TOXICOLOGICAL PROFILE SUMMARIES

For each of the COPCs detected in the media tested at HPA, a detailed toxicological profile has been developed to provide a thorough understanding of the transport, fate, and toxic properties. Sources of information reviewed to develop these profiles included the published scientific literature, government

reports, databases, supporting material for regulatory criteria, and other available sources. Detailed toxicological profiles will be presented in the final ecological risk assessment report; this section presents a summary of properties for both terrestrial and aquatic media.

The profile summaries focus on a general discussion of migration pathways, chronic exposures, sublethal effects, and bioconcentration potentials for receptors that are most applicable and related to those at HPA. Appendix A presents tabular compilations of available data on toxic effects of COPCs for terrestrial plants, soil invertebrates, avian, and mammalian receptors. Available data on bioconcentration potential of COPCs in aquatic and terrestrial systems are provided in Appendix B. In addition, sediment and water quality criteria are presented in Section 4.2 of this report. A summary table of the lowest available effects level for each CPOC for various classes of receptors is presented at the end of Section 4.1.

4.1.1 Inorganic Chemicals

The current test method approved by the U.S. Environmental Protection Agency (EPA) for measuring concentrations of inorganic elements in media involves quantifying the total concentration. However, the toxicity and bioaccumulation of certain elements depend on the bioavailability of the element in the medium (Adriano 1986). For instance, in soil, elements that are dissolved into solution in moist soil or held onto chemical exchange sites are mobile and available for uptake by plants or other organisms. Elements that are incorporated into mineral matrices, coprecipitated with other chemicals in the soil or sediment, or incorporated into biological material are generally immobile, although they may become mobile and available with time. Bioavailability may also be significantly influenced by the characteristics of the medium (Adriano 1986). The factors governing the equilibrium of trace elements between solution and solid phases are complex. The primary factors include pH, water content, percent of organic matter, cation exchange capacity, and the amount of clay in relation to the amount of silt and sand. Generally, increasing pH and percent of organic matter decreases the bioavailability of trace elements in the soil, and elements are more available in sandy soils than in soils with a high clay content. Studies on the uptake of soil metals by earthworms (Ma 1982; Ma and others 1983) have shown that pH significantly influences the rate of uptake of cadmium, lead, and zinc, but not that of copper. In the soils studied by Ma, lead was the only metal significantly influenced by percent organic matter. The influence of these factors on trace element bioavailability

may depend on properties of the element or the oxidation state of the element in the soil. In addition, interactions with other trace elements can affect bioavailability and uptake.

The availability of an element for gastrointestinal absorption is also dependent upon the chemical form. Elements in food and water are most often found in the ionic form. The element is usually a free ion in water and exists more often in a complex form in food. Thus, elements incorporated into food items or soil or sediment may be less available for gastrointestinal absorption than elements in water, although this varies with the individual element. Experimental studies on the toxicology of metals have generally used soluble salts for dietary and drinking water exposures, and consequently may overestimate the toxicity of elements that may be actually bound onto food items or soil or sediment in the natural diet. Toxicity of inorganic contaminants also varies with a variety of biotic factors, including the type of receptor, the life stage, feeding behavior, prey type, and habitat utilization.

The following sections discuss in more detail the fate and transport, toxicity, and bioaccumulative properties of elements found in terrestrial and aquatic media at HPA.

4.1.1.1 Arsenic

Although arsenic occurs naturally in air, water, soil, and all living tissues and may be an essential nutrient, it is also a teratogen and a carcinogen that can traverse placental barriers and produce fetal death and malformations in many species of mammals (Eisler 1988a). Arsenic exists in four oxidation states in inorganic and organic forms, and changes its chemical form and state readily as a result of chemically and microbiologically mediated reactions. Its bioavailability and toxicity are modified by numerous biotic and abiotic factors that include the physical and chemical forms of arsenic, the route of exposure, the dose, and the species of organism. In general, inorganic arsenic compounds are more toxic than organic arsenic compounds, and trivalent species are more toxic than pentavalent species. Cells take up arsenic through an active transport system normally used in phosphate transport. Arsenic is bioconcentrated, but not biomagnified, by organisms (Eisler 1988a).

In water, arsenic occurs in both inorganic and organic forms and in dissolved and gaseous states (EPA 1980, as cited in Eisler 1988a), and may undergo complex transformations that include

biotransformation (Callahan and others 1979, as cited in ATSDR 1993b). Arsenic in water primarily exists as a dissolved ionic species (Maher 1985a, as cited in Eisler 1988a). The formation of inorganic pentavalent arsenic, the most common species in water, is favored under conditions of high dissolved oxygen, basic pH, high Eh, and reduced content of organic material; the reverse conditions usually favor the formation of trivalent arsenics and arsenic sulfides (NRCC 1978, Pershagen and Vahter 1979, and EPA 1980; as cited in Eisler 1988a).

Pentavalent arsenics are readily sorbed by colloidal humic material and may also coprecipitate with or adsorb onto hydrous iron oxides and form insoluble precipitates with calcium, sulfur, aluminum, and barium compounds (EPA 1980, as cited in Eisler 1988a). In reducing environments, such as sediments, pentavalent arsenics become more strongly adsorbed to sediments than other forms of arsenic (EPA 1980, as cited in Eisler 1988a).

Arsenic in soil also can undergo many complex transformations. Pentavalent arsenics predominate in aerobic soils, and trivalent arsenics predominate in slightly reduced or temporarily flooded soils (EPA 1982a, as cited in ATSDR 1993b).

Arsenic Bioconcentration. Bioconcentration factors (BCF) experimentally determined for arsenic in aquatic organisms are relatively low. Marine phytoplankton have the ability to accumulate high concentrations of inorganic arsenics and transform them to methylated arsenics that are later efficiently transferred in the food chain (Eisler 1988a). While arsenic does bioconcentrate, it does not appear to be biomagnified through the food chain (Callahan and others 1979, and EPA 1982a and 1983e; as cited in ATSDR 1993b).

Arsenic Effects on Aquatic Biota. The toxicity of arsenic in the water column to marine organisms depends on the form (oxidation state and counter ion) of the arsenic. Both carcinogenic and mutagenic effects have been reported (Eisler 1988a). Acute toxicity, as well as sublethal effects, have been observed in fish and invertebrates (NOAA 1991), although, acute toxicity can differ among even taxonomically similar species. Temperature, pH, speciation, and many other factors can also influence the toxicity; however, inorganic arsenics are generally more toxic than organic forms of arsenic (Eisler 1988a). Adverse effects to marine organisms have been observed at concentrations in water of 0.0019 mg/L to 500 mg/L.

Arsenic Effects on Terrestrial Biota. Terrestrial plants may accumulate arsenic by root uptake from the soil or by adsorption of airborne arsenic deposited on the leaves, and certain species may accumulate substantial levels (EPA 1982a, as cited in ATSDR 1993b). A significant depression in crop yields for many plants was evident at soil arsenic concentrations of 3 to 28 mg/L of water-soluble arsenic and 25 to 85 mg/kg of total arsenic (NRCC 1978, as cited in Eisler 1988a). Soil microorganisms are capable of tolerating and metabolizing relatively high concentrations of arsenic (Wang and others 1984, as cited in Eisler 1988a). However, arsenic may result in reduced growth and metabolism of soil microbiota, diminished numbers of bacteria and protozoans, and the absence of earthworms (NRCC 1978, as cited in Eisler 1988a).

As with other organisms, the effects of arsenic on mammals varies with the species, the exposure route, and the physical and chemical form of the arsenic. Many mammals can rapidly excrete ingested inorganic arsenic (Eisler 1988a); however, arsenic does get distributed to most tissues including the placenta and fetus. Chronic exposure has resulted in reductions in litter size of mice.

4.1.1.2 Beryllium

In most types of soil, beryllium is expected to be tightly sorbed onto clay particles and to remain precipitated as insoluble complexes at higher pHs (Fishbein 1981, as cited in ATSDR 1993d). Beryllium is expected to have limited mobility in soil, although its mobility may increase as a result of formation of soluble hydroxide complexes in soils of higher pHs (Callahan and others 1979, as cited in ATSDR 1993d).

In aquatic environments, most beryllium is usually present either sorbed to suspended matter or to sediment, particularly to clay particles, rather than in a dissolved form. Beryllium may precipitate into sediment as a result of formation of insoluble complexes and is usually sorbed onto clay particles in sediment. A high percentage of beryllium is expected to be immobile in water as a result of this association with sediment particles, although at a high pH, the formation of water-soluble complexes with hydroxide ions may increase the solubility and mobility of beryllium (Callahan and others 1979, as cited in ATSDR 1993d).

Beryllium Bioconcentration. Bioconcentration of beryllium in fish to high level is not likely because of the low uptake of beryllium from water by aquatic animals. No evidence of significant biomagnification of beryllium in food chains was found (Fishbein 1981, as cited in ATSDR 1993d).

Beryllium Effects on Aquatic Biota. No data were found on the toxic effects of beryllium on aquatic biota.

Beryllium Effects on Terrestrial Biota. No data were found on the effects of beryllium on terrestrial invertebrates, plants, and birds. A lifetime no observed adverse effect levels (NOAEL) of 0.95 mg beryllium/kg-day for ingestion of drinking water in mice has been calculated (Schroeder and Mitchener 1975).

4.1.1.3 Cadmium

Cadmium is more mobile in aquatic environments than most other heavy metals. In natural waters, most cadmium will exist as the hydrated ion and as a complex with humic substances. Precipitation and sorption to mineral surfaces and organic materials are the most important removal processes for cadmium compounds. Sediment bacteria may also assist in the partitioning of cadmium from water to sediments (Burke and Pfister 1988, as cited in ATSDR 1993e). Cadmium may redissolve from sediments under varying ambient conditions of pH, salinity, and redox potential (Callahan and others 1979 and Feijtel others 1988, as cited in ASTDR 1993e).

Transformation processes for cadmium in soil are mediated by sorption from and desorption to water, processes influenced by the cation exchange capacity, pH, oxygen content, and the presence of clay minerals, carbonate minerals, oxides, and organic matter (McComish and Ong 1988, as cited in ATSDR 1993e). Cadmium in soils may leach into water, especially under acidic conditions (Callahan and others 1979 and Elinder 1985a, as cited in ATSDR 1993e). Cadmium-containing soil particles may also be entrained into the air or eroded into water (EPA 1985a, as cited in ATSDR 1993e).

Cadmium has no essential biological function, and, in sufficient concentrations, is highly toxic to plants and animals. It is a known teratogen and carcinogen and a suspected mutagen, and has severe deleterious sublethal effects on wildlife (Eisler 1985a).

Cadmium Bioconcentration. Aquatic and terrestrial organisms at all trophic levels bioaccumulate cadmium, although the evidence suggests that only lower trophic levels exhibit biomagnification (Eisler 1985a). Bioconcentration in fish depends on the pH and the organic content of the water (John and others 1987, as cited in ATSDR 1993e). Although some data indicate increased cadmium absorption in animals at the top of the food chain, the data available on biomagnification are inconclusive (Beyer 1986 Gochfeld and Burger 1982, as cited in ATSDR 1993e).

Cadmium Effects on Aquatic Biota. Cadmium has been shown to be highly toxic in aquatic environments and has been implicated as the cause of severe deleterious effects on fish and wildlife including acute mortality, reduced growth, and inhibited reproduction (Eisler 1985a). Marine organisms are more resistant than freshwater biota to cadmium. Decapod crustaceans, the most sensitive of the marine organisms, have been shown to die at concentrations of cadmium in seawater ranging from 14.8 to 420 ppb. Sublethal effects to marine animals, which included decreased growth, respiratory disruptions, altered enzyme levels, and abnormal muscular contractions, have been recorded at cadmium concentrations of 0.5 to 10 ppb (Eisler 1985a).

Cadmium Effects on Terrestrial Biota. Cadmium is known to be toxic to plants at much lower soil concentrations than other heavy metals such as copper, lead, and zinc. Although birds appear to be relatively resistant to the toxic effects of cadmium, sublethal effects in birds include growth retardation, nephrotoxicity, anemia, damage to the testicles and absorptive epithelium of the duodenum, reduced egg production, and effects on calcium absorption (Hammons and others 1978, Scheuhammer 1987). In mammals, prenatal exposure to cadmium has fetotoxic effects, usually reduced fetal or pup weights.

4.1.1.4 Chromium

Little is known about the relationship between concentrations of total chromium in a given environment and biological effects on the organisms living there. Depending on the physical and chemical state of the chromium, the same elemental concentration has a wide variety of mobilities and reactivities and thus has different effects (Steven and others 1976, as cited in Eisler 1986a). In general, hexavalent chromium is more toxic than trivalent chromium.

In both freshwater and marine environments, hydrolysis and precipitation are the most important processes that determine the fate and effects of chromium, whereas adsorption and bioaccumulation are relatively minor (Ecological Analysts 1981, as cited in Eisler 1986a). Most of the chromium released into water will ultimately be deposited in the sediment. Soluble chromium generally accounts for a very small percentage of the total chromium. Most of the soluble chromium is present as hexavalent chromium and, to a lesser extent in seawater, trivalent chromium complexes.

Chromium in soil is present mainly as insoluble oxide (EPA 1984, as cited in ATSDR 1993f), and is therefore not expected to be mobile in soil. Mobility of soluble chromium in soil depends upon the sorption characteristics of the soil. A smaller percentage of total chromium in soil exists as soluble hexavalent chromium and trivalent chromium, which are mobile in soil. Surface runoff can transport both soluble and precipitated chromium from soils to surface waters, and soluble and unadsorbed hexavalent chromium may leach to groundwater. The leachability of hexavalent chromium in soil increases as soil pH increases, while lower pHs in soil or in acid rain may facilitate leaching of acid-soluble trivalent or hexavalent chromium compounds in soil (Calder 1988, as cited in ATSDR 1993f).

In general, chromium is mutagenic, carcinogenic, and teratogenic to a wide variety of organisms tested under laboratory conditions, with hexavalent chromium having the greatest biological activity. Although little toxicity data are available for other chromium forms, they are generally thought to be less toxic (Gale 1978, as cited in Eisler 1986a).

Chromium Bioconcentration. Although high accumulations of chromium have been recorded among organisms from the lower trophic levels, there is little evidence of biomagnification through food chains (Osterberg and others 1964, as cited in Eisler 1986a). Studies of food chains including phytoplankton, brine shrimp, postlarval fish and adult fish have shown that when chromium was successively transferred through each of the four trophic levels, concentrations declined after each transfer. Bioaccumulation of chromium from soil to aboveground plant parts in unlikely because most of the chromium taken in is retained in the roots, and only a small fraction is translocated to the aboveground part of edible plants (Cary 1982, WHO 1988, and Petruzzelli and others 1987; as cited in ATSDR 1993f). As in aquatic food chains, there is no indication of chromium biomagnification in terrestrial food chains (Cary 1982, as cited in ATSDR 1993f).

Chromium Effects on Aquatic Biota. Acute and chronic toxicity to aquatic and marine organisms have been tested with trivalent and hexavalent chromium. Acute toxicity of hexavalent chromium to saltwater animals occurs at concentrations ranging from 2,000 to 105,000 ppb (NOAA 1991), whereas acute toxicity of trivalent chromium has been observed at concentrations of from 10,300 to 31,500 ppb (EPA 1986, as cited in NOAA 1991). Chromium accumulates heavily in sediments; bioaccumulation by clams has been observed at sediment concentrations as low as 150 ppm (EPA 1992b).

Chromium Effects on Terrestrial Biota. Data on toxicity of chromium to terrestrial invertebrates and plants are sparse. Chromium has been observed to be teratogenic in birds. In young American black ducks (*Anas rubripes*) fed chromium-contaminated diets, growth patterns were altered and survival was reduced (Haseltine and others 1985, as cited in Eisler 1986a). Hexavalent chromium is mutagenic and carcinogenic in mammals (Leonard and Lawerys 1980 and Norseth 1981, as cited in Eisler 1986a). When 32 mg trivalent chromium/kg-day was fed to rats for a 24- to 90-day period, results included diarrhea, tough coats, and reproductive sterility (EPA 1990). However, equivalent doses of hexavalent chromium produced no adverse reproductive effects in rats (Ivanokovic and Preussman 1975). In general, hexavalent chromium is more toxic than the trivalent form.

4.1.1.5 Copper

In general, copper is very mobile under oxidizing and acidic conditions and immobile in organic-rich environments. Most copper deposited in soil is strongly adsorbed to organic matter and remains in the upper few centimeters of the soil. Adsorption increases with pH and also with higher organic matter content. However, copper can leach from soils, especially from sandy soils, under acidic conditions (USPHS 1990).

In aquatic systems, copper binds primarily to organic matter and forms complexes with both organic and inorganic ligands (mainly with calcium carbonate) that settle out in sediments (Kirk-Othmer 1965). In reducing, acidic, or richly organic bed sediments, copper complexes can dissolve, resulting in remobilization of sorbed copper.

Copper Bioconcentration. Although copper is an essential nutrient, biomagnification is not a significant fate process for copper. Accumulated copper is stored in the liver, kidney, bone marrow, and hair (Hammond and Beliles 1980, as cited in Talmage and Walton 1991). Bioaccumulation has been demonstrated for fish, with BCFs for fish from the tens to the hundreds and BCFs for mollusks up to 30,000 (Perwack and others 1980; Chapman and others 1968; Raymont 1972). Bioaccumulation does not appear to occur in terrestrial ecosystems. Copper is highly bioavailable in the San Francisco estuary and can build up in aquatic organisms that, when consumed, may pose a threat to higher levels in the food chain (Merian 1991).

Copper Effects on Aquatic Biota. Copper is highly toxic in aquatic environments and is considered a pollutant of great concern (EPA 1992a). Saltwater organisms are acutely sensitive to copper in water concentrations ranging from 5.8 ppb to 600 ppb.

Copper Effects on Terrestrial Biota. Based on yield reductions of 14 to 28 percent in agronomic and grassland parts, 100 mg/kg total copper is considered to be a threshold concentration for toxicity to plants and soil invertebrates (EPA 1987 and Bengtsson and Tranvik 1989, as cited in ICA 1992). In a study measuring growth, no growth retardation was observed in chickens fed 40.5 mg/kg-day copper sulfate for 4 weeks (NAS 1980). Increased fetal mortality and developmental abnormalities were observed in pregnant mice fed copper sulfate.

4.1.1.6 Cyanide

Many chemical forms of cyanide are present in the environment, including free cyanide, metallocyanide complexes, and synthetic organocyanides, also known as nitrils. But only free cyanide (HCN and CN^-) is the primary toxic agent, regardless of origin. Cyanide is a potent and rapid asphyxiant, inducing tissue anoxia. All available evidence suggests that cyanides are neither mutagenic, teratogenic, nor carcinogenic. Moreover, there are no reports of cyanide biomagnification or cycling in living organisms, probably because of its rapid detoxification. Cyanide seldom persists in surface waters and soils owing to complexation or sedimentation, microbial metabolism, and loss from volatilization (Eisler 1991).

Cyanide occurs most commonly as free hydrocyanic acid in water, although it can also occur as the cyanide ion, alkali metal cyanides (for example, sodium cyanide and potassium cyanide), relatively stable metallocyanide complexes, moderately stable metallocyanide complexes, or easily decomposable metallocyanide complexes (for example, zinc cyanide). The environmental fate of these compounds varies (Callahan and others 1979, as cited in ATSDR 1993g). Alkali metal salts are very soluble in water and dissociate into their respective anions and cations when released into water. Volatilization rates increase with decreasing pH and with increased aeration (Ludzack and others 1951 and Raef and others 1977, as cited in ATSDR 1993g). Biodegradation is also a significant fate process in natural water systems. A number of pure cultures of microorganisms degrade low concentrations of cyanide under both aerobic and anaerobic conditions (Callahan and others 1978 and Towill and others 1978, as cited in ATSDR 1993g).

Additional data are necessary to assess the significance of cyanide sorption to suspended solids and sediments in water. The existing data indicate that the adsorption of hydrogen cyanide to suspended solids and sediment is not significant. However, soluble metal cyanides may show stronger adsorption than hydrogen cyanide. The extent of adsorption increases with decreasing pH and increases with increasing iron oxide, clay, and organic material contents of water. Adsorption is probably insignificant even for metal cyanides when compared to volatilization and biodegradation (Callahan and others 1979, as cited in ATSDR 1993g).

Volatilization of hydrogen cyanide would be a significant loss mechanism for cyanides from soil surfaces at a pH less than 9.2. Although cyanide has a low soil sorption capability (Callahan and others 1979), it is usually not detected in groundwater probably because of fixation by trace metals through complexation or transformation by soil microorganisms (Towill and others 1978, as cited in ATSDR 1993g). Leaching may occur at soil concentrations that are toxic to soil invertebrates (EPA 1984, as cited in ATSDR 1993g). Cyanide present in low concentrations in soil would biodegrade under aerobic conditions with the initial formation of ammonia, which would be converted to nitrate in the presence of denitrifying bacteria. Under anaerobic conditions, cyanides will denitrify to gaseous nitrogen (Towill and others 1978, as cited in ATSDR 1993g).

Cyanide Bioconcentration. The simple metal cyanides and hydrogen cyanide do not bioconcentrate in aquatic organisms (ATSDR 1993g). There is no evidence of biomagnification of cyanides in the food chain (Towill and others 1978, as cited in ATSDR 1993g).

Cyanide Effects on Aquatic Biota. Cyanide toxicity to aquatic organisms results from the presence of free cyanide (hydrogen cyanide or the cyanide anion) derived from dissociation, photodecomposition, hydrolysis, and biodegradation. Most metallocyanide complexes are not very toxic. Free cyanide is a more reliable index of the toxicity to aquatic life than total cyanide because total cyanide can include organic cyanides and relatively stable metallocyanide complexes. The EPA water quality criteria were developed on the basis of free cyanide measurements, although EPA acknowledges that no approved methods for such measurements are currently available (EPA 1985).

Cyanide Effects on Terrestrial Biota. In higher plants elevated cyanide concentrations can inhibit respiration, but the sublethal effects of cyanide on plants vary (Eisler 1991). At lower concentrations, effects include inhibition of germination and growth and chromosomal aberration. On the other hand, cyanide enhances seed germination and can be metabolized in some plants (Towill and others 1978 and Solomonson 1981, as cited in Eisler 1991).

Bird species sensitivity to cyanide is related to diet. As judged by acute oral dosages lethal to 50 percent of the birds (LD_{50}), birds that feed predominantly on flesh, such as vultures, kestrels, and owls, were more sensitive to cyanide than were species that feed mainly on plant material, with the possible exception of the mallard (Wiemeyer and others 1986, as cited in Eisler 1991).

Although much of the toxicological interest in cyanide relating to mammals has focused on its rapid lethal action, most researchers agree on the following (Towill and others 1978, EPA 1980, Way 1984, and Ballantyne and Marrs 1987a; as cited in Eisler 1991). First, cyanide has low persistence in the environment and is not accumulated or stored in any mammal studied. Second, biomagnification in food webs has not been reported, possibly because of rapid detoxification of sublethal doses by most species and death at higher doses. Third, cyanide has unusually low chronic toxicity, but chronic intoxication exists, and in some cases, can be incapacitating. Despite the high lethality of large single doses, repeated sublethal doses seldom result in cumulative adverse effects and can be tolerated by many species for long periods.

4.1.1.7 Lead

The fate of lead in soil is affected by the exchange adsorption at mineral interfaces, the precipitation of sparingly soluble solid forms of the compound, and the formation of relatively stable organic-metal complexed or chelates with soil organic matter. These processes depend on such factors as soil pH, organic content, ion-exchange characteristics, and the amount of lead in the soil (NSF 1977, as cited in ATSDR 1993i). Most lead is retained strongly in soil, and very little is transported into surface water or groundwater (EPA 1986a and NSF 1977, as cited in ATSDR 1993i). Lead is strongly sorbed to organic matter in soil, and although it is not subject to leaching, it may enter surface waters as a result of erosion of lead-containing soil particulates (Olson and Skogerboe 1975, as cited in ATSDR 1993i).

The movement of lead from soil to groundwater by leaching is very slow under most natural conditions except for acidic situations (NSF 1977, as cited in ATSDR 1993i). The conditions that induce leaching are the presence of lead in soil at concentrations that either approach or exceed the cation exchange capacity of the soil, the presence of materials in soil that are capable of forming soluble chelates with lead, and a decrease in the pH of the leaching solution (for example, acid rain) (NSF 1977, as cited in ATSDR 1993i).

The chemistry of lead in aqueous solution is highly complex because of its various chemical forms. Lead has a tendency to form compounds of low solubility with major anions found in natural waters. In most surface waters and groundwaters, the concentration of dissolved lead is low because the lead forms compounds with anions in the water such as hydroxides, carbonates, sulfates, and phosphates that have low water solubilities and will precipitate out of the water column (Mundell and others 1989, as cited ATSDR 1993i). A significant fraction of lead carried by river water is expected to be in an undissolved form, which can consist of colloidal particles or larger undissolved particles of lead carbonate, lead oxide, lead hydroxide, or other lead compounds incorporated in other components of surface particulate matters from runoff. Lead may occur either as sorbed ions or surface coatings on sediment mineral particles, or it may be carried as a part of suspended living on nonliving organic matter in water (Getz and others 1977, as cited in ATSDR 1993i).

Lead has been known for centuries as a poison, and environmental pollution from lead is extensive and severe (Eisler 1988b). Lead is a mutagen and a teratogen, and when it is absorbed in excessive amounts, has carcinogenic or cocarcinogenic properties, impairs reproduction and liver and thyroid functions, and interferes with resistance to infectious disease (EPA 1979, as cited in Eisler 1988b). Several ecotoxicological properties of lead are agreed upon by investigators. First, lead is a ubiquitous characteristic trace constituent in rocks, soils, water, plants, animals, and air. Second, lead is neither essential nor beneficial to living organisms, and all data show that its metabolic effects are adverse. Third, lead is toxic in most of its chemical forms and can be incorporated into the body via inhalation, ingestion, dermal absorption, and placental transfer. Fourth, lead is an accumulative metabolic poison that affects behavior, as well as the hematopoietic, vascular, nervous, renal, and reproductive systems (Eisler 1988b).

Lead Bioconcentration. Plants and animals may bioconcentrate lead, but biomagnification has not been detected. Several studies have shown that invertebrates can accumulate lead in their tissues, although they show variability in tendencies to accumulate lead.

Organolead compounds, such as trialkyl and tetraalkyl lead compounds, are more toxic than inorganic forms and have been shown to bioconcentrate in aquatic organisms as do inorganic lead compounds. High accumulations of lead from ambient seawater by marine plants are well documented. Although lead is concentrated by biota from water, there is no convincing evidence that it is transferred through food chains (Wong and others 1978, EPA 1979, Branica and Konrad 1980, and Settle and Patterson 1980; as cited in Eisler 1988b). Lead concentrations tend to decrease with increasing trophic level in both detritus-based and grazing aquatic, including marine, food chains (Wong and others 1978 and Stewart and Schulz-Baldes 1976, as cited in Eisler 1988b).

Lead Effects on Aquatic Biota. Lead is toxic to all phyla of aquatic biota, though effects are modified significantly by various biotic (species and physiological state) and abiotic variables. Fish continuously exposed to toxic concentrations of waterborne lead show various signs of lead poisoning: spinal curvature; anemia; degeneration of the caudal fin; destruction of spinal neurons; reduced ability to swim against a current; destruction of the respiratory epithelium; muscular atrophy; paralysis; renal pathology; growth inhibition; retardation of sexual maturity; testicular and ovarian histopathology; and death (Eisler 1988b).

Lead Effects on Terrestrial Biota. Lead is not essential for plants, and excessive amounts can cause growth inhibition, as well as reduced photosynthesis, mitosis, and water absorption (Demayo and others 1982, as cited in Eisler 1988b). A concentration of 12,800 mg lead/kg is associated with reductions in natural populations of decomposers, such as fungi, earthworms, and arthropods. The poisoning of decomposers may disrupt nutrient cycling, reduce the number of invertebrates available to other wildlife for food, and contribute to food chain contamination (Beyer and Anderson 1985, as cited in Eisler 1988b). Absorbed lead produces a variety of effects in avian species leading to death, including damage to the nervous system, muscular paralysis, inhibition of heme synthesis, and damage to kidneys and liver (Mudge 1983, as cited in Eisler 1988b). Sublethal lead exposure may also have adverse effects of reproduction in some avian species by decreasing plasma calcium, inhibition of growth, and reduced hatchability of chicks.

Significant differences occur between mammalian species in response to lead exposure. Effects on mammals are more pronounced following exposure to organoleads than inorganic compounds. Younger developmental stages are more sensitive to lead. Effects of lead include growth retardation, delays in maturation, and reduced body weight.

4.1.1.8 Mercury

Mercury is a toxicologically important heavy metal that is persistent in the environment, can concentrate and magnify in food chains, is slow to depurate, and is neurotoxic to all organisms (Peterle 1991). Mercury is a known mutagen, teratogen, and carcinogen, and its toxicity and environmental effects vary with its form, dose, route of ingestion, species, sex, age, and general condition of the organism (Eisler 1987a, Fimreite 1979). Methyl, phenyl, alkyl, and ethyl mercury are the most deleterious forms. Biotransformation is an important fate process in the environmental partitioning of mercury. Under favorable conditions, microorganisms in soil and sediment can convert various forms of mercury to methyl mercury, which is more available for uptake by various organisms and for transport in the food chain and is more soluble and mobile than inorganic forms (Peterle 1991).

Depending on the pH, salt content, and composition of the soil, mercury usually forms various complexes with chloride and hydroxide ions in the soil (ATSDR 1992b). Methyl mercury is water

soluble and becomes part of the water column, making it readily available for uptake by aquatic organisms (Peterle 1991).

Mercury Bioconcentration. Mercury has a high potential for bioaccumulation and biomagnification (Eisler 1987a). Methylated mercury is the form most readily accumulated (Kramer and Neidhart 1975). Biomagnification of methyl mercury has been documented for both aquatic and terrestrial food chains (Eisler 1987a). Concentrations of methyl mercury in high-trophic level carnivorous fish have reportedly been biomagnified on the order of 10,000 to 100,000 times those concentrations found in ambient waters (Callahan and others 1979). The accumulation of mercury by aquatic organisms is enhanced at elevated water temperatures, reduced water salinity or hardness, reduced water pH, increased age of the organism, and reduced organic matter in the medium.

Mercury transfer and biomagnification through mammalian food chains is well documented. Elemental mercury is extremely lipophilic and distributes to most body tissues, especially to the kidney. Fetal accumulation is also common, and the developing organism of all species investigated is the most susceptible to mercury toxicity (Eisler 1987a). Carnivores appear to bioconcentrate mercury to a greater extent than do herbivores (Eisler 1987a).

Mercury Effects on Aquatic Biota. Mercury at comparatively low concentrations adversely affects the reproduction, growth, behavior, metabolism, blood chemistry, osmoregulation, and oxygen exchange of marine and freshwater organisms (Eisler 1987a). Methyl mercury appears to be the most toxic form of mercury (MacDonald 1993).

Mercury Effects on Terrestrial Biota. Plants take up mercury from the soil in relatively insignificant amounts because roots appear to act as a barrier to uptake and tend to accumulate the small amount of mercury that may be absorbed. However, mercury compounds applied to other parts of plants appear to be readily absorbed and translocated (Adriano 1986).

Sublethal effects of mercury on birds, administered by a variety of routes, included adverse effects on growth, development, reproduction, blood and tissue chemistry, metabolism, and behavior (Eisler 1987a). In birds, however, species differences in intestinal absorption of mercury have been demonstrated. Frequently, reproductive effects are noted at low doses long before overt signs of

toxicity are apparent in adults (Scheuhammer 1987). Significant reproductive effects of chronic dietary inorganic mercury exposure in birds include delayed testicular development, altered mating behavior, reduced fertility, reduced survivability and growth in young, and gonadal atresia. Mercury is also transferred to the egg in avian species, where it has adverse effects on the developing embryo (Peterle 1991).

4.1.1.9 Molybdenum

Molybdenum is found in all living organisms and is considered to be an essential or beneficial nutrient. Molybdenum chemistry is complex and not adequately understood. In water at pH greater than seven, molybdenum exists primarily as the molybdate ion; at pH less than seven, various polymeric compounds are formed, including the paramolybdate ion (Busev 1969, as cited in Eisler 1989). In soils molybdate is sorbed most readily to alkaline, high calcium, high chloride soils. Retention is lowest in low pH, low sulfate soils (Smith and others 1987, as cited in Eisler 1989).

Molybdenum interacts toxicologically with other trace elements, especially copper and inorganic sulfates. In mammals molybdenum can protect against poisoning by copper, mercury, and probably other metals, and may have anticarcinogenic properties. However, ruminants are sensitive to molybdenum poisoning, especially when accompanied by a deficiency in copper or inorganic sulfates.

Molybdenum Bioconcentration. No information on the bioaccumulative potential of molybdenum was found.

Molybdenum Effects on Aquatic Biota. No data were found on the effects of molybdenum on aquatic invertebrates and fish.

Molybdenum Effects on Terrestrial Biota. Plants readily accumulate molybdate except in soils having low pH, high sulfate, and low phosphate, and high organic matter content (Gupta and Lipsett 1981, as cited in Eisler 1989). No toxicity of molybdenum to field-grown crops has been observed, although forages containing 10 to 20 mg/kg dry weight (dw) molybdenum are considered toxic to cattle and sheep (Soon and Bates 1985, as cited in Eisler 1989).

Data are lacking on the effects of molybdenum on avian wildlife under controlled conditions. All studies on birds have been restricted to domestic poultry, which appear to be relatively resistant to molybdenum exposure. Data on the effects of molybdenum on mammalian wildlife are scarce. Almost all studies conducted to date on molybdenum effects under controlled conditions have been on livestock, especially cattle and sheep. Molybdenum is essential in mammalian diets and can protect against poisoning by copper or mercury and may be useful in controlling cancer, although molybdenum poisoning in ruminants has been observed in livestock (Eisler 1989). All evidence indicates that other mammals besides cattle and sheep are comparatively tolerant of high dietary intakes of molybdenum (Underwood 1971, Buck 1978, Chappell and others 1979, and Friberg and Lener 1986; as cited in Eisler 1989).

4.1.1.10 Nickel

Nickel is strongly adsorbed by soil, although to a lesser degree than lead, copper, and zinc (Rai and Zachara 1984, as cited in ATSDR 1993j). There are many particle-sorbing species of nickel in soil, and many factors affect the extent to which nickels is adsorbed, making nickel adsorption site-specific. Amorphous oxides of iron and manganese, and to a lesser extent, clay minerals, are the most important adsorbents in soil. In alkaline soils, adsorption may be irreversible, thus limiting nickel's availability and mobility in these soils. Cations such as calcium and magnesium reportedly reduce adsorption as a result of competition for binding sites, whereas anions like sulfate reduce adsorption because of complexation. Nickel adsorption depends strongly on pH (Rai and Zachara 1984, as cited in ATSDR 1993j).

Nickel Bioconcentration. Nickel is not accumulated in significant amounts by aquatic organisms (Birge and Black 1980 and Callahan and others 1979, as cited in ATSDR 1993j). Recent studies of nickel levels in voles and rabbits living on sludge-amended land did not indicate any accumulation of nickel in these herbivores or in the plants that they fed on (Alberici and others 1989 and Dressler and others 1986, as cited in ATSDR 1993j).

Nickel Effects on Aquatic Biota. Nickel is both a carcinogen and a mutagen in the aquatic environment (EPA 1992a). Acute toxicity to organisms occurs at concentrations as low as 151.7 ppb

in saltwater. Chronic effects can occur at concentrations of 141 ppb or greater in seawater; the toxicity is influenced greatly by the water hardness and salinity (NOAA 1991).

Nickel Effects on Terrestrial Biota. Because nickel was not detected in soils at HPA, data on the effects of nickel on terrestrial biota were not compiled.

4.1.1.11 Selenium

Selenium is nutritionally important as an essential trace element, but harmful at slightly higher concentrations (Eisler 1985b). Results of laboratory studies and field investigations with fish, mammals, and birds have led to general agreement that elevated concentrations of selenium in diet or water were associated with reproductive abnormalities and growth retardation. Not as extensively documented are reports of selenium-induced chromosomal aberrations, intestinal lesions, shifts in species composition of freshwater algal communities, swimming impairment of protozoans, and behavioral modifications (Eisler 1985b).

In aerobic waters, selenium is present in the selenite (H_2SO_3, $HSeO_3^{-1}$, SeO_3^{-2}) or selenate (H_2SeO_4, $HSeO_4^{-}$, SeO_4^{-2}) quadravalent or hexavalent oxidation states. These chemical species are very soluble, and most of the selenium discharged into the aquatic environment is probably transported in these forms to the oceans (EPA 1979). Selenium has a sorptive affinity for hydrous metals oxides, clays, and organic materials. Sorption by bed sediments or suspended solids can result in enrichment of selenium concentrations in the bed sediments. Sorption or precipitation with hydrous iron oxides is probably the major control on mobility of selenium in aerobic waters (EPA 1979).

Selenium can be methylated by a variety of organisms, including benthic microflora. In a reducing environment, hydrogen selenide may be formed. Both the methylated forms and hydrogen selenide are volatile and may escape to the atmosphere. Formation of volatile selenium compounds in the sediments can remobilize sorbed selenium (EPA 1979).

Selenium Bioconcentration. Accumulation of selenium by aquatic organisms is highly variable (Eisler 1985). Accumulation is modified by water temperature, age of the organism, organ or tissue

specificity, mode of administration, and other factors (Eisler 1985). The soluble forms of selenium, principally sodium selenate, can be accumulated in plants.

Selenium Effects on Aquatic Biota. Selenium is teratogenic, and its toxicity depends greatly on its chemical form (EPA 1992a). It has been suggested that selenite is more toxic than selenate and is preferentially concentrated over selenate by mussels (EPA 1990, as cited in Eisler 1985b).

Selenium Effects on Terrestrial Biota. Selenium exposure in the diet or drinking water of avian species is associated with reproductive abnormalities, congenital malformations, selective bioaccumulation, and growth retardation (Eisler 1985). Sublethal effects of selenium on mammals include reproductive abnormalities such as congenital malformations; reduced numbers of young in litters; high mortality of young; infertility among surviving young in rats, mice, swine, and cattle; and intestinal lesions (Harr 1978 and NRC 1983, as cited in Eisler 1985b).

4.1.1.12 Silver

Silver occurs naturally in the environment in the form of silver nitrate, silver chloride, silver sulfide, or silver oxide (USPHS 1990). It tends to form complexes with inorganic chemicals and humic substances in soils. Consequently, the mobility of silver in soils is affected by drainage, oxidation-reduction potential and pH conditions, and the presence of organic matter. Silver tends to be mobile in well-drained soils. Organic matter complexes with silver and causes it to become immobile. Since silver is toxic to soil microorganisms and inhibits bacterial enzymes, biotransformation is not expected to be a significant process (Domsch 1984, as cited in ATSDR 1990c).

The transport and partitioning of silver in surface waters and soils is influenced by the particular form of the compound. The major forms of silver in water include the monovalent ion as sulfate, bicarbonate, or sulfate salts; more complex ions with chlorides and sulfates; and an integral part of, or adsorbed onto, particulate matter and aquatic biota (Boyle 1968, as cited in ATSDR 1989). Sorption is the dominant process leading to the partitioning of silver in sediments (Callahan and others 1979, as cited in ATSDR 1990c), and pH and oxidation-reduction conditions affect sorption (Anderson and others 1973, as cited in ATSDR 1990c). When decaying animal and plant material is

abundant, silver strongly precipitates as the sulfide or combines with humic materials (Smith and Carson 1977, as cited in ATSDR 1990c).

Silver Bioconcentration. There is no evidence that silver is biomagnified in terrestrial animals, although this may occur in some aquatic invertebrates (Adriano 1986). Silver accumulation in marine algae appears to be from adsorption rather than uptake.

Silver Effects on Aquatic Biota. Silver is one of the most hazardous trace elements to aquatic species (EPA 1992a). Available data indicate that chronic toxicity to freshwater organisms may occur at concentrations in water as low as 0.12 ppb and that concentrations in seawater should not exceed 2.3 ppb at any time (NOAA 1991).

Silver Effects on Terrestrial Biota. Silver is not considered to be highly phytotoxic; even large amounts in soil appear to have little or no effect on plant growth. No data were found on the effects of silver on avian species. Little information is available on the effects of silver to mammals; a decrease in weight gain was observed in rats exposed to 222 mg/kg-day silver in drinking water (Matuk and others 1989, as cited in ATSDR 1990c).

4.1.1.13 Tributyltin

Mariners have commonly used tributyltin (TBT) based antifoulant paints to slow the growth of barnacles, seaweeds, and tubeworms on ship hulls. A 1985 study estimated that up to 300,000 pounds of TBT were used annually in antifoulant paints used in the United States. Elemental or inorganic tin appears to cause negligible toxicological effects. However, organotins are toxic and bioaccumulating. Of the organotins, TBT is the most toxic to aquatic life. TBT has an increased fat solubility, which allows greater penetration of biological membranes. This may be one reason for its greater toxicity (Champ and Pugh 1987).

Because TBT is a hydrophobic compound, the equilibrium partitioning of TBT between the dissolved phase and the particle-sorbed phase is strongly shifted toward the sorbed phase. However, the sorption process is reversible, and contaminated sediments may act as a source for dissolved TBT in the water column (Unger and others 1987). Once introduced to the water column, TBT will partition

into the suspended and bottom sediment. TBT will be removed from the system by volatilization, degradation, and bulk transport via currents and tidal action. During degradation, organotins are converted to inorganic tin by the progressive removal of organic groups. UV-photolysis and biological degradation are the major environmental causes of this conversion (Thain and others 1987), and dibutyltin is the major degradation product (Lee and others 1987).

Tributyltin Bioconcentration. Because of TBT's hydrophobic properties, it tends to bioconcentrate. A study by Humphrey and Hope (1987) measured tissue and ambient water concentrations of TBT; the BCF from water was 23,000 for shellfish and 141,000 for finfish.

Tributyltin Effects on Aquatic Biota. Sensitivity varies among aquatic species; gastropods and bivalves are the most susceptible, followed by crustaceans, algae, and then fish. TBT is slow acting, and it effects development and reproductive functions. A species of Pacific oyster (*Crassostrea gigas*) has often been used in TBT studies because of its sensitivity to TBT. One documented effect of exposure to TBT on *C. gigas* is shell thickening.

Tributyltin Effects on Terrestrial Biota. No data were found on the effects on TBT on avian species. Few data are available on mammals; the oral LD_{50} of TBT chloride is 122 mg/kg for rats.

4.1.1.14 Zinc

Zinc is an essential trace element for all living organisms. Zinc deficiency is a problem for both plants and animals, and manifestations of zinc poisoning in animals include growth retardation, testicular atrophy, skin changes, and poor appetite (Prasad 1979, as cited in Eisler 1993).

Most of the zinc introduced into aquatic environments is eventually partitioned into the sediments. Zinc release from sediment is enhanced under conditions of high dissolved oxygen, low salinity, and low pH. Dissolved zinc usually consists of the toxic hydrated zinc ion and various organic and inorganic complexes. These hydrated ions and other toxic species greatly affect aquatic organisms under conditions of comparatively low pH, low alkalinity, low dissolved oxygen, and elevated temperatures.

Zinc Bioconcentration. BCF for zinc accumulation from the medium varied widely between and within species of aquatic organisms (Eisler 1993). In marine environments the most effective zinc accumulators included red and brown algae, ostreid and crassotreid oysters, and scallops. Vegetation studies indicate little uptake of zinc from soil. Zinc accumulates in invertebrates and may be passed on to higher trophic level consumers. However, total body burdens must be used with some caution in analyzing trophic transfer because studies show that bony structures can act as long-term repositories for zinc (Macapinlac and others 1966), and bone is rarely consumed or digested by predators of small mammals.

Zinc Effects on Aquatic Biota. Significant adverse effects of zinc on growth, survival, and reproduction occur in representative sensitive species of aquatic plants, protozoans, sponges, mollusks, crustaceans, echinoderms, fish, and amphibians at nominal water concentrations between 10 and 25 ppb (Eisler 1993). Biological effects have not been observed in association with zinc concentrations of about 50 ppm or less in sediments (NOAA 1991). In general, zinc was more toxic to embryos and juveniles than to adults, to starved animals, at elevated temperatures, in the presence of cadmium and mercury, in the absence of chelating agents, at reduced salinity, at decreased water hardness and alkalinity, and at low dissolved oxygen concentrations (Eisler 1993).

Zinc Effects on Terrestrial Biota. Although zinc is one of the essential nutrients for plant growth, plants require it only in small amounts. Zinc concentrations in acidic soils that produced toxicity to corn ranged from 450 to 1,400 mg/kg available zinc; toxicity to cowpeas ranged from 180 to 700 mg/kg available zinc (Gall and Barnette 1940, as cited in Adriano 1986).

Feeding studies indicate that decreased weight gain is first observed at zinc dietary concentrations of 270 mg/kg in Japanese quail, 800 mg/kg in chickens, and 4,000 mg/kg in turkeys (NAS 1980). Zinc is relatively nontoxic in mammals; however, excessive zinc intake adversely affects survival of all tested mammals and produces a wide variety of neurological, hematological, immunological, hepatic, renal, cardiovascular, developmental, and genotoxic effects (PHS 1989, as cited in Eisler 1993).

4.1.2 Volatile and Semivolatile Organic Compounds

The following sections present information on the toxicology and fate and transport of the important volatile and semivolatile organic compounds at HPA.

4.1.2.1 Benzene

The high volatility and water solubility of benzene are the physical properties with the greatest influence on environmental transport and partitioning (Mackay and Leinonen 1975, as cited in ATSDR 1993c). Benzene released to the soil partitions to the atmosphere through rapid volatilization, to surface water in runoff, and to groundwater as a result of leaching. Estimates of the organic carbon partitioning coefficient (Koc) range from 60 to 83 (Karickhoff 1981 and Kenaga 1980, as cited in ATSDR 1993c) and 31.7 to 143 (Hansch 1985, as cited in HSDB 1994), classifying benzene as highly mobile.

A model developed to predict the environmental fate of benzene following leakage from an underground storage tank indicated that most (67 percent) of the benzene would volatilize from the shallow, sandy soil within 17 months. Of the remaining benzene, 29 percent would leach to groundwater, 3 percent would remain in the soil, and 1 percent would be degraded (Tucker and others 1986, as cited in ATSDR 1993c). Benzene also volatilizes rapidly from water and is not expected to significantly adsorb to sediment or hydrolyze. A half-life of 3.1 days (summer) to 23 days (spring) was reported for seawater in a mesocosm simulating Narragansett Bay, Rhode Island, containing the associated planktonic and microbial communities (Wakeham and others 1983, as cited in HSDB 1994).

Benzene biodegrades in both surface water and groundwater. Benzene biodegradation is much slower under anaerobic conditions than under aerobic conditions, making biodegradation in any groundwater other than shallow aerobic groundwater unlikely. After 40 weeks benzene concentrations had been reduced by 72 percent under laboratory anaerobic conditions, and after 120 weeks, 99 percent degradation had taken place (Wilson and others 1972, as cited in ATSDR 1993c). Benzene is biodegraded in soil under aerobic conditions. However, one study concluded that only small numbers of the specific organisms that degrade benzene were present in the soil.

Benzene Bioconcentration. The octanol-water partitioning coefficient (Kow) for a given compound is a measure of that compound's propensity to be associated with water or with octanol. Because the Kow is an indicator of a compound's fat-solubility, it is also a strong predictor of the compound's tendency to bioaccumulate. Some calculations of the Kow for benzene range from 2.13 to 2.15 (HSDB 1990 and Gossett and others 1983, as cited in ATSDR 1993c). Studies suggest that bioaccumulation in aquatic food chains is not significant (ATSDR 1993c).

Benzene Effects on Aquatic Biota. No data were found on the effects of benzene on aquatic invertebrates and fish.

Benzene Effects on Terrestrial Biota. Since benzene exists primarily in the vapor phase, root uptake is not expected to be a major source of vegetative contamination, making air to leaf transfer the major pathway of vegetative contamination (Hattemer-Frey and others 1990, as cited in ATSDR 1993c). No data on the effects of benzene exposure on birds or mammals were found.

4.1.2.2 Polynuclear Aromatic Hydrocarbons

Polynuclear aromatic hydrocarbons (PAH) contain substituted benzene rings. There are thousands of PAH compounds, each differing in the number and position of the aromatic rings and in the position of substituents on the basic ring system. The physical and chemical properties of PAHs vary with molecular weight. With increasing molecular weight, aqueous solubility decreases, and the logarithmic transformations of Kow values (log Kow) increases (Eisler 1987b).

In water, PAHs may either evaporate, disperse in the water column, become incorporated into bottom sediments, concentrate in aquatic biota, or experience chemical oxidation and biodegradation (Suess 1976, as cited in Eisler 1987b). The most influential degradative processes for PAHs in aquatic systems are photooxidation, chemical oxidation, and biological transformation by bacteria and animals (Neff 1979, as cited in Eisler 1987b). Most PAHs in aquatic environments are associated with particulate materials; only about 33 percent are present in dissolved form (Lee and Grant 1981, as cited in Eisler 1987b). PAHs in aquatic sediments degrade very slowly in the absence of penetrating radiation and oxygen (Suess 1976, as cited in Eisler 1987b), and may persist indefinitely in oxygen-poor basins or in anoxic sediments (Neff 1979, as cited in Eisler 1987b).

PAHs cause a wide variety of adverse biological effects in numerous organisms under laboratory conditions, including effects on survival, growth, metabolism, and especially tumor formation. The higher molecular weight PAHs are known to be carcinogenic, mutagenic, and teratogenic to a wide variety of organisms, including fish and other aquatic life, amphibians, birds, and mammals. The lower molecular weight PAHs are generally not carcinogenic, but are more acutely toxic than their higher molecular weight relatives (Eisler 1987b). PAHs also strongly induce biotransforming enzymes, including the enzyme component cytochrome P-448 in the liver, and suppress the immune system. Immunosuppression often results in decreased responses to mitogens in mammals, causing increased susceptibility to disease. Inter- and intraspecies responses to PAHs are quite variable and are modified by interaction with other inorganic and organic compounds, including other PAHs (Eisler 1987b). In general, PAHs show little tendency to biomagnify in food chains, despite their high lipid solubility, probably because most PAHs are rapidly metabolized.

PAH Bioconcentration. Authorities generally agree that most species of aquatic organisms studied rapidly accumulate PAHs from low concentrations in the ambient medium, although uptake is highly species-specific. BCFs tend to increase with increasing molecular weight, Kow, dissolved organic carbon content, and lipid content of the organism (Eisler 1987b). Contaminated sediments play an important role in PAH uptake kinetics in aquatic ecosystems.

PAH Effects on Aquatic Biota. PAHs vary substantially in their toxicity to aquatic organisms. In general, toxicity increases with increasing molecular weight and with increasing alkyl substitution on the aromatic ring. Toxicity appears to be most pronounced in crustaceans (Neff 1979, as cited in Eisler 1987b). PAHs are believed to cause liver tumors in a variety of fish. A positive correlation has been established between sediment PAH levels and prevalence of liver lesions in English sole and in brown bullheads (Eisler 1987b).

PAH Effects on Terrestrial Biota. Plants can absorb PAHs from soils through their roots and translocate them to other plant parts. However, most plants appear to metabolize benzo(a)pyrene and possibly other PAHs.

Information on the effects of PAH exposure on avian wildlife is limited. PAHs cause embryotoxicity to mallard eggs when applied externally. The most embryotoxic PAH was

7,12-dimethylbenz(a)anthracene, followed by chrysene. Several investigations have suggested that the presence of PAHs in petroleum, including benzo(a)pyrene and chrysene, significantly enhances the overall embryotoxicity in avian species, and that the relatively small percent of the aromatic hydrocarbons contributed by PAHs in petroleum may confer much of the adverse biological effects reported after eggs have been exposed to microliter quantities of polluting oils (Hoffman and Gay 1981 and Albers 1983, as cited in Eisler 1987b).

Numerous PAH compounds are distinct in their ability to produce tumors in skin and in most epithelial tissues of practically all mammal species tested. Malignancies are often induced by acute exposures to microgram quantities. Dietary benzo(a)pyrene leads to leukemia, lung adenoma, and stomach tumors in mice. Acute and chronic exposure to various carcinogenic PAHs have resulted in destruction of hematopoietic and lymphoid tissues, ovotoxicity, antispermatogonic effects, adrenal necrosis, and changes in the intestinal and respiratory epithelia. There is a scarcity of data available on the toxicological properties of PAHs that are not demonstrably carcinogenic to mammals (EPA 1980 and Lee and Grant 1981, as cited in Eisler 1987b).

4.1.2.3 Total Petroleum Hydrocarbons as Diesel (Fuel Oil Number Two)

The transport and dispersion of fuel oils are dependent upon the water solubility of the aliphatic and aromatic hydrocarbon fractions. Larger aliphatic hydrocarbons (greater than C_9 chain length), such as fuel oil number two, may be sorbed to organic particles in water or soil. Koc measures a compound's propensity to be associated with organic carbon and describes the equilibrium between water and organic carbon. Logarithmic transformations of Koc values (log Koc) calculated for fuel oil number two range from 3.0 to 6.7 (Air Force 1989, as cited in ATSDR 1993h). Aromatic hydrocarbons will be dissolved in the aqueous phase in both soil and water and may undergo some volatilization. Kow values calculated for fuel oil number two range from 3.3 to 7.06 (Air Force 1989, as cited in ATSDR 1993h).

In aquatic systems evidence shows that the aliphatic fractions of fuel oil can be transported to the microlayer at the air-water interface or to the sediment on suspended particulate matter. The surface microlayer interface may be composed of different fuel oil hydrocarbons than the subsurface waters. When fuel oil number two was added below the surface of a marine ecosystem, the microlayer was

more enriched than the subsurface water for both saturated and unsaturated hydrocarbons. There was a significantly greater proportion of total saturates compared with aromatics. The hydrocarbons in the microlayer were predominantly higher molecular weight hydrocarbons greater than C_{18}. This indicates that all of the hydrocarbons are transported from the water column to the microlayer and weathered by evaporation once they are in the microlayer (Gearing and Gearing 1982b, as cited in ATSDR 1993h).

A spill of fuel oil number two to waters off Massachusetts showed that the oil was detected at distant monitoring locations 46 months after the spill, although the greatest concentration in the sediments remained in the vicinity of the original spill. Oil continued to move in pulses from the more polluted areas to less polluted areas for several years after the spill, with much of the oil being exhumed from sediments in the shallower waters as a result of storm action (EPA 1981, as cited in ATSDR 1993h). Movement of the oil to adjacent salt marshes indicated that the oil was sorbed to the anoxic sediment where it persisted (Burns and Teal, 1979, as cited in ATSDR 1993h). Biodegradation and dissipation resulted in the decrease of n- and branched alkanes from the surface sediment within 4 years. Even 8 years after the spill, some sediments contained over 1,200 ppm petroleum hydrocarbons with naphthalene and heavier aromatics expected to persist for many more years (Burns and Teal 1979, as cited in ATSDR 1993h).

Fuel oils can undergo a variety of degradation processes. The photooxidation of fuel oil number two in water is quite rapid in sunlight. Biodegradation of fuel oils is dependent on the degradation of the various hydrocarbon fractions present in the oils. The relative order for biodegradation of the hydrocarbon fractions from the most readily degraded to the least is as follows: n-alkanes, iso-calkanes, olefins, low molecular weight aromatics (at nontoxic concentrations), PAHs, and cycloalkanes (Bartha and Atlas 1977 and Edgerton and others 1987, as cited in ATSDR 1993h). A mixed culture of estuarine bacteria was observed to degrade fuel oil number two by 55 percent in 28 days (Walker and others 1976, as cited in ATSDR 1993h). Biodegradation of fuel oils in sediments is inhibited under anaerobic conditions (Bartha and Atlas 1977, as cited in ATSDR 1993h).

Microbial degradation in soils is greatest for the aromatic fractions of fuel oils, and the biodegradation of the aliphatic hydrocarbons decreases with increasing carbon chain length. Evaporation is the primary fate process for these aliphatics (Air Force 1989, as cited in ATSDR 1993h). Application of

diesel oil to soil at 1 percent or 10 percent showed that based on carbon dioxide evolution, degradation did occur. After 12 weeks, the application of 1 percent diesel oil was degraded by 45 percent, whereas the 10 percent application showed that only 10 percent of each oil was degraded in this time.

TPH-Diesel Bioconcentration. Aquatic organisms are known to bioconcentrate hydrocarbons. Adding fuel oil number two containing 38 percent aromatics to a commercial shrimp pond indicated that the concentrations in marine organisms are more closely correlated with water concentrations than with sediment concentrations of aromatic hydrocarbons (Cox and others 1975, as cited in ATSDR 1993h).

Organism-specific differences show that organisms in close contact with the sediment had relatively higher concentrations of lower molecular weight napthalenes, as these compounds are more readily retained in the sediment than in the water column. Similarly, organisms that lived in the water column had fewer napthalenes. These results indicate that fuel oil number two degradation products are taken up by benthic organisms, that they may be selectively retained in both sediments and aquatic organisms, and may thus enter the food chain (Farrington and others 1982b, as cited in ATSDR 1993h).

TPH-Diesel Effects on Aquatic Biota. The State of Alaska has determined a criterion for total hydrocarbons of 15 μg/L for the protection of aquatic life in marine ecosystems (State of Alaska 1989, as cited in ATSDR 1993h).

TPH-Diesel Effects on Terrestrial Biota. No data were found on the effects of fuel oil number two on terrestrial invertebrates, plans, and mammals. Behavioral changes in mallard ducklings were evident when they were exposed to larvicide and fuel oil during incubation (Albers and Heinz 1983, as cited in Peterle 1991).

4.1.3 Organochlorine Compounds

The following sections present information on the toxicology and fate and transport of important organochlorine compounds, including pesticides and polychlorinated biphenyls (PCB).

4.1.3.1 Polychlorinated Biphenyl (PCB)

Although currently banned from use and production in the United States, PCBs are still ubiquitous in the environment. PCBs are mixtures of different congeners of chlorobiphenyl, and the relative importance of environmental fate mechanisms depends upon the degree of chlorination. In general the persistence of the PCB increases with the degree of chlorination. Mono- through trichlorinated biphenyls (Aroclors 1221 and 1232) biodegrade relatively rapidly, tetrachlorinated biphenyls (Aroclors 1016 and 1242) biodegrade slowly, and higher chlorinated biphenyls (Aroclors 1248, 1254, and 1260) are resistant to biodegradation. Although biodegradation of higher chlorinated congeners may occur very slowly, no other degradation mechanisms have been shown to be important in natural water and soil systems (EPA 1987, Callahan and others 1979, Swackhamer and others 1986, and Baker and others 1985; as cited in HSDB 1994).

PCBs tightly adsorb to soil, and adsorption generally increases with the degree of chlorination. Koc values ranging from 510 to 13,300,000, mostly above 5,000, for a variety of PCBs have been reported (Sklarew and others 1987, as cited in HSDB 1994). PCBs generally do not leach significantly in aqueous soil systems, but the lower chlorination PCBs have a greater ability to leach than the higher chlorinated PCBs. However, in the presence of organic solvent, PCBs may leach quite rapidly through soil. Vapor loss of PCBs from soil surfaces is an important fate mechanism, with the rate decreasing with increasing chlorination (EPA 1987, Callahan and others 1979, Swackhamer and others 1986, and Baker and others 1985; as cited in HSDB 1994).

When released to water, PCBs can volatilize relatively rapidly; however, adsorption to sediment and suspended matter is a more influential fate process. Although adsorption can immobilize PCBs for long periods of time, eventual resolution can occur. Lower chlorinated PCBs are more water soluble than their higher chlorinated counterparts (EPA 1987, Callahan and others 1979, Swackhamer and others 1986, and Baker and others 1985; as cited in HSDB 1994).

PCB Bioconcentration. PCBs bioconcentrate significantly. All PCB congeners are highly lipophilic and most are readily distributed to all tissues. PCBs are cleared from tissues at very different rates, with metabolism varying widely and depending upon the animal species and the degree and positions of chlorination. PCBs that are not metabolized concentrate in the fatty tissue. The ability of animals

to metabolize PCBs increases in the following order: fish, birds, and mammals (Matthews and others 1984, as cited in HSDB 1994).

In studies of marine food chains, biomagnification becomes increasingly important with high trophic level carnivores, especially piscivores, such as gulls and pelicans, but is less important at lower trophic levels. PCB body burdens in marine organisms, especially benthic organisms, were directly related to the log of the PCB concentration in sediments (Shaw and Connell 1982, as cited in Eisler 1986b).

PCB Effects on Aquatic Biota. Documented effects of exposure to PCBs in aquatic organisms include decreased growth, reproductive toxicity, mutagenicity, histopathology, and a variety of biochemical perturbations (Eisler 1986b). Reproductive toxicity has been reported for several aquatic species including threshold values beyond which reduced survival of developing eggs be expected have been observed as 0.12 mg PCBs/kg fresh weight (fw) in ovaries of Baltic flounder, mortality in eggs and fry of Atlantic salmon, and reduced fertilization success and lowered survival of sea urchin eggs (Eisler 1986).

PCB Effects on Terrestrial Biota. Common soil alga exhibited depressed photosynthesis and growth inhibition when exposed to concentrations of PCBs as low as 1 μg/L and mortality when exposed to 50 μg/L. Very little information exists on the effects of PCBs on terrestrial invertebrates.

Generally, avian species are less susceptible to the acute toxic effects of PCBs than mammals. The dietary LD_{50} for various bird species ranged from 604 to 6,000 mg Aroclor/kg diet (Eisler 1986b). PCBs disrupt normal patterns of growth, reproduction, metabolism, and behavior in birds. In general, accumulation of PCBs is rapid and depuration is slow. Diet is an important route of PCB accumulation; concentrations in liver were highest in birds that feed on fish, followed by species that feed on small birds and mammals, worms and insects, and lowest in herbivorous species (NAS 1979, as cited in Eisler 1986b). PCBs adversely affect reproductive behavior and success in birds. Effects include reproductive impairment, reduced hatchability, and embryonic chromosomal aberrations (Peakall and others 1972, as cited in Eisler 1986b).

Documented effects of PCBs in mammals include reproductive failure, physiological effects, altered behavior, and mutagenic, carcinogenic, and teratogenic effects (Eisler 1986b). The most consistent pathological changes occurring in mammals after exposure to PCBs are in the liver. PCBs are transferred through lactation and placenta from adult to offspring.

4.1.3.2 Chlordane

Chlordane is a broad-spectrum, organochlorine pesticide consisting of approximately 45 components. Historically used to control soil invertebrate pests, chlordane also adversely affected nontarget species. Although federal law currently prohibits use of chlordane in the United States, persistent chlordane residues and metabolites from past use continue to pose an ecological threat (Eisler 1990). Chlordane and its metabolites do not degrade to any significant extent either biotically or abiotically in either aquatic or terrestrial ecosystems. Chlordane in water does not undergo considerable hydrolysis, oxidation, or photolysis, and biodegradation is minimal (Gore and others 1971, as cited in HSDB 1994). Because of chlordane's low water solubility, high lipid solubility, low vapor pressure, and strong tendency to adsorb to soil and sediment particles, it persists in soil and sediments for extended periods (EPA 1988, as cited in Eisler 1990). Adsorption of chlordane to sediment is expected to be a major aquatic fate process (Callahan and others 1979, as cited in HSDB 1994).

Based on field tests, soil column leaching tests, and Koc estimation, chlordane is expected to be generally immobile in soil (Cohen 1986, as cited in HSDB 1994). Surface-applied chlordane usually remains in the top 20 centimeters of most soils. Little biodegradation occurs in soil; chlordane residues have been detected in soils 14 years (EPA 1988, as cited in Eisler 1990) and 20 years (Beeman and Matsumura 1981, as cited in ATSDR 1993) after application. Volatilization is the most important pathway for chlordane release from the surface of soils (Beeman and Matsumura 1981, as cited in ATSDR 1993), and its vapor toxicity can be persistent and extreme.

Chlordane is a nerve stimulant whose physiological target sites are nerve and muscle membranes (Greenhalgh 1986, as cited in Eisler 1990). Warm-blooded animals readily absorb chlordane through dermal contact, ingestion, and inhalation; residues concentrate in the fat, liver, kidney, brain, and muscle (WHO 1984, as cited in Eisler 1990). Organisms metabolize chlordane into several isomers, of which oxychlordane is the most toxic and persistent, even when compared to its parent chemicals

(WHO 1984, as cited in Eisler 1990; Kawano and others 1988). These isomers can persist in avian tissues for up to 35.4 years (Eliott and others 1988, as cited in Eisler 1990). Chlordane interacts with other chemicals to cause additive or synergistic effects, especially when combined with other organochlorines, such as endrin, methoxychlor, and aldrin in mice (Klassen and others 1986, as cited in Eisler 1990) and northern bobwhites (Ludke 1976, as cited in Eisler 1990).

Chlordane Bioconcentration. Chlordane is known to bioconcentrate, especially in long-lived carnivorous organisms having large amounts of body lipids (Eisler 1990). Chlordane bioconcentration has been shown to increase with trophic levels from zooplankton to marine mammals in marine ecosystems, causing biomagnification of total chlordanes through the food chain, with residues peaking in marine mammals (Kawano and others 1986). Chlordane biomagnification in carnivorous birds is also considerable.

Chlordane Effects on Aquatic Biota. Chlordane adversely affects survival, reproduction, and growth in aquatic invertebrates and fish. Reduced survival was measured in shrimp and crabs at concentrations in water between 0.2 and 3.0 μg/L chlordane. Sediment concentrations of 5.8 mg/kg chlordane were fatal to sandworms (*Nereis virens*) in 12 days (McLeese and others 1982, as cited in Eisler 1990).

Chlordane Effects on Terrestrial Biota. Residues of chlordane have been detected in vegetable crops grown in soils treated with chlordane (USDA 1965, as cited in HSDB 1994). Chlordane is toxic to target and nontarget terrestrial invertebrates. Lethal effects of chlordane in birds result primarily from chlordane metabolites, most notably oxychlordane and heptachlor epoxide; oxychlordane is the most toxic and persistent of the two (Stickle and others 1983, as cited in Eisler 1990). Reproductive impairment has been documented in waterfowl species from a wetland treated with chlordane. Chronic dietary chlordane adversely affects physiology, growth, and fertility of laboratory mammals, including enlargement and increased pathology of the liver in rats and mice. It also elevated residues in cow's milk, reduced sexual activity and litter viability in rats, depressed growth and delayed development, decreased immune competence, and decreased viability of offspring in mice (Talamantes and Jang 1977 and WHO 1984, as cited in Eisler 1990).

4.1.3.3 Dichlorodiphenyltrichloroethane and Its Congeners

Dichlorodiphenyltrichloroethane (DDT) is a broad-spectrum organochlorine pesticide whose large-scale use began in 1944 and ceased in 1972 following a ban on general use in the United States. DDT has many environmental congeners, two of the most important of which are dichlorodiphenyldichloroethylene (DDE, also used as a pesticide itself) and dichlorodiphenyldichloroethane (DDD, an impurity of DDT). DDT, DDE, and DDD have very similar fate and transport properties. DDT, like many organochlorine contaminants, remains an ecological threat because of persistence in the environment, potential for volatilization and transport to untreated areas, lipophilic nature, the toxicity of metabolites, primarily DDE, and the variability of species responses to chronic exposure to DDT and DDT metabolites (Peterle 1991). DDT, DDE, and DDD are only slightly soluble in water, and adhere to soil and sediment particles, as predicted by their high Kocs ranging in the order of 10^5 to 10^6 (ATSDR 1993). DDT and its derivatives are fairly resistant to biodegradation and hydrolysis in soil and in aquatic ecosystems (Alexander 1965, as cited in Forsyth and others 1983). Consequently, these compounds are highly persistent in soil, with half-life estimates of greater than 15 years (Lichtenstein and others 1959 and Stewart and others 1971, as cited in ATSDR 1993). Volatilization of DDT residues is believed to be an important fate process in soils and in aquatic ecosystems (Callahan and others 1979). Volatilization accounts for the majority of initial decreases of DDT residues in soils (Willis and others 1971, as cited in Forsyth and others 1983).

DDT and its Congeners Bioconcentration. DDT and its metabolites are highly lipid soluble, which, combined with their relatively long half-life, results in very high bioconcentration factors. The Kow for DDD is 3.63×10^5, and the log Kow for DDT is 6.2. Biomagnification has been demonstrated for DDT and its metabolites in aquatic ecosystems, with organism residues increasing with increasing trophic levels (Bevenue 1976, as cited in Forsyth and others 1983). DDT is most widely known for its reproductive effects on birds, primarily predatory species, and for its toxicity to aquatic fish and invertebrates.

DDT and its Congeners Effects on Terrestrial Biota. Persistence of DDT in soil of agricultural plots, orchards, and forests, and its accumulation by plants and invertebrates has been demonstrated in several studies (Forsyth and others 1983). Reproductive effects of DDT and its metabolites on birds

has been extreme. To date, organochlorines, including DDT, have been implicated in the thinning of eggshells of at least 54 species of 10 orders of birds (Stickel 1975, as cited in Peterle 1991). Reproductive effects of DDT and its metabolites extend beyond eggshell thinning. Mallards treated with DDE showed delayed egg laying, reduced hatchability, and alteration of egg size, weight, and content (Vangilder and Peterle 1980, 1981, and 1983, as cited in Peterle 1991). Subacute effects of DDT on mammals range from transient effects on respiration to reproductive and behavioral effects in various species.

4.1.3.4 Dieldrin

Dieldrin has been used in the past as an insecticide; however, it is also an abiotic and biotic degradation product of the insecticide aldrin. Neither aldrin nor dieldrin is currently registered for general use as an insecticide in the United States (IARC 1973, as cited in HSDB 1994). Because of its low water solubility and high Koc, dieldrin is extremely persistent in soils (Schnoor and others 1983, as cited in HSDB 1994), with a half-life observed in the field of greater than 7 years (Nash and others 1967, as cited in HSDB 1994). Dieldrin can reach the atmosphere through volatilization from water or soil, a process which is slower from soil than from water (Gile and others 1979, as cited in HSDB 1994). Dieldrin may also be sorbed to fugitive dust (Schnoor and others 1983, as cited in HSDB 1994). A strong tendency to adsorb to soil particles generally prevents dieldrin from leaching to groundwater; however, dieldrin will reach surface water sorbed to soil particles in runoff. Once in water, dieldrin adsorbs strongly to sediments (Schnoor and others 1983, as cited in HSDB 1994). Dieldrin in water systems will not undergo hydrolysis or biodegradation (Schnoor and others 1983, as cited in HSDB 1994), and an aquatic half-life of greater than 4 years has been determined (EPA 1979, as cited in HSDB 1994). Biodegradation is also insignificant in soils (Tabak and others 1981, as cited in HSDB 1994).

Dieldrin Bioconcentration. Dieldrin is extremely nonpolar and has a strong affinity for organic matter, including animal fat and plant waxes (Cliath and others 1971, as cited in ATSDR 1993a). Dieldrin has a high potential for bioaccumulation, as indicated by its log Kow ranging from 4.32 (Geyer and others 1987, as cited in ATSDR 1993a) to 6.2 (Briggs 1981, as cited in ATSDR 1993a). Dieldrin is stored in adipose tissue, liver, brain, and muscle of mammals, fish, and birds, in algae,

plankton, insects, earthworms; and in the eggs of many bird species (IARC 1974, as cited in HSDB 1994).

Dieldrin Effects on Aquatic Biota. Population-level effects have also been demonstrated to result from exposure to dieldrin. In tests of exploitation rates on guppy populations, populations treated with 1 ppb dieldrin showed reduced growth and reproduction, which would have made them unable to sustain prolonged or high exploitation rates. Such population effects on species that serve as prey to many ecosystem predators could reduce ecosystem productivity and cause nutrient loss (Liss and others 1982, as cited in Peterle 1991).

Dieldrin Effects on Terrestrial Biota. Dieldrin adversely affects reproduction in birds. Japanese quail given 20 mg dieldrin/kg feed (approximately 2.5 mg/kg-day) for 9 weeks exhibited reduced parental and chick survival rates and laid fewer eggs. Like DDE exposure, exposure of birds to dieldrin has been associated with eggshell thinning. Dieldrin administered to mice at had varying effects on reproduction and survival (Virgo and Bellward 1975). Doses of 5 and 6.25 mg/kg-day caused high mortality, while doses of 2.5 mg/kg-day and higher caused severe behavioral abnormalities resulting in offspring mortality as a result of abandonments and direct mortality because of behavioral abnormalities in the dam.

4.1.3.5 Endrin

Endrin is a neurotoxin that has been used as an insecticide, an avicide, and a rodenticide. Endrin aldehyde, an impurity of technical endrin has fate and transport properties similar to endrin. Like most organochlorines, endrin is very persistent in soil, remaining for 14 years or more. Endrin's soil persistence results from its high Koc values (8,500 to 45,000), is low water solubility, its resistance to biodegradation, and slow volatization rate. Calculated half-lives in soil range from 4 to 14 years or more (Menzie 1972, Nash and others 1967, Alexander 1973, and Kenaga 1980; as cited in HSDB 1994).

Endrin can reach the atmosphere by slow evaporation and adsorbed to fugitive dust. Because of its strong tendency to adhere to soil particles and its low solubility in water, endrin generally does not leach to ground or surface water, but can reach surface water adsorbed to soil particles in runoff

(SRC 1980, as cited in HSDB 1994). In water, endrin adsorbs to sediment and does not evaporate, hydrolyze, or biodegrade. A half-life of greater than 4 years has been calculated for endrin in water (Eichelberger 1971 and EPA 1979, as cited in HSDB 1994). Photodegradation of endrin to ketoendrin in water does occur at low levels (Lyman and others 1982, as cited in HSDB 1994).

Endrin Bioconcentration. In microcosm and mosquito fish studies, endrin has been found to bioconcentrate significantly in aquatic organisms.

Endrin Effects on Aquatic Biota. Endrin is highly toxic to aquatic organisms. A 28-day LC_{50} of 0.42 ng endrin/L has been observed for the iosopod *Asellus* (Anderson and others 1980, as cited in HSDB 1994).

Endrin Effects on Terrestrial Biota. No data on the toxicity of endrin to avian species were found. Endrin has been shown to affect fetal weight when given to female mice during gestation.

4.1.3.6 Lindane

Lindane, also known as gamma-BHC, is a hexachlorocyclohexane organochlorine insecticide commonly used in vaporizers for insect control (Peterle 1991). Although lindane is very persistent, like most organochlorines, its environmental partitioning differs from other insecticides, like DDT. In aquatic systems, lindane's sorption to suspended sediment and biota is not extensive (Callahan and others 1979). In a year-long study (Hamelink and Waybrant 1973, as cited in Callahan and others 1979) of the persistence of DDT and lindane in a quarry, DDT and DDE (as a result of metabolism) were rapidly sorbed onto particulate matter, and DDE disappeared from the water column 15 times faster than lindane. After 3 months, 85 percent of the DDE introduced was in the sediment, whereas 72 percent of the lindane was still in the water. Suspended sediment, collected in traps, contained essentially no lindane compared to DDE, indicating that sorption to suspended sediment and sedimentation is not a significant fate process. Lindane in the bottom mud attained a maximum concentration of about 2 ppb compared, to the maximum for DDE of 35 ppb. Additionally, the DDE was contained in the top few centimeters of mud, whereas lindane rapidly diffused to the lower mud layers presumably because of lindane's solubility in the interstitial water in mud.

The fate of lindane in aquatic systems is controlled by the availability of biotransformation processes. Lindane transformation will be favored in biologically rich, anaerobic environments (Callahan and others 1979). Hydrolysis and oxidation do not appear to be important fate processes for lindane, and the data on photolysis of lindane are inconclusive. Volatilization from aquatic media is not significant (Spacie and others 1977, as cited in Callahan and others 1979).

Lindane Bioconcentration. No data were found on the bioconcentration of lindane.

Lindane Effects on Aquatic Biota. No data were found on the effects of lindane on aquatic receptors.

Lindane Effects on Terrestrial Biota. No data were found on the effects of lindane on terrestrial invertebrates, plants, birds, or mammals.

4.1.4 Radium-226

The only radioisotope posing a potential risk to ecological receptors at HPA is radium-226 (^{226}Ra), a naturally occurring radioactive element formed from the disintegration of uranium and thorium. Low levels can be detected in soil, water, rocks, coal, plants, and food. The isotope is an alpha and gamma emitter, with daughter products including alpha, beta, and gamma rays. Alpha particles do not pose biological risks because they cannot penetrate even the most sensitive cells (i.e., lung tissue) of potential receptors. However, beta and gamma particles may penetrate cell membranes.

Historically, high concentrations of ^{226}Ra have been detected at uranium mills, coal fired power plants, and on instrumentation dials and gauges illuminated with radioactive paints. Potential sources of radiation at HPA include radium-containing dials and also mixed fission products that may be present on the surfaces of ships docked for repair and maintenance. A surface confirmation radiation survey identified point source contamination of ^{226}Ra in several IR sites at HPA; however, no mixed fission products were detected at any site in concentrations above normally expected background levels (PRC 1992). Air sampling indicated only background concentrations of airborne ^{226}Ra, and gross alpha and beta emitting particulates. Additionally, no traces of the gaseous decay product of ^{226}Ra, radon-222, were detected (PRC 1991).

In association with particulate matter, ^{226}Ra may be transported in the atmosphere. Readily adsorbed by sediments and soils, ^{226}Ra is not a readily mobile constituent in the environment (Benes and others 1986; Landa and Reid 1982). The solubility of radium salts in water increases with increased pH levels (Langmuir and Riese 1985). However, ^{226}Ra exists primarily as a stable divalent ion in water, and its concentration is usually controlled by adsorption-desorption mechanisms at solid liquid interfaces and by the solubility of radium-containing minerals (ATSDR 1990b). Radium-226 does not hydrolyze, nor is it significantly influenced by oxidation-reduction reactions (Ames and Rai 1978). As a result, ^{226}Ra is not expected to migrate significantly from its point of origin. Surveys at HPA have preliminarily identified elevated gamma count rays in the soil immediately surrounding the ^{226}Ra contamination point sources (PRC 1992). The soil contamination is attributed to the physical decay of the dials, thereby releasing radium to the environment. However, surveys have indicated ^{226}Ra in the soil migrates only a short distance from the dials.

In the absence of sufficient scientific data, known chemical properties and site-specific pathways will be examined to characterize ^{226}Ra risk to the biotic community. Several factors may significantly reduce the potential Ra^{226} risk at HPA. Radium-226 contamination is buried between 3 and 5 feet below the soil surface. The point surfaces have a limited effects-range, and their subsurface location makes them unlikely to contact most receptors, with the exception of plants and burrowing organisms. The high rate of excretion and low rate of metabolic absorption reduce the potential effects of ^{226}Ra.

Radium-226 Bioconcentration. Existing information states ^{226}Ra may be bioaccumulated by plants and animals, and thus transferred in the food chain to higher trophic levels (ATSDR 1990b). At ^{226}Ra-contaminated sites, several mammals (meadow voles, coyote, and otters) and waterfowl have been examined and found to contain elevated levels of ^{226}Ra. While uptake in water may not pose a significant risk, sediment and vegetation serve as indirect sources of radionucleotide exposure (Arthur and Markham 1982; Wren and others 1987). Other studies conducted at uranium mines and mills indicate the mill tailings contained high levels of ^{226}Ra available to the environment. Elevated levels of ^{226}Ra were detected in receptors in the immediate vicinity of waste management areas (Clulow and others 1991).

4.1.5 Summary of Lowest Effects Levels

Table 4-1 presents the lowest available effects level for CPOCs for various classes of receptors. These values were selected from supporting material for regulatory criteria, government reports, data bases, published scientific literature, and other available sources. Available toxicity data for terrestrial receptors has been compiled and presented in tabular form in Appendix A. For the summary table, toxicity information on species closely related to receptor species at HPA was used whenever possible. Measurement endpoints related to growth, reproduction, survival, and bioaccumulation were preferred over physiological, neurological, and biochemical endpoints. These criteria could not always be met because of the limitations of the available data.

4.2 BENCHMARK VALUES FOR AQUATIC ORGANISMS

For COPCs in aquatic pathways at HPA, sediment quality criteria were compiled from available published sources. Described below are several studies that identified various contaminant effect levels, mean contaminant levels, or criteria for aquatic resource protection in San Francisco Bay and other bays in the United States. These criteria are used here to help evaluate the levels of COPCs in sediment and the potential biological effects of those contaminant levels to receptors at HPA. The effect levels derived from each study are presented for each COPC found off-shore at HPA in Table 4-2.

4.2.1 San Francisco Bay Basin Means

Mean sediment concentrations for various contaminants in the San Francisco Bay Basin were calculated (Bay Basin Means) and presented in "Status and Trends in Concentrations of Contaminants and Measures of Biological Stress in San Francisco Bay" (NOAA 1988). This report compiled sediment contaminant concentrations collected in several studies of San Francisco Bay from the 1970s and 1980s (San Francisco Bay includes Suisun Bay/Carquinez Strait, San Pablo Bay, Central Bay and South San Francisco Bay). The report refers to areas such as Mare Island Strait, Richmond Harbor Channel, Oakland Inner and Outer Harbor, Islais Creek Waterways, China Basin, and Redwood Creek as "periphery" areas, and data from these periphery areas are not included in the data set used

TABLE 4-1
TOXICITY OF CONTAMINANTS TO WILDLIFE

Contaminant	Effect on Plants and Soil Invertebrates	Effect on Aquatic Invertebrates	Effect on Fish	Effect on Birds	Effect on Mammals
ARSENIC (As)					
Effect Level and Medium	25 mg As/kg soil	0.1 mg As^{+5}/L in seawater	2.5 mg As^{+3}/L in saltwater	99.8 mg copper arsenite/kg diet	5 mg sodium arsenate/kg diet for three generations
Effect and Organism	significant depression in crop yield	reduced survival of juvenile copepod (*Eurytemora affinis*)	10 day LC50 for pink salmon (*Oncorhynchus gorbuscha*)	11-day LC50 for brown-headed cowbird (*Molothrus ater*)	reduced litter size in mice
Reference	(Eisler 1988a)	(Eisler 1988a)	(Eisler 1988a)	(Eisler 1988a)	(Eisler 1988a)
BERYLLIUM (Be)					
Effect Level and Medium	ND	ND	ND	ND	0.95 mg Be/kg-day in drinking water
Effect and Organism	ND	ND	ND	ND	NOAEL in mice
Reference	ND	ND	ND	ND	(Schroeder and Mitchener 1975)
CADMIUM (Cd)					
Effect Level and Medium	100 mg Cd/kg in soil	20.8 mg Cd/kg sediment	0.5 ppb in seawater for 30 days	0.4 mg/kg-day in diet	0.73 mg/kg-day in diet
Effect and Organism	reduced yield for agricultural and grassland species	EC-50 for amphipod (*Rhepjoxynius abronius*) reburial	decreased oxygen consumption in juvenile striped bass (*Morone saxatalis*)	reduced egg production in chickens	fetotoxicity: reduced fetal and pup weights in mice
Reference	(EPA 1987)	(NOAA 1991)	(Eisler 1985a)	(Scheuhammer 1987)	(ATSDR 1993e)

TABLE 4-1
TOXICITY OF CONTAMINANTS TO WILDLIFE (continued)

Contaminant	Effect on Plants and Soil Invertebrates	Effect on Aquatic Invertebrates	Effect on Fish	Effect on Birds	Effect on Mammals
CHROMIUM (Cr)					
Effect Level and Medium	10 ppm Cr in irrigation water applied to soil	0.5 ppm Cr^{+6} in sediment	16 ppb Cr^{+6} in freshwater for 14 to 16 weeks	100 ppm Cr^{+6} in diet	1,500 mg Cr^{+3}/kg-day in diet
Effect and Organism	mortality in earthworms after 58 days	shell disease syndrome produced in 41% test grass shrimps (*Palaemonetes pugio*) in 28 days	reduced growth of rainbow trout and chinook salmon fingerlings	no adverse effects on survival, growth, or food utilization in chickens in 32 days	NOAEL on organ systems in rats for a lifetime exposure
Reference	(Eisler 1986a)	(Eisler 1986a)	(Eisler 1986a)	(Eisler 1986a)	(Ivankovic and Preussman 1975)
COPPER (Cu)					
Effect Level and Medium	100 mg total Cu/kg in soil	14.7 ppm Cu in sediment	ND	40.5 mg copper sulfate/kg-day	78 mg copper sulfate/kg-day
Effect and Organism	threshold concentration for toxicity to plants and soil invertebrates	reduced burrowing efficiency in little neck clam (*Protothaca staminea*)	ND	no effects on growth in 4 week	no effects on fetal mortality or development in pregnant mice
Reference	(ICA 1992)	(NOAA 1992)	ND	(NAS 1980)	(ATSDR 1990a)
CYANIDE (CN)					
Effect Level and Medium	ND	5.221 μg CN/L in freshwater	2 μg CN/L in seawater	1.1 mg CN/kg bw as a single oral dose of sodium cyanide	2.5 mg CN/kg bw as a single oral dose of hydrogen cyanide
Effect and Organism	ND	proposed chronic criterion designed to protect 95% of freshwater species	threshold concentration for adverse chronic effects on marine organisms	LD06 for mallards	LD50 for rabbits (*Oryctolagus* spp.)
Reference	ND	(Eisler 1991)	(Eisler 1991)	(Eisler 1991)	(Eisler 1991)

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TABLE 4-1
TOXICITY OF CONTAMINANTS TO WILDLIFE (continued)

Contaminant	Effect on Plants and Soil Invertebrates	Effect on Aquatic Invertebrates	Effect on Fish	Effect on Birds	Effect on Mammals
LEAD (Pb)					
Effect Level and Medium	500 mg Pb/kg dw in soil	17 μg Pb/L in water	50 μg tetramethyl Pb/L in seawater	1.25 mg Pb/kg-day in diet	50 mg lead acetate/kg in drinking water
Effect and Organism	90% and 87% reduced pollen and seed germination	lifetime MATC for mysid (*Mysidopsis bahia*)	96-hour LC50 in plaice (*Pleuronectes platessa*)	significantly decreased egg production and plasma calcium in quail	significant growth retardation, delays in maturation, and reduced body weight in rat pups
Reference	(Eisler 1988b)	(Eisler 1988b)	(Eisler 1988b)	(Edens and Gerlich 1983)	(Kimmel and others 1980)
MERCURY (Hg)					
Effect Level and Medium	0.79 ppm Hg in soil	0.03 ppb Hg in water	0.01 ppm Hg in water	0.32 mg mercuric chloride/kg-day in diet	1,100 μg/kg fw in diet
Effect and Organism	LD50 for earthworms in 60 days	inhibited reproduction in aquatic organisms	mosquitofish suffered higher predation by large-mouth bass than controls	LOAEL for quail	dietary concentration protective of mammals
Reference	(Eisler 1987a)	(Eisler 1987a)	(Peterle 1991)	(Hill and Soares 1984)	(Eisler 1987a)
MOLYBDENUM (Mo)					
Effect Level and Medium	ND	147 mg Mo/L in seawater	28 μg Mo/L in freshwater	200 mg Mo/kg diet	100 mg Mo/kg diet
Effect and Organism	ND	development of mussel larvae (*Mylitus edulis*) reduced 50% in 48 hours, based on survival and abnormalities	28-day LC1 for rainbow trout embryos and larvae	minor growth inhibition in chicks	reduced growth, skeletal deformities in rabbits for lifetime exposure
Reference	ND	(Eisler 1989)	(Eisler 1989)	(Eisler 1989)	(Eisler 1989)

TABLE 4-1
TOXICITY OF CONTAMINANTS TO WILDLIFE (continued)

Contaminant	Effect on Plants and Soil Invertebrates	Effect on Aquatic Invertebrates	Effect on Fish	Effect on Birds	Effect on Mammals
NICKEL (Ni)					
Effect Level and Medium	ND	141 ppb	141 ppb	ND	ND
Effect and Organism	ND	general chronic effect level for marine organisms	general chronic effect level for marine organisms	ND	ND
Reference	ND	(NOAA 1991)	(NOAA 1991)	ND	ND
SELENIUM (Se)					
Effect Level and Medium	ND	127 ppb in seawater	47 ppb Se	300 ppb Se in water	.8 ppm dietary Se, lifetime exposure
Effect and Organism	ND	96-hour LC-50 in a life cycle test on mysid shrimp eggs (Mysidopsis bahia)	anemia and reduced hatch of rainbow trout	adverse reproductive effects on ducks (*Anas* spp.), American coot (*Fulica americana*), and other aquatic birds	intestinal lesions in rats
Reference	ND	(Eisler 1985b)	(Eisler 1985b)	(Eisler 1985b)	(Eisler 1985b)
SILVER (Ag)					
Effect Level and Medium	ND	0.36 ppb Ag in water	ND	ND	18.1 mg/kg-day in drinking water
Effect and Organism	ND	growth retardation in sea urchin larvae	ND	ND	NOAEL for unspecified systemic endpoints in mice
Reference	ND	(EPA 1992)	ND	ND	(ATSDR 1990c)

TABLE 4-1
TOXICITY OF CONTAMINANTS TO WILDLIFE (continued)

Contaminant	Effect on Plants and Soil Invertebrates	Effect on Aquatic Invertebrates	Effect on Fish	Effect on Birds	Effect on Mammals
TRIBUTYLTIN (TBT)					
Effect Level and Medium	ND	0.02 ppb TBT in seawater	0.96 ppb TBT in water	ND	122 mg TBT chloride/kg
Effect and Organism	ND	NOEL for unspecified effects in Pacific oyster (*Crassostrea gigas*)	LC-50 for sheepshead minnow	ND	oral LD^{50} for rats
Reference	ND	(Rexrode 1987)	(Rexrode 1987)	ND	(Rexrode 1987)
ZINC (Zn)					
Effect Level and Medium	97 mg Zn/kg in soil	10 μg Zn/L in seawater	10 ppb in water	33.8 mg/kg-day in diet	500 mg Zn/kg in diet
Effect and Organism	proposed criteria for earthworm survival (may still have adverse effects)	threshold concentration for adverse effects on Pacific oyster (*Crassostrea gigas*)	growth in fish	LOAEL for Japanese quail	unspecified adverse effects on rats
Reference	(Eisler 1993)	(Eisler 1993)	(Eisler 1993)	(NAS 1980)	(Eisler 1993)
BENZENE					
Effect Level and Medium	ND	ND	ND	ND	ND
Effect and Organism	ND	ND	ND	ND	ND
Reference	ND	ND	ND	ND	ND

TABLE 4-1
TOXICITY OF CONTAMINANTS TO WILDLIFE (continued)

Contaminant	Effect on Plants and Soil Invertebrates	Effect on Aquatic Invertebrates	Effect on Fish	Effect on Birds	Effect on Mammals
POLYCYCLIC AROMATIC HYDROCARBONS (PAHs)					
Effect Level and Medium	ND	1.0 μg chrysene/L in water	0.1 μg/L benzo(a)pyrene in water for 5 days	4,000 mg PAHs/kg in diet for 7 months	0.002 mg PAH/kg bw chronic oral exposure
Effect and Organism	ND	high accumulations of chrysene in Pink shrimp (*Penaeus duorum*)	reduced and delayed hatch of sand sole eggs; gross abnormalities in sand sole larvae	only sign of toxicity evident in Mallards during exposure was increased liver weight and liver blood flow	induced cancer in laboratory rodents
Reference	ND	(Eisler 1987b)	(Eisler 1987b)	(Eisler 1987b)	(Eisler 1987b)
TOTAL PETROLEUM HYDROCARBON (TPH)					
Effect Level and Medium	ND	15 μg TPH/L in seawater	15 μg TPH/L in seawater	ND	ND
Effect and Organism	ND	Alaska state criterion for protection of marine organisms	Alaska state criterion for protection of marine organisms	ND	ND
Reference	ND	(ATSDR 1993h)	(ATSDR 1993h)	ND	ND
PCBs					
Effect Level and Medium	1 μg PCBs/L in water	<0.03 μg PCBs/L in seawater, 24 hour average	<0.5 ppm PCBs in diet	<3 ppm PCBs in diet	<100 μg PCB/kg diet (fw) (1.5 μg/kg-day)
Effect and Organism	common soil algae exhibited decreased photosynthesis	proposed criteria for protection of aquatic organisms	proposed criteria for protection of fish species	proposed criterion for protection of avian species	proposed criteria for protection of mink
Reference	(HSDB 1994)	(Eisler 1986b)	(Eisler 1986b)	(Eisler 1986b)	(Eisler 1986b)

TABLE 4-1
TOXICITY OF CONTAMINANTS TO WILDLIFE (continued)

Contaminant	Effect on Plants and Soil Invertebrates	Effect on Aquatic Invertebrates	Effect on Fish	Effect on Birds	Effect on Mammals
CHLORDANE					
Effect Level and Medium	ND	<0.004 μg chlordane/L in seawater, 24 hour average	0.2 μg chlordane/L in seawater	0.1 ppm chlordane in diet	<5 ppm chlordane in diet (0.25 mg/kg-day)
Effect and Organism	ND	proposed criteria for protection of aquatic organisms	reduced survival of marine fishes	proposed criterion for protection of avian species	NOEL for unspecified effects in rat
Reference	ND	(Eisler 1990)	(Eisler 1990)	(Eisler 1990)	(Eisler 1990)
DDT AND ITS CONGENERS (DDD AND DDE)					
Effect Level and Medium	ND	ND	ND	0.12 mg DDD/kg-day in diet	2.86 mg DDE/kg-day in diet
Effect and Organism	ND	ND	ND	NOAEL for reproductive effects in a raptor population	lowest chronic level associated with adverse hepatic effects
Reference	ND	ND	ND	(Lincer 1975)	(NCI 1992)
DIELDRIN					
Effect Level and Medium	ND	ND	2.3 ppb dieldrin in diet for 30 days	0.3 mg dieldrin/kg in diet	1.25 mg dieldrin/kg-day in diet
Effect and Organism	ND	ND	increased susceptibility to mortality in darter with water temp. increases	NOEL for mallard ducklings	reduced litter size in mice
Reference	ND	ND	(Peterle 1991)	(Nebeker and others 1992)	(Virgo and Bellward 1975)
ENDRIN					
Effect Level and Medium	ND	.42 ng endrin/L in water	ND	ND	ND
Effect and Organism	ND	LC-50 for isopod Asellus	ND	ND	ND
Reference	ND	(HSDB 1994)	ND	ND	ND

TABLE 4-1
TOXICITY OF CONTAMINANTS TO WILDLIFE (continued)

Contaminant	Effect on Plants and Soil Invertebrates	Effect on Aquatic Invertebrates	Effect on Fish	Effect on Birds	Effect on Mammals
LINDANE					
Effect Level and Medium	ND	ND	ND	ND	ND
Effect and Organism	ND	ND	ND	ND	ND
Reference	ND	ND	ND	ND	ND

MATC maximum acceptable threshold concentration
LC concentration causing death in a given percentage of test organisms over a specified period of time (A 96-hour LC50 is the concentration causing death in 50 percent of test organisms in 96 hours.)
EC concentration causing a specified (or any) effect in a given percentage of test organisms over a specified period of time (A 96-hour EC50 is the concentration causing a particular effect in 50 percent of the test organisms in 96 hours.)
bw body weight
fw fresh weight
LC lethal concentration
ND no data
NOAEL no observed adverse effects level
EC environmental (effects) concentration
bw body weight
dw dry weight
MATC maximum acceptable tissue concentration
LOAEL lowest observed adverse effects level
fw fresh weight
TPH total petroleum hydrocarbon
NOEL no observed effects level

TABLE 4-2
PROPOSED CRITERIA FOR PROTECTION OF SEDIMENT AND SURFACE WATER

Constituent	Sediment					Surface Water	
	Bay Basin Mean[a] (No. of samples) (Concentration Range)	NOAA[b] ER-L/ ER-M	Bivalve AET[c]	Amphipod AET[c]	Benthic AET[c]	Ambient Water Quality Criteria[d]	SFBBP[e]
Metals (ppm)							
Arsenic	--	8.2 / 70.0*	70.0	> 72.0	70.0	0.036	0.036
Cadmium	0.94 (256) (0.02-17.3)	1.2 / 9.6*	0.57	1.7	0.48	0.0093	0.0093
Chromium	108 (140) (9 - 769)	81.0 / 370.0*	> 240.0	> 240.0	> 240.0	0.05	0.05
Copper	36 (376) (3 - 145)	34.0 / 270.0*	66.0	98.0	66.0	--	0.0029
Lead	32 (461) (1 - 421)	46.7 / 223.0*	71.0	120	41.0	0.0085	0.0056
Mercury	0.45 (396) (<0.01 - 6.60)	0.15 / 0.71*	0.51	1.2	0.51	0.000025	0.0021
Nickel	--	20.9 / 51.6*	> 170.0	> 170.0	> 170.0	0.0083	0.0083
Silver	0.51 (148) (0.01 - 2.4)	1.0 / 3.7*	2.3	> 8.6	2.4	0.00092	0.0023
Selenium	--	--	--	0.2	--	0.071	0.005
Zinc	--	150 / 410*	150.0	230	150.0	0.086	0.086
Nonmetals (ppb)							
PCBs	40.5 (37) (6 - 140)	50.0 / 400.0	88.0	260.0	88.0	0.03	--
Pesticides (ppb)							
Total DDT	9.0 (75) (0.25 - 9100)	3.0 / 350.0	27.0	27.0	24.0	0.001	0.001
DDT	--	1.0 / 7.0	9.6	9.6	4.7	0.001	--

TABLE 4-2 (continued)
PROPOSED CRITERIA FOR PROTECTION OF SEDIMENT AND SURFACE WATER

Constituent	Sediment					Surface Water	
	Bay Basin Mean[a] (No. of samples) (Concentration Range)	NOAA[b] ER-L/ ER-M	Bivalve AET[c]	Amphipod AET[c]	Benthic AET[c]	Ambient Water Quality Criteria[d]	SFBBP[e]
DDE	--	2.0 / 15.1	2.6	2.9	> 2.6	--	--
DDD	--	2.0 / 20.0	3.1	2.4	1.4	--	--
Chlordane	--	0.5 / 6.0	--	--	--	0.004	--
Dieldrin	--	0.02 / 8.0	6.6	6.6	6.2	0.0019	0.0019
PAH (ppb)							
Anthracene	--	85.3 / 1100.0*	60.0	1100.0	60.0	--	--
Benzo(a)-anthracene	--	261.0 / 1600.0*	150.0	1100.0	150.0	--	--
Benzo(a)-pyrene	--	430.0 / 1600.0*	430.0	> 1300.0	430.0	--	--
Chrysene	--	384.0 / 2800.0*	190.0	2100.0	190.0	--	--
Dibenz(a,h)-anthracene	--	63.4 / 260.0*	63.0	> 300.0	63.0	--	--
Fluoranthene	--	600.0 / 5100.0*	39.0	> 3700.0	390.0	16.0	--
Perylene	--	--	170.0	230.0	170.0	--	--
Phenanthrene	--	240.0 / 1500.0*	170.0	510.0	170.0	4.6	--
Pyrene	--	665.0 / 2600.0*	490.0	2600.0	490.0	--	--
Total PAH	4100 (101) (2.0-80900)	4022.0 / 44790.0	--	--	--	--	--

a NOAA 1988
b NOAA 1991; * MacDonald 1993
c PTI 1989
d EPA 1992
e RWQCB 1992

NOAA National Oceanic and Atmospheric
ER-L effects range-low
ER-M effects range-median
Bivalve bivalve larvae abnormality bioassay
AET apparent effects threshold
SFBBP San Francisco Bay Basin Plan
PCB polychlorinated biphenyls
DDT dichlorodiphenyltrichloroethane
DDE dichlorodiphenyldichloroethylene
DDD dichlorodiphenyldichloroethane
PAH polynuclear aromatic hydrocarbons

to calculate the Bay Basin Means. The Bay Basin Means for contaminants found in the off-shore environment at HPA are presented in Table 4-2.

4.2.2 NOAA Effects Range-Low and Effects Range-Median

The National Oceanic and Atmospheric Administration has developed and presented biological effects ranges in "The Potential for Biological Effects of Sediment-Sorbed Contaminants in The National Status in the National Status and Trends Program" (NOAA 1991). The report is a compilation of many sediment toxicity evaluations that each used one of four different approaches: 1) equilibrium-partitioning approach, 2) spiked-sediment bioassay approach, 3) apparent effects approach, and 4) the cooccurrence of analysis of field date approach. The sediment toxicity results of all studies were ranked for each individual parameter. The lower 10 percentile toxicity value in the data was identified as the effects range-low (ER-L), and the median toxicity value was identified as the effects range-median (ER-M). The ER-L is the chemical concentration above which adverse biological effects may be expected at least 10 percent of the time. The ER-M is the chemical concentration above which adverse biological effects may be expected at least 50 percent of the time. Adverse biological effects include mortality or sublethal effects such as reduced growth or reproduction success. ER-Ls and ER-Ms for off-shore COPCs at HPA are presented in Table 4-2.

4.2.3 Apparent Effects Threshold

An apparent effects threshold (AET) is defined as the sediment concentration of a given chemical above which statistically significant (greater than or equal to 0.05) biological effects are always observed in the data set used to generate AET. If any chemical exceeds its AET for a particular biological indicator, an adverse biological effect is predicted for that indicator. If all chemical concentrations are below their AET for a particular biological indicator, no prediction is made. The AET values for contaminants found off-shore at HPA presented in Table 4-2 were developed from Northern California data with three biological indicators: 1) the amphipod mortality bioassay (amphipod AET), 2) the bivalve larvae abnormality bioassay (bivalve AET), and 3) benthic effects (benthic AET) (PTI 1989).

4.2.4 Ambient Water Quality Criteria

Federal Ambient Water Quality Criteria (AWQC) documents have been published for 65 pollutants listed as toxic under the Clean Water Act. They are designed to protect most aquatic life against acute or chronic toxicity (24-hour average). For many chemical compounds, specific criteria have not been established because of insufficient data. Marine chronic AWQC for contaminants found in marine surface water off-shore of HPA are presented in Table 4-2.

4.2.5 San Francisco Bay Basin Plan Effluent Discharge Limits

The San Francisco Bay Basin Plan (SFBBP) Effluent Discharge Limits were developed under the National Pollutant Discharge Elimination System, which allows states to establish their own water quality protection programs. The SFBBP (RWQCB 1992) provides both shallow water and deep water effluent discharge limits designed to be protective of the aquatic environment. The shallow water effluent discharge limits for contaminants found off-shore at HPA are presented in Table 4-2.

4.3 SEDIMENT AND SURFACE WATER BIOASSAY DATA EVALUATION

The HPA interim berthing reassessment study (MBL 1987b), the homeporting verification study (MBL 1987a), and the environmental sampling and analysis plan (ESAP) (ATT 1991) were evaluated to identify potentially contaminated sediment habitats and potentially contaminated San Francisco Bay water. The data on sediment toxicity, bay water toxicity, and mussel bioaccumulation studies from the ESAP were evaluated for reliability and quality. The analysis of these data are presented below.

4.3.1 Interim Berthing Reassessment Study

In 1987, a dredged material evaluation was conducted to determine the potential ecological impact of sediments outside the entrance to drydock 4 at HPA (MBL 1987b). Three 5-foot sediment cores were collected outside the drydock entrance. Composite samples were prepared from the three upper 2-foot segments and the three lower 3-foot segments. Data were reported for the following:

- Bulk sediment chemistry on:

  - Volatile organic compounds
  - PAHs
  - Priority pollutant heavy metals

  - Other parameters including petroleum hydrocarbons, free sulfides, pH, cyanide, total organic carbon, moisture, gross alpha and gross beta radioactivity, and particle size distribution

- Suspended particulate phase (elutriate) bioassays on:

  - Survival of the mysid shrimp, *Acanthomysis sculpta*

  - Survival of the sanddab, *Citharichthys stigmaeus*

  - Fertilization of the mussel, *Mytilus edulis*

- Bulk sediment bioassays on:

  - Survival of the mysid shrimp, *Acanthomysis sculpta*

  - Survival of the clam, *Macoma nasuta*

  - Survival of the polychaete worm, *Nephthys caecoides*

  - Survival, emergence, and reburial of the amphipod, *Rhepoxynius abronius*

- For surviving *Macoma nasuta* and *Nephthys caecoides* from the bulk sediment bioassays, bioaccumulation (tissue) data on:

  - Heavy metals

  - PAHs

  - Organotins

  - Chlorinated pesticides

  - PCBs

  - TPH

  - Oil and grease

The results of the chemical analyses of the bulk sediment included a number of potentially toxic compounds, including (1) the volatiles acetone and total xylenes, (2) all of the priority pollutant heavy metals, and (3) numerous PAHs including phenanthrene, fluoranthene, pyrene, benzo(a)anthracene, chrysene, benzo(b,k)fluoranthene, and benzo(a)pyrene. No gradients between the surface and subsurface sediments were apparent. The chemical data were not statistically compared with data from a reference site.

To determine whether the drydock 4 bulk sediment toxicity test responses were significantly affected or if the tissue contaminant results were significantly elevated, the data were compared with bulk sediment toxicity data and bioaccumulation data collected from a reference sediment collected in the vicinity of the disposal site. To determine whether the drydock 4 elutriate toxicity test responses were significantly reduced, the data were compared with elutriate data from a natural seawater control. Statistical comparisons were conducted at a 95 percent significance level. The results are presented in Table 4-3.

The results of MBL (1987a) indicate (1) that the upper and lower sediments contain detectable concentrations of toxic heavy metals and PAHs and (2) that heavy metals and the pesticide DDE accumulate in tissue of infauna. The bulk sediment bioassay data suggest that the surface and subsurface sediments may be toxic to benthic infauna as sensitive as the burrowing amphipod, *Rhepoxynius abronius*. The elutriate data suggest that toxic contaminants associated with the lower sediments may be released to the pelagic environment if the sediments are disturbed.

4.3.2 Homeporting Verification Study

A chemical and toxicity investigation of sediments at HPA was conducted to support the Environmental Impact Statement for the Battleship Missouri Homeporting in San Francisco Bay (MBL 1987b). Triplicate 2-foot sediment core samples were collected from each of three stations off the east shore of HPA, including (1) in the North Berthing Slip (sample HP-1C), (2) off the south side of the terminus of North Pier (sample HP-2A), and (3) off the south side of the middle South Pier

TABLE 4-3
RESULTS OF BULK SEDIMENT BIOASSAYS, ELUTRIATE BIOASSAYS, AND BIOACCUMULATION TESTS FROM HPA REASSESSMENT STUDY

Sample	Test	Organism/Endpoint	Statistically different from reference?
Upper 2-foot sediment composite	Suspended particulate (elutriate) bioassay	• 96-hr mysid survival • 96-hr sanddab survival • 48-hr mussel fertilization	Yes No No
Upper 2-foot sediment composite	Bulk sediment bioassay	• 20-day mysid survival • 20-day clam survival • 20-day polychaete survival • 10-day amphipod - survival - emergence - reburial	No No No Yes No No
Upper 2-foot sediment composite	Tissue bioaccumulation	• 20-day clam tissue • 20-day polychaete tissue	Arsenic Nickel[a] —[b]
Lower 3-foot sediment composite	Suspended particulate (elutriate) bioassay	• 96-hr mysid survival • 96-hr sanddab survival • 48-hr mussel fertilization	Yes Yes Yes
Lower 3-foot sediment composite	Bulk sediment bioassay	• 20-day mysid survival • 20-day clam survival • 20-day polychaete survival • 10-day amphipod - survival - emergence - reburial	No No No Yes No Yes
Lower 3-foot sediment composite	Tissue bioaccumulation	• 20-day clam tissue • 20-day polychaete tissue	Nickel DDE[a] —[b]

Notes:

a Arsenic and nickel, and nickel and dichlorodiphenyldichlroethylene (DDE) were statistically elevated in clam tissue from upper 2-foot composite sediment toxicity test and lower 3-foot composite sediment toxicity test, respectively. No other parameters were statistically elevated above clam tissue concentrations from bulk sediment reference toxicity test.

b No parameter was statistically elevated above its concentration in polychaete tissue from bulk sediment reference toxicity test

(sample HP-4B). The replicates were composited before analysis. The depth from which the sediment core samples were collected is not evident.

Data from two reference sediment stations were used to determine the potential ecological effect of the target dredge material. The reference stations included one at Alcatraz Island and one off-shore.

Chemical data were collected from each of the three stations and bioassay data were collected from stations HP-2A and HP-4B. Data were reported for the following:

- Bulk sediment chemistry on:
  - Volatile organic compounds
  - Semivolatile organic compounds
  - Priority pollutant heavy metals
  - Chlorinated pesticides and PCBs
  - Organotins
  - Other parameters including total organic carbon, particle size distribution, oil and grease, cyanide, sulfide, and moisture
- Suspended particulate phase (elutriate) bioassays on:
  - Survival of the mysid shrimp, *Acanthomysis sculpta*
  - Development of larvae of the oyster, *Crassostrea gigas*
- Bulk sediment bioassays on:
  - Survival, emergence, and reburial of the amphipod, *Rhepoxynius abronius*

During the chemical analyses of the sediments, numerous contaminants were detected, most of which may be toxic and may bioaccumulate. The only volatile constituent detected was acetone. PAHs, which were the only semivolatile compounds detected, included benzo(a)anthracene, benzo(a)pyrene, benzofluoranthene, benzo(g,h,i)perylene, chrysene, fluoranthene, indeno(1,2,3-c,d)pyrene, pyrene, and phenanthrene. All priority pollutant metals were detected in the three samples. The pesticides

DDD and DDE were also detected. No organotins were detected. The chemical data were not statistically compared with data from the reference sites.

To determine whether the bulk sediment toxicity test responses were significantly reduced, the data were compared with bulk sediment toxicity data collected from the reference sediments. To determine whether the elutriate toxicity test responses were significantly reduced, the data were compared with elutriate data from a natural seawater control. Statistical comparisons were conducted at a 95 percent significance level. The results are presented in Table 4-4.

The bioassay results indicate that the sediments at the sites on the east side of the facility may be toxic to benthic fauna.

4.3.3 Environmental Sampling and Analysis Plan

The bioassay data collected under the ESAP program (ATT 1991) included toxicity information on four types of samples: (1) storm water, (2) bay water near the storm water outfalls, (3) solid phase sediments, and (4) sediment elutriates. The sediment samples were collected from the subtidal zone around HPA.

Storm water toxicity data were collected from four locations (ST1 through ST4). Three freshwater organisms were tested: (1) fathead minnow (*Pinephalses promelas*), (2) water flea (*Cerrondaphnia dubia*), and (3) green alga (*Selenastrim capricornitum*).

Bay water toxicity data were collected from four stations (B-1 through B-4). Three marine organisms were tested: (1) inland silverside (*Menidia beryllina*), (2) purple sea urchin (*Strongylocentrotus purpuratus*), and (3) diatom (*Skeletonema costatum*). The number designation of the bay water stations corresponds to the number designation of the storm water samples. For example, station B-1 was near the outfall of ST1 storm water.

Solid phase sediment toxicity data were collected from 17 subtidal stations (S-01 through S-17) circumscribing HPA. Three organisms were tested: (1) amphipod (*Eshaustories estuaris*), (2) polychaete (*Nephytys caecoides*), and (3) mysid shrimp (*Holmesimysis costata*).

TABLE 4-4
RESULTS OF BULK SEDIMENT BIOASSAYS AND ELUTRIATE BIOASSAYS, FROM HPA VERIFICATION STUDY

Sample	Test	Organism/Endpoint	Statistically lower than ref. water or sediment?		
			Seawater Control	Alcatraz	Off-shore
HP-2A	Elutriate bioassay	• 96-hr mysid survival	No	--	No
		• 48-hr oyster larvae develop[a]	Yes	--	Yes
HP-2A	Bulk sediment bioassay	• 10-day amphipod			
		- survival	--	No	Yes
		- emergence	--	No	No
		- reburial	--	No	Yes
HP-4B	Elutriate bioassay	• 96-hr mysid survival	--	No	No
		• 48-hr oyster larvae develop[a]	--	Yes	Yes
HP-4B	Bulk sediment bioassay	• 10-day amphipod			
		- survival	--	No	Yes
		- emergence	--	No	No
		- reburial	--	No	Yes

Notes:

a Oyster larvae development tests conducted at treatments of 100 percent, 50 percent, and 10 percent elutriate concentration. Only results of 100 percent treatment reported here.

Reference sediment was collected from three stations (RS1 through RS3). RS1 and RS2 are south of HPA in San Francisco Bay. RS1 is located off the southern shore of candlestick point, and RS2 is located about 800 meters north of the northeast corner of Sierra Point. RS3 is in San Pablo Bay at the intersection of Marin, Contra Coasta, and Sonoma counties.

The bioassay data were validated (PRC 1993b), but much of the results were inconclusive because of numerous confounding factors, including poor test designs, anomalous information, and data gaps. The objective of the usability analysis was to determine whether the data could be used as a tool to focus the forthcoming quantitative data collection activities under the Phase 1B ecological risk assessment.

The following factors were used to determine the usefulness of the bioassay data:

- Whether the test acceptability criteria were achieved and, if not, the degree of deviation
- Reasonableness of the dose-response curves
- Usability of dose-response data
- Whether proper procedures and test methods were employed
- Appropriateness of the exposure conditions
- Results of quality control practices
- Results of the reference station bioassays

The results of the bioassay data usefulness analysis are presented in Table 4-5. The analysis found some data could be used to (1) identify and rank the media in terms of the potential to cause adverse ecological effects and (2) determine whether specific bioassays were consistent with the test medium.

In general, the following conclusions can be drawn:

1. Solid phase sediment samples exhibited the highest and most consistent toxicity.

TABLE 4-5
USABILITY OF ENVIRONMENTAL SAMPLING AND ANALYSIS PLAN BIOASSAY DATA

Test Medium/Stations	Test	Test Acceptability Criteria Achieved for Control?	Dose-Response Curves Reasonable?	Specific Dose-Response Data Useable?	Proper Procedures and Methods Used?	Test Conditions Appropriate?
Storm water ST1-ST4	Fathead minnow survival and growth	Yes	Yes-all stations	Not applicable	Yes	Yes
Storm water ST1-ST4	Water flea survival and reproduction	Yes	No-all stations (no relationship between dose and response)	Yes-complete mortality in full strength (100%) ST4 sample	Inconclusive	Inconclusive-no laboratory documentation
Bay water B1-B4	Inland silverside survival and growth	Yes	Yes-all stations	Not applicable	Yes	Yes
Bay water B1-B4	Purple sea urchin fertilization	No-low survival No-low fertilization	No-all stations (no relationship between dose and response)	Yes-all full strength (100%) samples demonstrated high fertilization rates	Generally yes; minor deviations	No-true control and diluent toxic
Bay water B1-B4	Diatom growth	Yes	No-all stations (no relationship between dose and response)	Yes-Full strength (100%) B4 sample shows algal stimulation	No-samples not filtered; procedures unproven as they were modified from freshwater algal method.	Inconclusive-no laboratory documentation
Sediment elutriate ST1-ST17	Mysid survival	No-low survival	No-all stations (no relationship between dose and response)	Not applicable	Inconclusive	No-low dissolved oxygen; excessive turbulence from aeration

TABLE 4-5
USABILITY OF ENVIRONMENTAL SAMPLING AND ANALYSIS PLAN BIOASSAY DATA

Test Medium/Stations	Test	Test Acceptability Criteria Achieved for Control?	Dose-Response Curves Reasonable?	Specific Dose-Response Data Useable?	Proper Procedures and Methods Used?	Test Conditions Appropriate?
Sediment elutriate ST1-ST17	Sanddab survival	No-low survival	Yes. Flat curves indicate adverse effects do not increase as concentration increases	See comments	Inconclusive	No-low dissolved oxygen; excessive turbulence; fish stressed
Sediment elutriate ST1-ST17	Bivalve survival and abnormality	Yes	Yes. Shallow-little change in response as sample concentration increases	Yes. Abnormality significantly elevated at stations S16, S3, S15, S1, and S6	Yes	Yes
Solid phase sediment ST1-ST17	Mysid survival	Yes	Not applicable	Not applicable	Inconclusive	Overlying water required aeration, causing resuspension of sediment and physical turbulence
Solid phase sediment ST1-ST17	Amphipod survival	Yes-survival No-replicate mortality	Not applicable	Not applicable	Inconclusive	Overlying water required aeration, causing resuspension of sediment and physical turbulence
Solid phase sediment ST1-ST17	Polychaete survival	Yes	Not applicable	Not applicable	Inconclusive	Overlying water required aeration, causing resuspension of sediment and physical turbulence.

TABLE 4-5
USABILITY OF ENVIRONMENTAL SAMPLING AND ANALYSIS PLAN BIOASSAY DATA

Test Medium/Stations	Test	Reference Station Test Results	Validated Adverse Effects?	Comments
Sediment elutriate ST1-ST17	Sanddab survival	60-82% survival; generally consistent with HPA survival	Test acceptability criteria not achieved-test invalid	Comments on laboratory bench sheets indicate fish eating each other. Statistically significant mortality at stations S6 and S8. Excessive (not stat. sig.) mortality at station S-16.
Sediment elutriate ST1-ST17	Bivalve survival and abnormality	8-11% mortality and 26-33% abnormality; consistent with HPA mortality and abnormality	Yes. Significantly elevated abnormality at stations S16, S3, S15, S1, and S6.	No comments
Solid phase sediment ST1-ST17	Mysid survival	29-39% survival; consistent with HPA group 1 survival but lower than HPA group 2 survival	Inconclusive. Significant reduction in survivorship at stations S16, S3, S15, S1, S14, S6, and S8. See comments	Mysids tested in two batches. Survivorship markedly different between batches but very similar within batch. Suggests test organisms or test conditions differed between batches. Reference station mortality consistent with HPA mortality.
Solid phase sediment ST1-ST17	Amphipod survival	43-52% survival; consistent with HPA survival	Yes. Significant reduction in survivorship at all stations except S10.	Turbulence of overlying water from aeration may be responsible for mortality. Test may be inconsistent with sediment composition. Reference station mortality consistent with HPA mortality.
Solid phase sediment ST1-ST17	Polychaete survival	27-57% survival; consistent with HPA survival	Yes. Significant reduction in survivorship at all stations.	Turbulence of overlying water from aeration may be responsible for mortality. Test may be inconsistent with sediment composition. Reference station mortality consistent with HPA mortality.

2. Sediment elutriate samples exhibited moderate toxicity; however, the samples may not be amenable to long-duration tests because consistency for high sediment oxygen demand may stress the test organisms.

3. Bay water generally demonstrated no adverse effects.

4. Storm water samples generally showed no adverse effects except for station ST4.

5. The Bay reference stations exhibited toxicity very similar to HPA sediment toxicity.

6. Future bioassays should be conducted in stages to identify appropriate test conditions, appropriate test organisms, and appropriate test methods.

The results of the data usability analysis indicates the Phase 1B quantitative data collection efforts should focus on offshore sediments as sources of contaminates causing adverse ecological effects. Moreover, data collection efforts should concentrate on bioassay methods that examine the toxicity of the bioavailable fraction in the sediments. The lack of potential toxicity from storm water and bay water should be confirmed with another round of testing.

The ESAP mussel (*Mytilus californianus*) bioaccummulation data (HLA 1993) were collected in duplicate from 17 HPA stations and two reference stations south of HPA in San Francisco Bay. Baseline data were collected from mussels to determine background levels. The data collection procedures followed those used in the California state mussel watch program (RWQCB). The locations of the transplant stations were selected based on (1) proximity to areas of known or potential contamination, (2) proximity to groundwater seepage, surface water runoff, and storm sewer outfalls, (3) historical shoreline and berth uses, and (4) accessibility. After a 30-day exposure period, the tissue were analyzed for metals, semivolatile organic compounds, organochlorine pesticides, PCBs, and organotins.

The results demonstrated that metals and organotins were detected in nearly all samples, including the baseline tissues and reference station tissues. The results of the organic analysis demonstrated that a few organic compounds were detected in a few samples; organic compounds were not detected in the baseline tissues nor the reference station tissues. The results are summarized in Table 4-6. In general, the concentration of chemicals in tissues of mussels exposed at HPA were slightly elevated

above the reference station concentrations. The background contaminant tissue levels were the lowest, except for tributyltin, which was on the same order as the concentrations observed at HPA.

TABLE 4-6
CHEMICALS DETECTED IN THE ENVIRONMENTAL SAMPLING AND ANALYSIS PLAN MUSSEL TISSUE STUDY

Chemical	Units	HPA			Baseline		Reference Station		
		Freq. Detect.	Avg. Conc.	Max. Conc.	Freq. Detect.	Conc.	Freq. Detect.	RS1 Conc.	RS2 Conc.
Aluminum	mg/kg	33/33	605.76	1150.00	1/1	27.00	2/2	430.0	640.0
Antimony	mg/kg	10/33	0.16	0.58	0/1	-	0/2	-	-
Arsenic	mg/kg	33/33	5.65	9.20	1/1	1.3	2/2	4.6	5.7
Barium	mg/kg	33/33	2.40	3.90	1/1	1.7	2/2	1.6	2.5
Cadmium	mg/kg	33/33	5.80	9.10	1/1	3.6	2/2	5.4	7.3
Chromium	mg/kg	33/33	2.62	4.10	1/1	1.9	2/2	1.7	2.5
Cobalt	mg/kg	33/33	3.06	11.00	1/1	2.0	2/2	1.6	3.3
Copper	mg/kg	33/33	7.63	12.00	1/1	4.5	2/2	6.9	8.0
Iron	mg/kg	33/33	648.48	980.00	1/1	400.0	2/2	460.0	670.0
Lead	mg/kg	33/33	2.34	3.80	1/1	1.6	2/2	1.3	3.2
Manganese	mg/kg	33/33	18.21	26.00	1/1	14.0	2/2	11.0	19.0
Mercury	mg/kg	33/33	0.27	0.42	1/1	0.19	2/2	0.26	0.30
Molybdenum	mg/kg	33/33	0.51	0.96	1/1	0.35	2/2	0.33	0.55
Nickel	mg/kg	33/33	3.61	8.00	1/1	2.1	2/2	2.1	2.9
Selenium	mg/kg	33/33	2.28	4.00	1/1	2.4	2/2	1.5	1.9
Silver	mg/kg	33/33	0.22	0.54	1/1	0.09	2/2	0.24	0.26

TABLE 4-6
CHEMICALS DETECTED IN THE ENVIRONMENTAL SAMPLING AND ANALYSIS PLAN MUSSEL TISSUE STUDY
(Continued)

Chemical	Units	HPA			Baseline		Reference Station		
		Freq. Detect.	Avg. Conc.	Max. Conc.	Freq. Detect.	Conc.	Freq. Detect.	RS1 Conc.	RS2 Conc.
Tin	mg/kg	33/33	0.37	1.11	1/1	0.07	0/2	-	-
Vanadium	mg/kg	33/33	8.93	20.00	1/1	4.5	2/2	4.3	7.9
Zinc	mg/kg	33/33	168.79	300.00	1/1	120.0	2/2	140.0	220.0
Dibutyltin	µg/kg	24/33	74.77	350.00	1/1	99.0	1/2	33.0	-
Monobutyltin	µg/kg	10/33	13.08	58.00	1/1	12.0	0/2	-	-
Tributyltin	µg/kg	32/33	238.82	480.00	1/1	250.0	2/2	80.0	65.0
Phenanthrene	µg/kg	3/33	617.53	2000.00	0/1	-	0/2	-	-
Pyrene	µg/kg	3/33	666.01	2800.00	0/1	-	0/2	-	-
Aroclor 1254	µg/kg	3/32	79.36	647.00	0/1	-	0/2	-	-
Aroclor 1260	µg/kg	1/32	89.47	933.00	0/1	-	0/2	-	-
DDE	µg/kg	8/33	18.89	82.00	0/1	-	0/2	-	-
Fluoranthene	µg/kg	3/33	614.50	1900.00	0/1	-	0/2	-	-

DDE dichlorodiphenyldichloroethylene

HUNTERS POINT ANNEX
PHASE 1A ECOLOGICAL RISK ASSESSMENT

TASK 5 SUMMARY REPORT
IDENTIFICATION AND EVALUATION
OF CONTAMINANT MIGRATION PATHWAYS

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TASK 5.0 IDENTIFICATION AND EVALUATION OF CONTAMINANT MIGRATION PATHWAYS

Before the effects of a contaminant on an ecological receptor can be evaluated it is necessary to identify the contaminants that reach the potential receptor and to determine the way these contaminants are taken in by potential receptors. This section serves as a narrative to accompany the contaminant migration and exposure pathways depicted in the general conceptual ecological models for Parcels B, C, D, and E presented in Task 6. These models and descriptions represent a general first step in evaluating significant exposure pathways for receptor species at Hunters Point Annex (HPA).

5.1 CONTAMINANT MIGRATION PATHWAYS

The Phase 1A ecological risk assessment focuses on the migration pathways and exposure routes for ecological receptors that may be exposed to contaminants associated with the residual contamination in the soil, groundwater, storm water and offshore sediments. These pathways are presented in the conceptual models presented in Task 6 and are discussed below.

Soil, groundwater, and offshore contamination at HPA is largely the result of past waste disposal activities. Current activities that may be responsible for contamination of terrestrial and aquatic environments include possible illegal discharges from tenants of HPA buildings. Illicit discharges are currently being investigated by the Navy.

5.1.1 Soil

Contaminants in soils may be transferred to ecological receptors through direct contact or ingestion. Soil contaminants may also be transported to other media (for example, through leaching into the groundwater and into storm water and sewer lines). Soil contaminants may also be transported to the marine environment by surface runoff.

5.1.2 Groundwater

While animals do not generally burrow into the water table, some deep-rooted plants may contact groundwater at HPA. However, much of the shallow groundwater is highly saline as a result of tidal influence; only salt-tolerant plants are expected to contact groundwater. Groundwater directly discharges to the bay along the shore of HPA. Chemicals in groundwater may be transferred offshore, where they may become bound to the sediments or released to the water column. Utility lines such as storm drains may act as conduits for the groundwater.

5.1.3 Storm Water

While the magnitude of storm drain discharges has been greatly reduced since shipyard activities were curtailed in 1984, storm water is still a major contaminant migration pathway. The extent that these conduits are still responsible for contaminant migration depends on the amount of illegal discharges that are conveyed in the storm drains and sanitary sewers, the mass of hazardous substances transported to the storm system, and the extent of residual contamination in the storm drains and sewers that is resuspended and transported to the surface waters.

Contaminants associated with surface runoff are deposited directly into the storm water systems that flow directly into the bay. Sanitary sewers currently discharge to an off-site wastewater treatment facility. However, it is suspected that some sanitary sewers, possibly receiving discharges from small buildings on base, have not been separated from the storm water systems. In addition, breeches in both the storm water and sewer lines may allow contaminated soil and groundwater to be transported to the bay.

5.1.4 Sediments

Contaminant migration in the aquatic environment is driven by wind waves and tidal currents. The tide and wind-influenced surface waters disturb the top layer of the sediment, causing the chemical equilibrium at the sediment-surface water interface to be in constant flux. In addition, contamination can be transferred on and off Navy property by resuspended and deposited sediment.

5.2 EXPOSURE PATHWAYS

Ecological receptors may become exposed to a chemical through a number of routes. Plants and animals may contact the contamination medium directly and absorb contaminants through roots or dermal surfaces. Both plants and animals may become exposed through respiration, and animals may become exposed through ingestion of contaminated media (including contaminated food). The following sections will discuss these exposure pathways in more detail for terrestrial and aquatic receptors.

5.2.1 Terrestrial Exposure Pathways

Terrestrial receptors are potentially exposed to contaminants by (1) direct ingestion of soil, groundwater, or surface water, (2) ingestion of contaminated food, and (3) dermal contact with soil, groundwater, or surface water. Each of these pathways is discussed below.

5.2.1.1 Ingestion

Many organisms present at HPA either feed on decaying organic material in soil or burrow through soil. This pathway represents the entrance of soil contamination into the terrestrial food web as organisms such as earthworms, Botta's pocket gopher (*Thomomys bottae*), and California meadow vole (*Microtus californicus*) burrow through the soil. When these organisms are preyed upon, they possibly pass these contaminants up the food chain. Birds may indirectly ingest soils when foraging on insects and plants. Ingestion of soil is considered a major pathway for Parcel E since there is open terrestrial habitat with known soil contamination.

Primary, secondary, and top trophic level consumers may accumulate contaminants by consuming plants or animals carrying burdens of contaminants. This pathway is important to raptors such as the red-shouldered hawk (*Buteo lineatus*), red-tailed hawk (*Buteo jamaicensis*), and American kestrel (*Falco sparverius*) that have been observed foraging at HPA. Adverse effects to receptors and progeny may result once contaminant burdens reach critical concentrations. Contaminants transferred through the trophic pathway include toxic substances that are ingested and directly absorbed into tissues and organs, as well at the toxic metabolic by-products of these substances. This is a

significant pathway because there are numerous hazardous substances in the soil that may biomagnify in higher trophic levels at which adverse effects can occur.

Freshwater surface water present from the Parcel A seeps as well as general runoff can be ingested by terrestrial animals and absorbed by plants. Bird species such as house finch (*Carpodacus mexicanus*) or sparrows may drink from these seeps. However, because of the low levels of contamination detected in the Parcel A groundwater and the fact that general runoff is inconsistent, this is considered a minor pathway.

It is not likely terrestrial animals or plants contact the groundwater at HPA. The small mammals present are not expected to burrow to groundwater, which occurs at least 2 feet below ground surface.

5.2.1.2 Dermal Contact And Root Uptake

Terrestrial insects, reptiles, amphibians, and burrowing animals such as Botta's pocket gopher (*Thomomys bottae*) and California meadow vole (*Microtus californicus*) that live in soils may absorb contaminants through direct dermal contact with soil. This pathway is a concern in areas of Parcel B and E that have open habitat and known soil contamination.

Dermal contact with freshwater surface water is expected to be minor as a result of the limited amount and sporadic occurrence of the freshwater at HPA. This is considered a minor pathway.

Terrestrial biota are not expected to contact the groundwater directly at HPA; therefore, this pathway is incomplete.

The majority of plant species present at HPA (shrubs, grasses, or weeds with shallow root systems) are not expected to reach groundwater, and deep root systems likely extend to the top of the saline groundwater and then grow laterally above the groundwater. Therefore, this pathway is incomplete and not considered a concern.

5.2.2 Aquatic Exposure Pathways

Aquatic receptors are potentially exposed to contaminants by (1) direct ingestion of sediment and bay water, (2) ingestion of contaminated food, (3) dermal contact with sediments and bay water, and (4) direct respiration in bay water. Each of these pathways is discussed below.

5.2.2.1 Ingestion

The ingestion of contaminated sediments is a major exposure pathway. Several invertebrates including oligochaetes, polychaetes, crustaceans, copepods, isopods, insects, gastropods, bivalves, and chordates feed directly on or near the sediments. Indirect sediment ingestion may occur by shorebirds, fishes, and carnivorous invertebrates as they feed on sediment-dwelling organisms.

Ingestion of contaminated food is a major exposure pathway for the aquatic habitat. Species which contain residual levels of contaminants can transfer contamination up the food chain. The intertidal and subtidal organisms (oligochaetes, polychaetes, crustaceans, copepods, isopods, insects, gastropods, bivalves, and chordates) are a food source for carnivorous invertebrates, benthic fishes, pelagic fishes, and shorebirds. Birds such as western sandpiper (*Calidris mauri*), willet (*Catoptrophorus semipalmatus*), sanderling (*Calidris alba*), and dunlin (*Calidris alpina*) feed extensively on these organisms at low tide. Some of these predatory species may only contact contamination through ingestion of contaminated food.

Shorebirds and fishes may directly or indirectly ingest bay water off of HPA. This pathway is considered minor because it is an insignificant portion of contaminant exposure to both shorebirds and aquatic organisms when compared to ingestion of contaminated food and ingestion of sediments. Benthic organisms do not typically ingest water, so they are excluded from this pathway.

5.2.2.2 Dermal Contact

Benthic and epibenthic invertebrates such as amphipods, polychaetes, and fish species like the California halibut (*Paralichthys californicus*) are in constant contact with sediments. They are often buried or partially buried in the sediments. Therefore, dermal contact with sediments may be a major pathway. Shorebirds may also contact sediments with their beaks and feet when foraging. However, this dermal exposure route to sediments is considered minor in comparison with the ingestion pathways.

Organisms that spend all or a portion of their lives in water may be exposed by through dermal contact with Bay water. Larval forms of aquatic invertebrates as well as fish may absorb dissolved chemicals through dermal surfaces. Shorebirds and water birds also have long exposure to bay waters which dermal absorption may occur. However, dermal contact with Bay water is a minor pathway compared to ingestion.

5.2.2.3 Respiration

Respiration is a primarily pathway for fish such as anchovies (*Anchoa mitchiilli* and *Engraulis mordax*) and pacific herring (*Clupea harengus palasii*), that pass water over their gills and filter out oxygen and any other small dissolved molecules present in the water. Fish gills can filter out some toxins, but fish are likely exposed over time. Invertebrates such as species of bivalves also pass water over modified filter systems, but these systems are for feeding purposes instead of respiration. This pathway is considered minor compared to other pathways.

HUNTERS POINT ANNEX
PHASE 1A ECOLOGICAL RISK ASSESSMENT

TASK 6 SUMMARY REPORT
IDENTIFY MAJOR DATA GAPS AND
PROVIDE RECOMMENDATIONS FOR FUTURE WORK

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6.0 INTRODUCITON

The initial objective of Task 6 was to identify major gaps in information that would be addressed in the forthcoming HPA Phase 1B ERA. Two additional objectives were agreed upon by the Navy and the regulatory agencies: (1) prepare general ecological conceptual models and (2) select assessment and measurement endpoints. The conclusions of Task 6 consist of a discussion of the following: general ecological conceptual models, assessment and measurement endpoints, major data gaps, and recommendations for future work.

6.1 GENERAL ECOLOGICAL CONCEPTUAL MODELS

The general ecological conceptual models for Parcels B, C, D, and E presented in Figures 6-1 through 6-4 should be viewed together with the terrestrial and aquatic food webs (Figures 6-5 and 6-6). The data indicate that soils and offshore sediments act as both contaminant sinks and contaminant sources. These contaminants may adversely affect the structure and function of the aquatic ecosystem and, to a lesser extent, the terrestrial ecosystem, as the models depict.

In addition to acting as a reservoir of contaminants, the offshore sediments may be receiving contaminants from the groundwater, overland flow, storm water discharges, and other mechanisms, such as circulation and deposition of sediments from other bay sources and input from Yosemite Creek to South Basin. In addition, anoxia in the deeper sediments may prevent or retard contaminant degradation. Through wind, tide, and wave action, contaminants in sediments may become redissolved and/or resuspended into surface water, where they may be more readily bioavailable. Contaminants may slowly exfiltrate from the sediment pore water through the sediment-surface water interface into the water column. The chemicals in the water, however, would be expected to undergo significant dilution, potentially minimizing toxicity. Nonetheless, contaminants may still be transferred through the aquatic food web and cause adverse effects at higher trophic levels.

Soil in Parcels B, C, D, and E may be receiving contaminants from various IR, PA, and UST sites. Soil contaminants may be unavailable to biota because the high soil pH (7.7 - 8.4) generally shifts heavy metal equilibria toward the solid phase. Organic constituents may be bound to organic matter in the soil. However, terrestrial receptors may incidentally ingest or dermally contact contaminated

soil. Soil-associated contaminants can migrate through the soil to the groundwater or migrate to the storm sewers. Once reaching the groundwater, contaminants may migrate to the offshore sediments, at which point they become available to ecological receptors. The storm sewers also convey soil contaminants to the surface waters. These substances may precipitate to the sediments by various means.

Food chain transfer is a significant exposure pathway at HPA because much of the contamination in the soil and sediments has high bioaccumulative potential. These contaminants, principally the organochlorine insecticides, biomagnify through trophic transfer and may cause deleterious effects at higher trophic levels. Contaminants that bioaccumulate may also cause adverse ecological effects, such as acute and chronic toxicity, at lower trophic levels. It is difficult to predict which receptors may be affected, since site-specific bioavailability of most contaminants is not known.

6.2 GENERAL DISCUSSION OF FOOD WEBS

The terrestrial community at HPA is a simple ecosystem severely limited by several soil characteristics. The terrestrial habitat, primarily in Parcels A, B, and E, is dominated by a variety of opportunistic plant species. The richest terrestrial flora is found in the landscaped habitat of Parcel A. The habitat likely provides food for granivorous, omnivorous, and scavenging birds observed in Parcel A. Evidence of burrowing animals has also been found in Parcel A. These burrowing animals may provide a food source for the raptors that are consistently observed in Parcel A, sometimes perching in trees there. The American kestrel, which consumes birds, small mammals, and large insects, and the loggerhead shrike which feeds primarily on large insects have both been observed at HPA and would be expected to forage in Parcel A.

In Parcel B, ornamental plants dominate in the small landscaped area, and small brush is found near the shoreline of India Basin. No mammals have been observed in Parcel B.

The low abundance of flora in Parcel E results from well-drained, nutrient-deficient soil that is rich in heavy metals. The poor habitat in Parcel E limits the faunal diversity. As in Parcel A, flocking birds and burrowing animals may be a source of prey for raptors, like the peregrine falcon, American

kestrel and red-shouldered hawk, which have been observed forging in Parcel E. The importance of Parcel E as foraging grounds to these raptors is not known.

The aquatic ecosystem is more complex and diverse than the terrestrial community. Nutrient-releasing decaying organic matter and primary producers form the foundation of the aquatic food web and supply carbon to the primary consumers. Primary consumers, such as zooplankton, crustaceans (amphipods, isopods, and decapods), and annelids (polychaetes and oligochaetes) form an integral prey base for shore birds and fish, especially benthic gobies and other pelagic species. The gobies and pelagic fish are consumed by many species of piscivorous birds and fish and play central ecosystem roles. Top carnivores such as the California halibut, red-shouldered hawk, peregrine falcon, California brown pelican, and osprey may be particularly susceptible to bioaccumulated contaminants transferred up the food chain. Linkage between the terrestrial and aquatic food webs is provided by the red-shouldered hawk and the peregrine falcon, which prey on both terrestrial and aquatic birds.

6.3 ASSESSMENT AND MEASUREMENT ENDPOINTS

The guidance set forth by the U.S. Environmental Protection Agency (EPA) in the document, "Ecological Assessment of Hazardous Waste Sites: A Field and Laboratory Reference" was applied to Hunters Point Annex (HPA) to select assessment and measurement endpoints for the terrestrial and aquatic ecosystems. As defined by the agency (EPA 1989), assessment endpoints are formal expressions of the actual environmental values that are to be protected. Stressors that may adversely affect the assessment endpoints will be evaluated in the course of the ecological risk assessment. Assessment endpoints are environmental characteristics of biological and social significance, such as protection of sensitive habitats or organisms that if found to be significantly affected, may indicate a need to evaluate remedial alternatives. However, an assessment endpoint is useless unless it can be quantitively measured in some way. Therefore, an assessment endpoint must be unambiguously defined so that it can be evaluated either through direct quantitative measurement of the assessment endpoint itself or through direct quantitative measurement of measurement endpoints. Often, an assessment endpoint may not be directly measurable, consequently, use of measurement endpoints to predict effects on assessment endpoints is common. A measurement endpoint is a quantifiable

environmental characteristic that is directly related to the assessment endpoint. A model is generally used to describe the predictive relationship between the measurement and assessment endpoints.

Assessment and measurement endpoints were selected by applying the EPA criteria to the conditions at HPA (EPA 1989). The assessment endpoints adopted reflect the ecological and social relevance of the organisms and food web interactions in the terrestrial and aquatic ecosystems. Assessment endpoints were chosen to be clearly definable and quantifiable through measurement endpoints. Measurement endpoints were selected based their ability to predict effects on assessment endpoints, their low natural variability, and their ability to be readily quantified. Furthermore, ecological and toxicological information gathered in the ERA tasks 1.0, 2.0, 3.0, and 4.0 was incorporated into endpoint selection. These site-specific considerations include the occurrence, abundance, feeding behavior, and susceptibility of various receptors, the appropriate or relevant and applicable requirements, spatial and temporal scales of exposure, and exposure pathways. The assessment and measurement endpoints for the terrestrial and aquatic ecosystems at HPA and the specific rationale for selection of each, describing considerations of habitat use at HPA, trophic level, exposure pathway, biological significance, and social and economic value, are presented in detail in the narrative below and summarized in Table 6-1.

6.3.1 Terrestrial Endpoints

The protection of habitat for the raptorial species, peregrine falcon (*Falco peregrinus*), American kestrel (*Falco spaverius*), red-shouldered hawk (*Buteo lineatus*), and loggerhead shrike (*Lanius ludovicianus*) living or foraging at HPA with the ultimate goal of protecting regional populations of these birds, was selected as the terrestrial assessment endpoint. All of these birds have been observed using habitat at HPA, and the American kestrel and red-shouldered hawk have both been observed foraging in Parcels A and E; they may also nest in the trees in Parcel A. The precise residence status of these species at HPA is not known; however, the potential exists for these species to use HPA habitat all year. Although the peregrine falcon and red-shouldered hawk forage over large areas, they have been observed at HPA and may consume prey living at the facility. The American kestrel and loggerhead shrike have smaller foraging ranges, and if they reside on base, prey from HPA may compose a large portion of their diet.

Each of these birds is a high trophic level carnivore that, through bioaccumulation in the terrestrial food chain, has the potential to be adversely affected by contaminants in the terrestrial environment, especially pesticides and bioconcentrating heavy metals such as mercury, cadmium, and lead. The primary exposure pathway of concern to these raptors is the ingestion of contaminated prey at HPA. Prey, in turn, accumulate pollutants through a variety of pathways, the most important of which is the ingestion of contaminated food and soil.

As depicted in the terrestrial and intertidal assessment and measurement endpoint food webs (Figures 6-5 and 6-6), these raptors consume prey from a variety of trophic levels. The peregrine falcon feeds primarily on birds. The granivorous mourning dove, the granivorous and scavenging rock dove, and the omnivorous northern mockingbird and red-winged blackbird are potential peregrine falcon prey species that are abundant in the terrestrial ecosystem at HPA. Some other birds in HPA's aquatic ecosystem that could serve as prey for the peregrine falcon are shorebirds and ducks.

The red-shouldered hawk shares much of the peregrine falcon's prey base, with the addition of small mammals like the Botta pocket gopher, the California meadow vole, and the California black-tailed hare. Black-tailed hares have been observed consistently at HPA; however, the residential status of the gopher and vole is not as definite. Evidence of a burrowing animal, probably a mammal, has been seen in nearly every portion of Parcel E and in some portions of Parcel A, both areas where raptors have been observed foraging. Because the peregrine falcon and the red shouldered hawk may also forage on the margins of the aquatic ecosystem for shorebirds, they represent an important link between the terrestrial and aquatic systems at HPA. The American kestrel also consumes small birds and mammals but also feeds on large insects, which comprise the main prey base for the loggerhead shrike (Figure 8). In seeking to protect the habitat of these raptor populations, some protection for their prey base of birds, mammals, and terrestrial invertebrates is simultaneously afforded.

The raptors' value to society is evidenced by their protection under various federal and state regulations. The peregrine falcon is a state and federally endangered species whose once declining populations have been recently invigorated through an intensive captive breeding and release program under the Endangered Species Act. Protection of local populations of peregrine falcons is crucial to the continued success of this breeding and release program. Loggerhead shrike populations have recently been found to be declining for unknown reasons. Consequently, shrikes are considered a

California species of special concern. The red-shouldered hawk is listed on the Audubon Society's Blue List of avian species of special concern. Furthermore, populations of all of these species have important social and economic value through the interest of bird-watchers and visitors to local and state parks who hope to observe these birds.

Measurement endpoints that may be used to evaluate potential effects of terrestrial contaminants on populations of raptors may include tissue residue analysis and bioaccumulation studies of prey at HPA. These data would be used in a food chain bioaccumulation model to predict potential adverse effects of terrestrial contaminants of concern on assessment endpoints. This model would employ bioconcentration factors for these contaminants from the scientific literature to predict potential contaminant levels in HPA raptors based on data obtained through these tissue analysis and bioaccumulation studies. The predicted contaminant levels in raptors would be compared to levels causing deleterious biological effects to determine the potential for contaminants at HPA to pose significant hazards to raptors using habitats at HPA.

Tissue analysis and bioaccumulation studies could be performed on a variety of organisms that represent potential prey for the birds of prey to be protected as assessment endpoints. Data from these studies would be used to develop food web transfer models to predict potential adverse effects on raptors. To measure contaminant bioconcentration on the lower trophic levels, plant bioaccumulation studies could be performed to predict the potential contaminant content in the food of herbivorous prey species like the mourning and rock doves, the gopher, the vole, and the hare. To measure bioconcentration at higher trophic levels, tissue residue or bioaccumulation studies on earthworms, soil arthropods, and soil isopods could be performed. These studies would provide direct measurements of the potential contaminant concentrations of prey consumed by the loggerhead shrike.

Indirect measurements of potential contaminant concentrations of prey consumed by raptors could be made through a food web transfer model incorporating the soil invertebrate tissue residue and bioaccumulation data. The food web transfer model would trace the potential biomagnification of contaminants from the soil invertebrates, to carnivorous avian species, and finally to the raptors consuming these avian prey. Some of these carnivorous avian prey species are the northern mockingbird, red-winged blackbird, and western meadowlark. Comparison of the potential

contaminant content of these prey with adverse effects reported at dietary concentrations would indicate whether raptors at HPA are exposed to excessive contaminant levels through their prey. Another source of information on potential contaminant levels in the diet of the raptor species would come from tissue residue and bioaccumulation studies performed on mourning and rock doves and small mammals.

6.3.2 Aquatic Endpoints

Because the aquatic ecosystem is more complex than the terrestrial ecosystem at HPA, assessment endpoints representing different taxa were chosen. The protection of habitat for the avian species California brown pelican (*Pelecanus occidentalis californicus*), double-crested cormorant (*Phalocrocorax auritus*), Barrow's goldeneye (*Bucephala islandica*), and great blue heron (*Ardea herodias*) living or foraging at HPA, with the ultimate goal of protecting regional populations of these birds, was selected as one assessment endpoint in the aquatic ecosystem. The protection of local populations of the California halibut and the goby species such as yellowfin goby (*Acanthogobius flavimanus*), arrow goby (*Clevelandia ios*), cheekspot goby (*Ilypnus gilberti*), bay goby (*Lepidogobius lepidus*), and of the mollusks and crustaceans of the intertidal and subtidal areas was also selected as an assessment endpoint for the aquatic ecosystem. Some of these species are important prey species for a wide variety of bay fish and birds. Although several species of pelagic fish, like Pacific herring (*Clupea pallasi*), northern anchovy (*Engraulis mordax*), threadfin shad (*Dorosoma petenese*), and smelt species, are consistently observed in HPA waters, are pivotal components of the intertidal and subtidal communities there, and are commercially and socially important, these fish were not selected as assessment or measurement endpoints. Their high degree of mobility lessens their potential susceptibility to contaminant conditions at HPA.

All of the avian assessment endpoint species are consistently observed using habitat at HPA, principally in the Parcel E shoreline and the open water off Parcels B, C, D, and E. The pelican and the goldeneye are both migratory birds that are seasonal residents or visitors at HPA. The pelicans arrive in the bay area to nest during the spring and remain into early fall, whereas the goldeneye is present mainly during winter months to feed. Furthermore, the pelican can range over large areas in search of prey. Despite their seasonal presence and potential mobility, the biological and social value of the pelican and goldeneye is significant enough to warrant designation as assessment endpoints, as

described below. The precise residence status and mobility of the great blue heron at HPA is not known, although it has been consistently observed there during the winter. The cormorants appear to be year-round residents, making their potential exposure duration long.

The pelican, cormorant, and heron are high trophic level carnivores whose susceptibility to deleterious effects caused by bioconcentrated contaminants is well documented. The primary exposure pathways of concern for these birds would be ingestion of contaminated prey and sediment. As shown in the intertidal food web (Figure 6-6), the primary prey bases for these birds are fishes, which are known to bioaccumulate pollutants from both sediment and water and from ingestion of contaminated food.

In parallel to the selection of terrestrial endpoints, these piscivorous birds were designated as aquatic assessment endpoints not only for their susceptibility, but also for their biological importance as high level consumers and for the protective advantages conferred on their prey as a result of this endpoint designation. As depicted in the intertidal food web (Figure 6-6), the pelican feeds on a variety of fish, most notably pelagic and higher trophic level carnivorous fish. The cormorant and the heron's diets of pelagic fish and gobies, two taxa abundant at HPA, is supplemented occasionally with invertebrates. These fish populations would benefit from efforts made to protect habitats for these predatory birds. Furthermore, some of these fish and invertebrate prey are also assessment endpoints themselves, as discussed below.

Barrow's goldeneye, though not a high level carnivore like the pelican, cormorant, or heron, is biologically important at HPA because it functions as both predator and prey. The prey base of Barrow's goldeneye differs from those of the pelican, cormorant, and heron, because it consumes primarily mollusks and crustaceans, two taxa forming a critical foundation for the health of aquatic ecosystems. Abundant potential mollusk and crustacean prey species are presented in Table 6-2. These invertebrates are known to bioaccumulate contaminants from both the sediment and the water column and represent important vehicles for bioconcentration in the aquatic food chain. Once again, protecting habitat for Barrow's goldeneye would also confer some protection on its prey. Moreover, ducks like the goldeneye are preyed upon by raptors.

These birds' value to society is evidenced by their protection under various California and federal regulations. The California brown pelican, like the peregrine falcon, is a federal and state endangered species whose once declining populations are now making a significant resurgence in numbers. Protection of habitats for local populations of pelicans is integral to the reestablishment of pelicans in California. The double-crested cormorant and Barrow's goldeneye are both designated under California regulation as species of special concern to be monitored by the California Department of Fish and Game (CDFG). Although not protected under any regulations other than that for California native species, the great blue heron, as well as the pelican, cormorant, and goldeneye, is socially and economically important to bird-watchers and to visitors to local and state parks hoping to see these birds. Additionally, the goldeneye is a harvested game species, making it economically and socially important as a result of human consumption and the sale of game permits.

The protection of local populations of the benthic California halibut and the goby species such as yellowfin goby (*Acanthogobius flavimanus*), arrow goby (*Clevelandia ios*), cheekspot goby (*Ilypnus gilberti*), bay goby (*Lepidogobius lepidus*), was chosen as an aquatic assessment endpoint for several reasons. First, both species have been caught in HPA waters, and the gobies occur at HPA in abundance. Second, these species' relative immobility makes them more susceptible to potential effects of contaminants at HPA, especially, for example, when compared to more mobile pelagic fish. Halibuts spend much of their lives in bays and estuaries, but can migrate to nearshore ocean coastal waters to spawn during the spring. Gobies live in burrows and do not have a large range.

Third, these benthic species's susceptibility is further increased as a result of their ability to bioaccumulate contaminants from a variety of pathways, most importantly, through ingestion of contaminated food and sediment and through contact with contaminated sediment. Both of these fish live and feed on the bay floor. The halibut partially buries itself in the sediment as camouflage, and the gobies share burrows with burrowing polychaetes and other invertebrates. Because these fish are benthic and spend most of their lives in, on, or very close to the bay floor sediments, exposure to any contaminants in the sediment would be significant. Furthermore, the halibut, as a high trophic level carnivore, consumes other benthic species such as bottom-dwelling flat fish, crustaceans, and annelids, all of which are prey that have high exposure potential due to their intimate association with bay sediments.

Fourth, both halibut and gobies play important ecological roles in the intertidal and subtidal communities at HPA (Figure 6-6). Halibuts are high trophic level carnivores that would be susceptible to contaminants bioconcentrated in the food chain. Conversely, gobies play a pivotal role in the subtidal and intertidal communities as common prey species for many carnivorous piscivorous fish and birds. Halibuts are an important commercially harvested fish. Gobies are not harvested; however, their economic importance lies in their central role as prey, supporting populations of many other harvested fish.

The protection of local populations of intertidal and subtidal mollusks, crustaceans, and annelids was selected as another assessment endpoint to complement the aquatic assessment endpoints representing other taxa. Table 2 presents the top 25 percent most abundant intertidal and subtidal invertebrates caught at HPA and describes important aspects of their feeding and life modes that were incorporated into the analysis of these invertebrates' importance as endpoints at HPA. Although occupying a low trophic level (Figure 6-6), these invertebrates, especially many filter-feeding and sediment-consuming mollusks, crustaceans, and annelids, bioconcentrate contaminants to levels several times greater than those in sediment and water, primarily as a result of feeding behavior. The major exposure pathways for these species are ingestion of contaminated food and ingestion of contaminants from water and sediment.

Like gobies, the intertidal mollusks, crustaceans, and annelids also perform a central ecosystem function as common prey species, as depicted in the intertidal food web (Figure 6-6). Mollusks, such as clams, mussels, and other bivalves, crustaceans, such as amphipods, isopods, copepods, and decapods (crabs and shrimp), and annelids, such as oligochaetes and polychaetes, are consumed by many bird species, including shorebirds, waders, and diving ducks, in addition to those discussed above as assessment endpoints. Some of these birds are themselves preyed upon by other higher trophic level consumers, such as the peregrine falcon and red-shouldered hawk.

Many commercially important fish species, including pelagic fish, flatfish, and other harvested bay fishes, consume crustaceans, especially decapods, amphipods, isopods, and copepods. Preservation and improvement of the productivity of the bay fishery requires protection of these invertebrate communities, making them biologically, economically, and socially notable. Several of these invertebrates, including crabs, shrimps, mussels, oysters, and others, were once extensively harvested

themselves, and efforts to protect these invertebrates may enable a viable fishery to redevelop in the future.

Additionally, these invertebrates were designated as assessment endpoints because their relative immobility makes them good indicators of contaminant conditions at HPA. This fact, combined with the existence of standardized effects tests for these invertebrates, also enables them to be used as indicator species in measurement endpoints. Measurement endpoints that could be used to gauge effects of sediment contaminants on assessment endpoints are toxicity, growth and reproduction, bioaccumulation, and tissue residue tests on bivalves, amphipods, and fish (gobies and, possibly, flat fish). Toxicity, growth, and reproduction tests would provide data on the potential direct effects of contaminants on invertebrate and fish assessment endpoints.

Bioaccumulation and tissue residue studies would provide data with which to build models to predict potential effects on higher trophic level assessment endpoints. The bioaccumulation and tissue residue studies on mollusks and crustaceans would be used to model food chain bioaccumulation potential for contaminants. These data on mollusks and crustaceans would also predict contaminant concentrations in the diets of predators of these species, such as shorebirds and carnivorous fishes, allowing extrapolation to potential effects on higher trophic level assessment endpoints using bioconcentration factors and contaminant effect levels.

6.4 MAJOR DATA GAPS

In order to perform a qualitative ecological risk assessment at HPA, additional information is needed regarding: (1) the relationship between onshore contamination and offshore contamination, and (2) sediment characteristics which influence bioavailability. These two areas are described in the following sections:

6.4.1 Relationship Between Onshore Contamination and Offshore Contamination

The following data gaps concerning the extent of contamination of onshore areas at HPA and the relationship between onshore sources and offshore contamination are described below:

- The extent of contamination around soil hot-spots and the probability of a complete contaminate migration pathway between the hot-spots and the offshore sediments.

- Whether oil observed and strong petroleum odor detected offshore of IR3, IR2, and IR1 are originating from HPA sources.

- Whether identified sediment hot-spots can be attributed to HPA activities.

- The contribution of contamination from Yosemite Creek, Islais Creek, Pier 80, combined sewer overflows, and other land and bay sources to HPA underwater property.

- The contribution of contamination from groundwater at HPA to ecological receptors, both from on-site springs, and on-site downgradient groundwater flow to the Bay.

6.4.2 Sediment Characteristics

Information is needed to provide a better understanding of sediment characteristics such as oxygen, ammonia, and sulfide levels affect the function of the biotic communities and contaminant toxicity. This knowledge will provide the means to better understand how contaminants may affect the structure and function of the biotic ecosystem and the probability of an adverse ecological effect. The following data gaps have been identified:

- Sediment characteristics such as pH, total organic carbon, grain size, oxygen, ammonia and sulfide levels that affect sediment toxicity and whether contaminants are bioavailable to key communities in the aquatic ecosystem.

- The physical, chemical, and biological factors such as temperature and partitioning coefficients affecting the equilibrium of contaminants between the solid phase (particle) and liquid phase (pore water).

- The degree to which sediment-associated contaminants are available to primary receptors at HPA, such as benthic macroinvertebrates that ingest, contact, or respire sediment particles and/or sediment pore water.

- The degree to which sediment-associated contaminants are available to receptors such as demersal fish and diving birds that incidentally ingest sediment particles.

- The major mechanisms responsible for removal of PCBs, PAHs, and organotins from offshore sediments.

- The extent of transfer of contaminants to higher trophic level consumers such as birds and carnivorous fish.

- The adverse effect of contaminants on benthic macroinvertebrates and the secondary effect on consumers of the macroinvertebrates.

- The adverse effect of contaminants on higher trophic level consumers such as birds and carnivorous fish.

- The ratio of aquatic to terrestrial prey in the red-shouldered hawk and peregrine falcon.

- Duration of exposure to contaminants by key receptors.

- Site specific bioaccumulation factors.

- Site specific contaminant tissue levels in terrestrial receptors.

6.5 RECOMMENDATIONS

To address the gaps in information concerning the extent of contamination of onshore areas at HPA and the relationship between onshore sources and offshore contamination.

The following information should be collected:

- The ambient level concentrations should be identified. Characterize risk at areas above ambient levels.

- Develop risk factors for the COPCs. Rank the COPCs to determine the chemicals of concern driving the ecological risk.

- Groundwater may serve as a pathway by which contaminants could reach the intertidal sediments. Intertidal, subtidal, groundwater, and soil data should be examined to determine whether the intertidal area, subtidal area, groundwater, and soil share similar fingerprints.

- Surface water and sediment samples should be collected from Yosemite Creek to assess whether the creek is a significant source of the contaminants detected in South Basin. Similarly, surface water and sediment data from Pier 80 and Islais Creek channel should be evaluated to assess whether they are a significant source of the contaminants found at HPA.

- Further discussion between the regulatory agencies and the Navy is required to address the issue of intra-bay contaminant transport and deposition to HPA underwater property.

- Information collected from the reference stations used in the ESAP program and aquatic surveys conducted in the HPA Phase 1A indicate that "clean" sediment sites in the bay may be as contaminated as HPA. Physical, chemical, and biological information about San Francisco Bay reference stations should be evaluated to provide an overall picture of the status of "clean" sites around the bay. This information may help provide a context in which HPA may be viewed to determine which HPA contaminants should be addressed in greater detail.

The following recommendations are offered to address information that is needed to identify and evaluate the major sediment characteristics at HPA that affect the aquatic ecosystem and the potential for contaminants at HPA to induce adverse ecological effects.

- Numerous contaminants in soil, sediment, and groundwater at HPA may cause an adverse effect to ecological receptors. To provide a more detailed understanding of how the contaminants may affect receptors, conceptual ecological models should be prepared for each class of the contaminants at HPA, including trace metals, PAHs, PCBs, organochlorine pesticides, and organotins.

- The existing HPA database should be sorted for data on physical and chemical factors such as grain size, pH, organic carbon, oxygen, sulfide, and ammonia levels that affect bioavailability of sediment-associated contaminants. Existing literature should be reviewed to identify other factors at HPA that may affect contaminant bioavailability. Additional physical, chemical, and physical data should be collected from HPA sediments. Simple models describing how these factors affect contaminant bioavailability at HPA should be prepared. The bioavailability models should be used to predict the bioavailability of the contaminants in the offshore sediments.

- Equilibrium of contaminants between the solid phase and liquid phase of sediments should be investigated using simple laboratory techniques to determine how contaminants are partitioning in the offshore sediments at HPA. These results should be used identify the appropriate laboratory tools, such as bioassays, for measuring adverse effects of contaminants on ecological receptors.

- Bioavailability predictions should be confirmed by collecting additional empirical data, such as bioassays coupled with chemical analyses and regression techniques to determine the contaminants responsible for adverse responses of organisms.

- The duration of exposure of key receptors to contaminants should be determined using available literature information and existing information about the ecosystems at HPA.

- To the extent feasible, site specific tissue/organ/body burden data should be collected from species thought to be important links in the aquatic food web. This could be accomplished by using field or laboratory techniques, resulting in relevant contaminant bioconcentration factors. Body burden of contaminants should be coupled with bioconcentration factors to predict contaminant concentrations expected in key high trophic level receptors.

- Develop exposure profiles for each COPC.

- Predicted tissue/organ/body burdens of contaminants in high trophic level receptors should be compared to literature information to identify potential adverse effects. This information should be integrated with contaminant exposure and relevant life history information to predict the probability of adverse effects occurring.

- Simple laboratory techniques should be used to determine the degradation rates of biodegradable contaminants such as PCBs, PAHs, and organotins.

For the HPA terrestrial ecosystem, information on the burdens of contaminants in mammal and bird receptors eaten by raptors should be collected. This could be accomplished using field and laboratory techniques. In addition, more detailed information on the diet of raptors at HPA is needed to determine the proportion of terrestrial-based and aquatic-based diets.

REFERENCES

U.S. Environmental Protection Agency (EPA). 1989. "Ecological Assessment of Hazardous Waste Sites: A Field and Laboratory Reference" Environmental Research Laboratory, Corvallis, Oregon. March

FIGURE 6-1

PARCEL B GENERAL CONCEPTUAL ECOLOGICAL MODEL

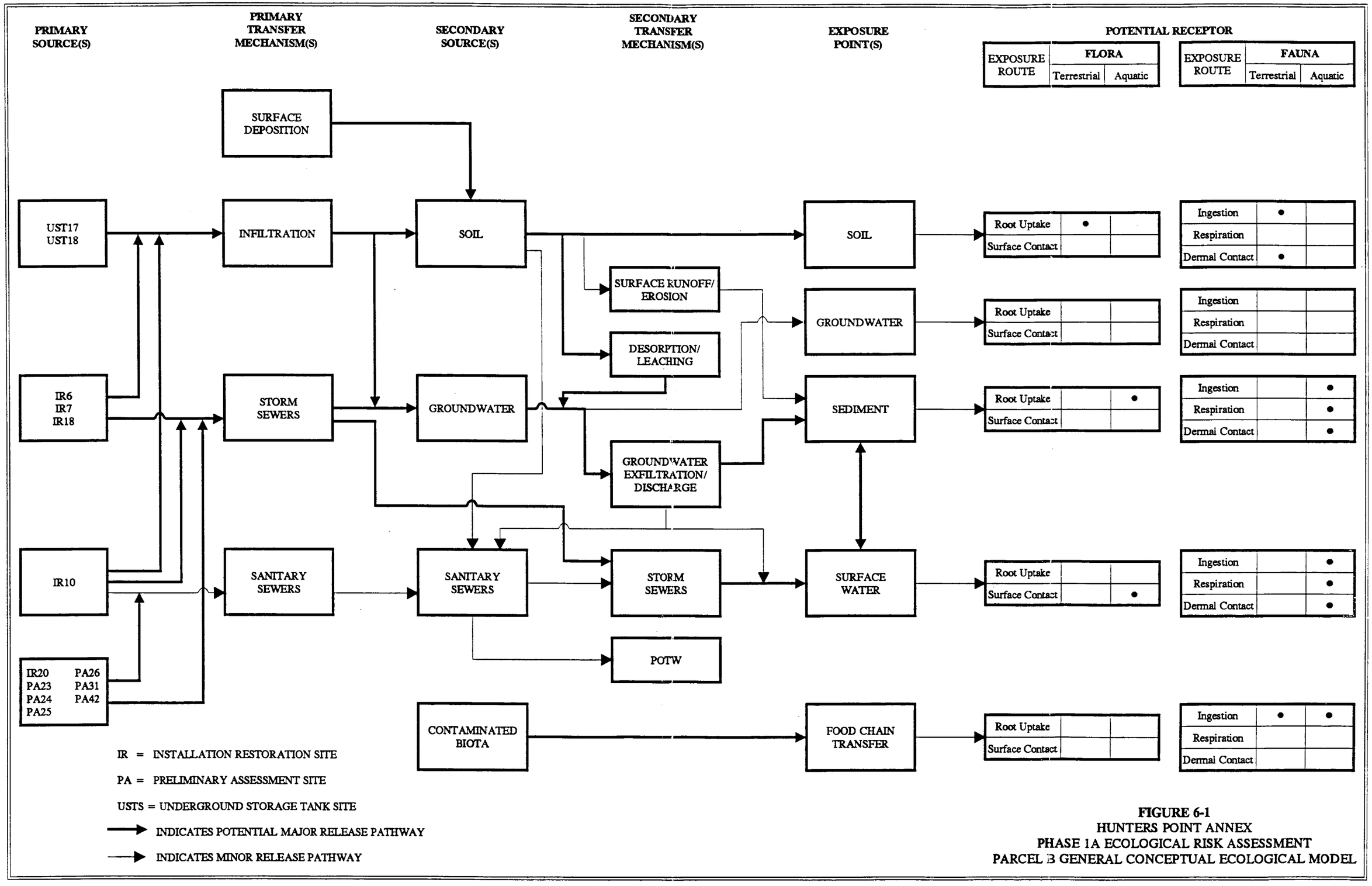

FIGURE 6-1
HUNTERS POINT ANNEX
PHASE 1A ECOLOGICAL RISK ASSESSMENT
PARCEL B GENERAL CONCEPTUAL ECOLOGICAL MODEL

FIGURE 6-2

PARCEL C GENERAL CONCEPTUAL ECOLOGICAL MODEL

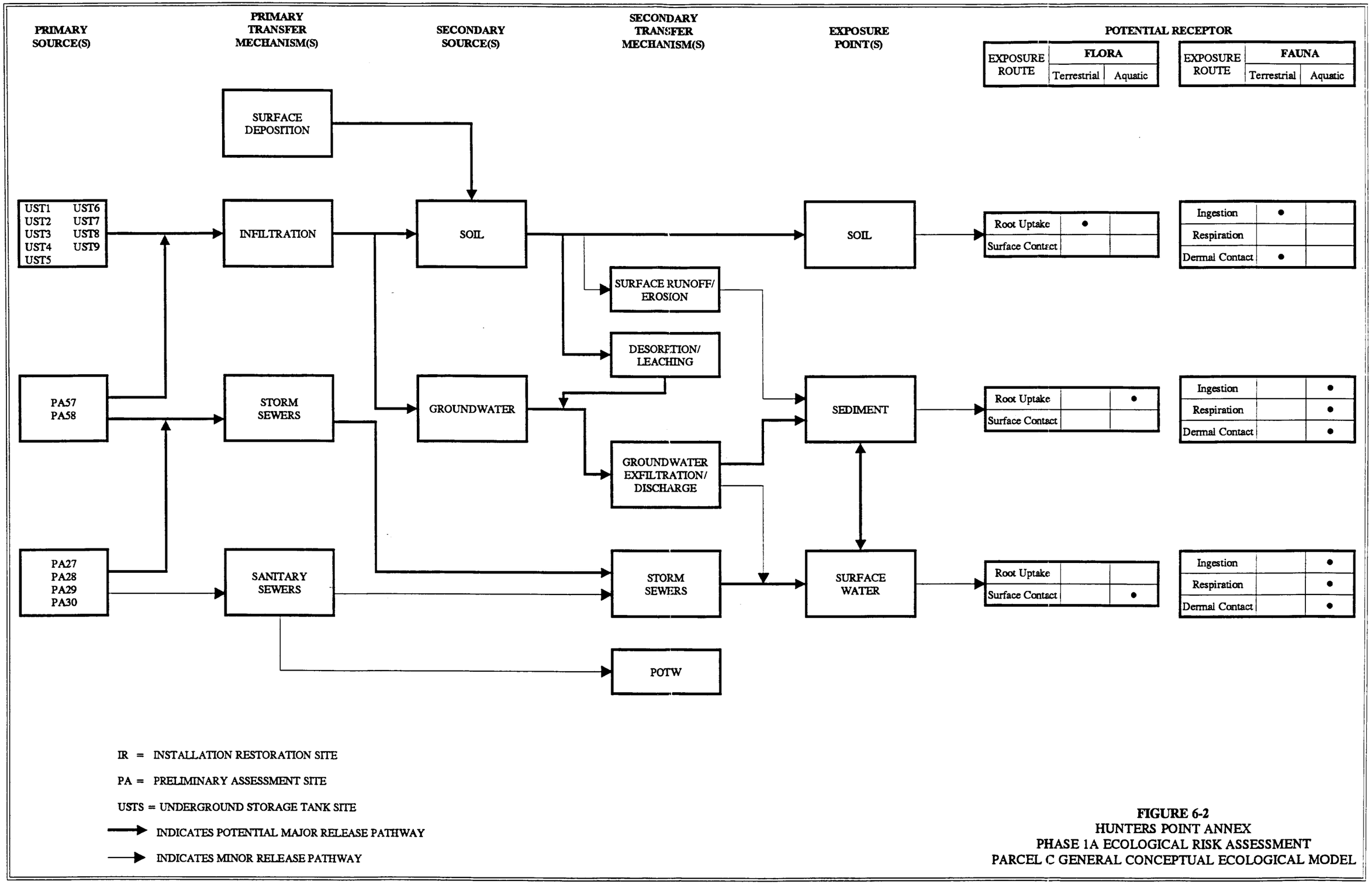

FIGURE 6-2
HUNTERS POINT ANNEX
PHASE 1A ECOLOGICAL RISK ASSESSMENT
PARCEL C GENERAL CONCEPTUAL ECOLOGICAL MODEL

FIGURE 6-3

PARCEL D GENERAL CONCEPTUAL ECOLOGICAL MODEL

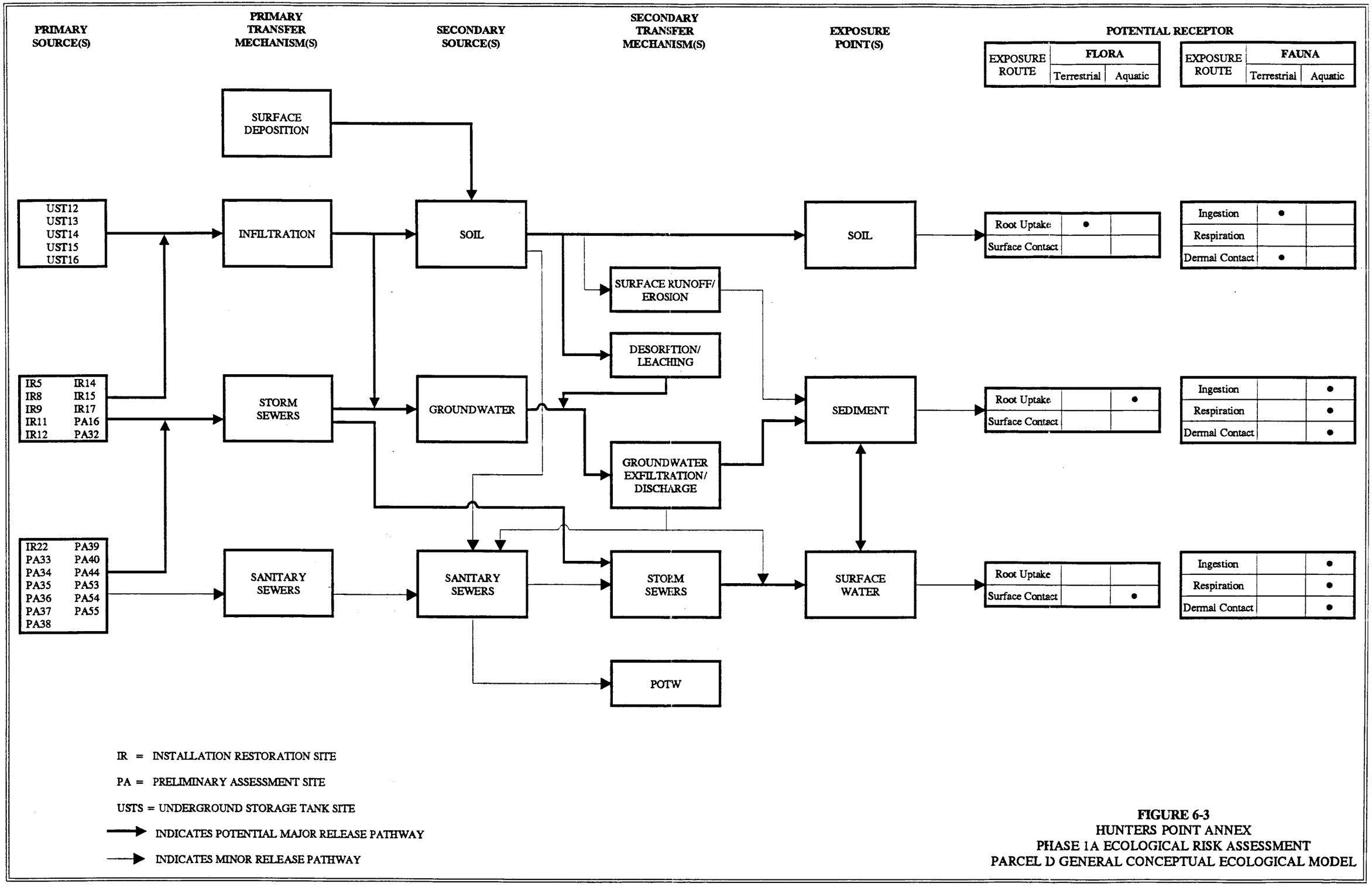

FIGURE 6-3
HUNTERS POINT ANNEX
PHASE 1A ECOLOGICAL RISK ASSESSMENT
PARCEL D GENERAL CONCEPTUAL ECOLOGICAL MODEL

FIGURE 6-4

PARCEL E GENERAL CONCEPTUAL ECOLOGICAL MODEL

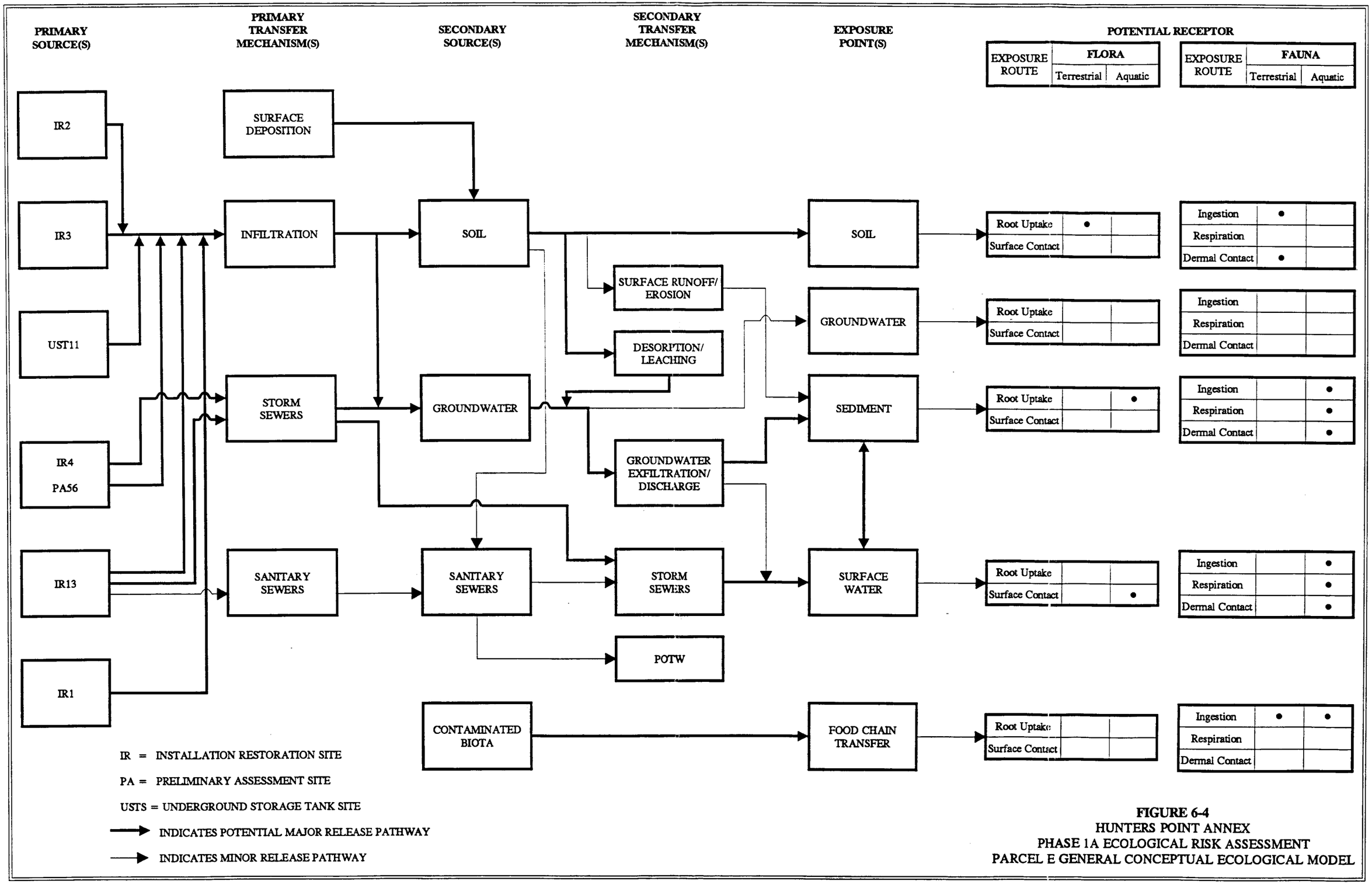

FIGURE 6-4
HUNTERS POINT ANNEX
PHASE 1A ECOLOGICAL RISK ASSESSMENT
PARCEL E GENERAL CONCEPTUAL ECOLOGICAL MODEL

FIGURE 6-5

SIMPLIFIED TERRESTRIAL FOOD WEB
EMPHASIZING ASSESSMENT AND MEASUREMENT ENDPOINTS

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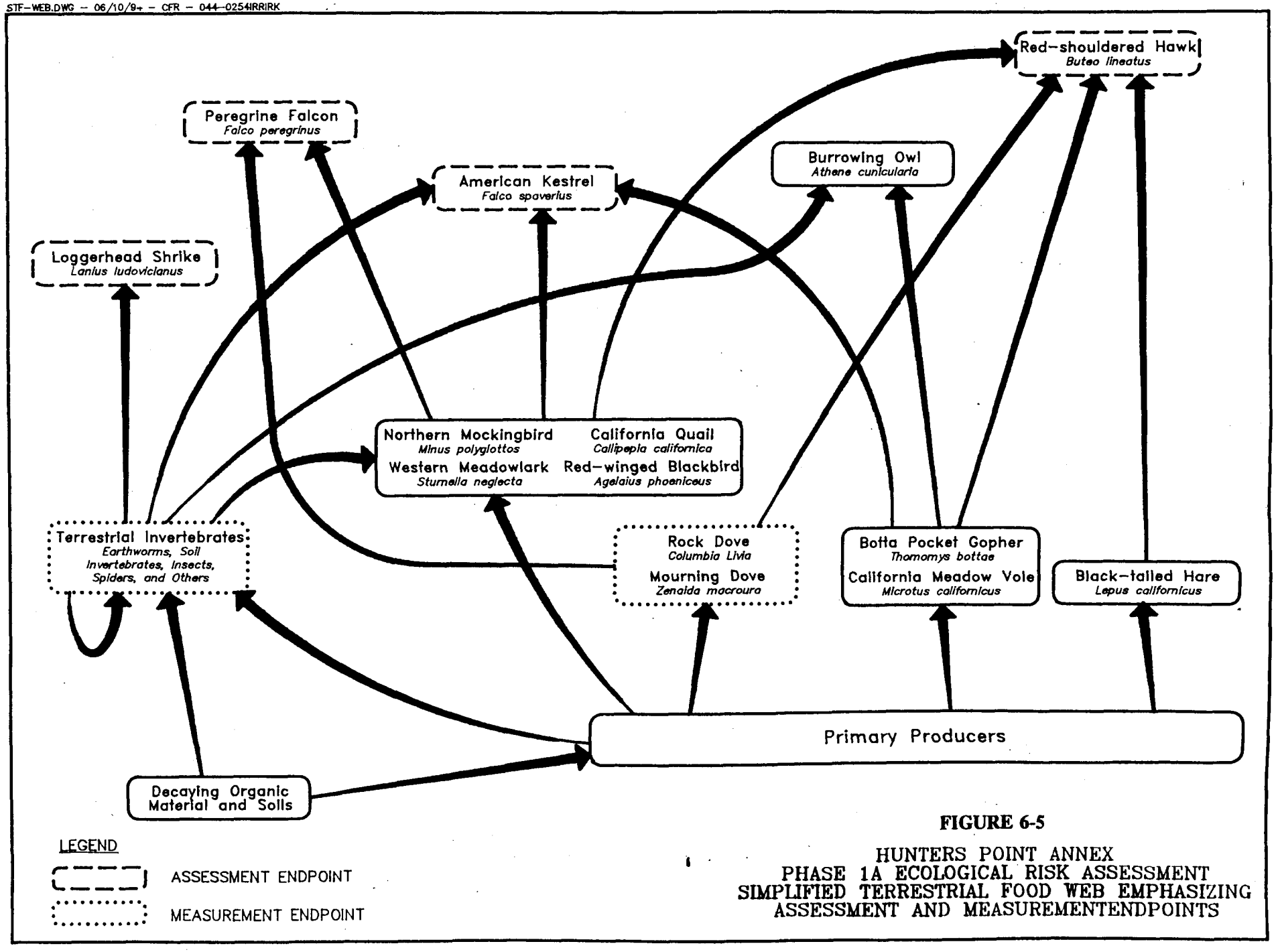
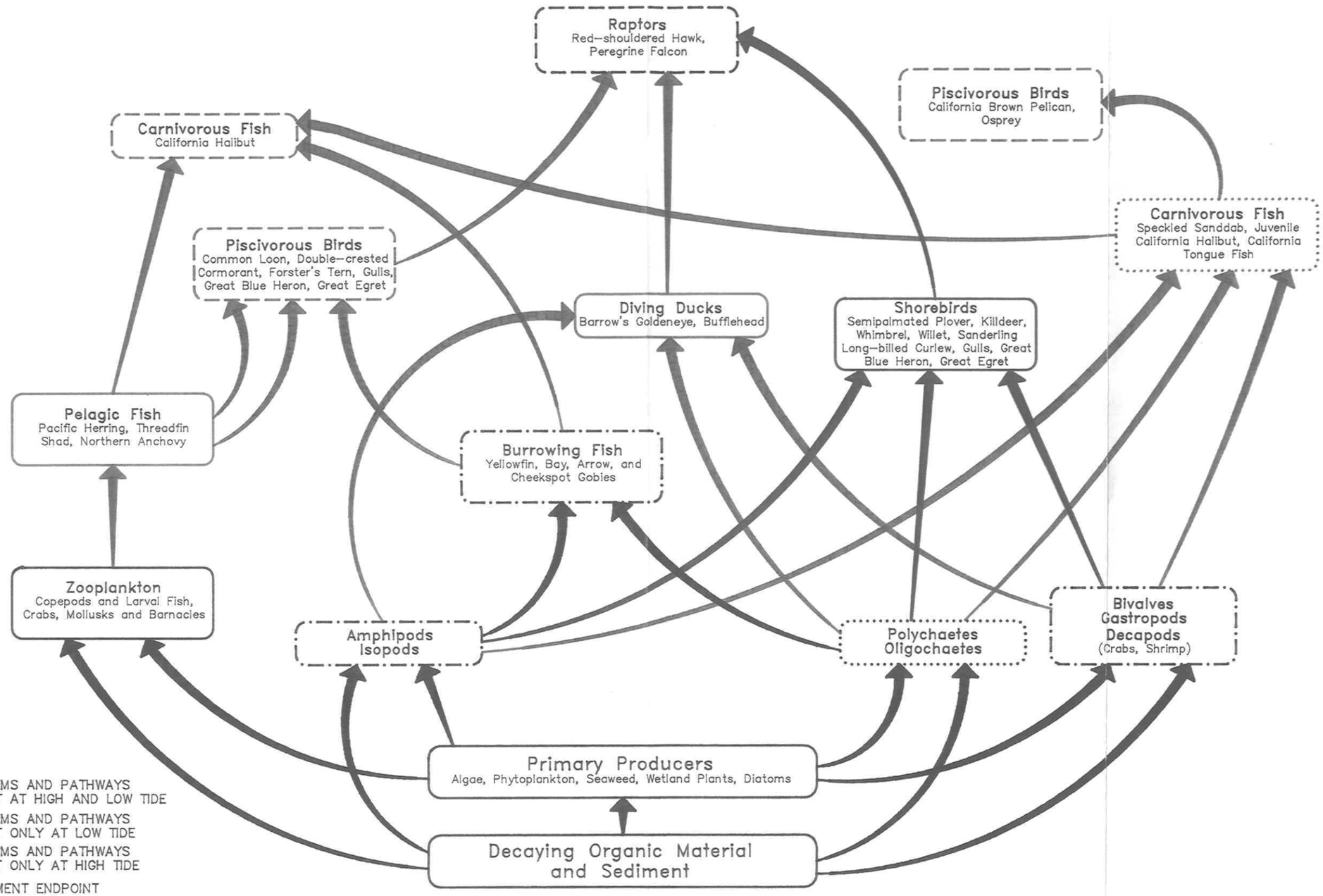


FIGURE 6-5

HUNTERS POINT ANNEX
PHASE 1A ECOLOGICAL RISK ASSESSMENT
SIMPLIFIED TERRESTRIAL FOOD WEB EMPHASIZING
ASSESSMENT AND MEASUREMENTENDPOINTS

FIGURE 6-6

SIMPLIFIED INTERTIDAL FOOD WEB
EMPHASIZING ASSESSMENT AND MEASUREMENT ENDPOINTS



LEGEND

- ORGANISMS AND PATHWAYS PRESENT AT HIGH AND LOW TIDE
- ORGANISMS AND PATHWAYS PRESENT ONLY AT LOW TIDE
- ORGANISMS AND PATHWAYS PRESENT ONLY AT HIGH TIDE
- ASSESSMENT ENDPOINT
- MEASUREMENT ENDPOINT
- ASSESSMENT AND MEASUREMENT ENDPOINT

NOTES: SCIENTIFIC NAMES FOR FOOD WEB ORGANISMS APPEAR IN THE TEXT.

PREDOMINANT FOOD WEB INTERACTIONS ARE REPRESENTED.

FIGURE 6-6

HUNTERS POINT ANNEX
PHASE 1A ECOLOGICAL RISK ASSESSMENT
SIMPLIFIED INTERTIDAL FOOD WEB EMPHASIZING
ASSESSMENT AND MEASUREMENT ENDPOINTS

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TABLE 6-1
SUMMARY OF ASSESSMENT AND MEASUREMENT ENDPOINTS

Assessment Endpoint	Measurement Endpoint
Terrestrial	
Protection of habitat at HPA used by the avian species peregrine falcon, American kestrel, red-shouldered hawk, ánd loggerhead shrike	Plant bioaccumulation studies Earthworm and soil arthropod bioaccumulation and tissue residue studies Mourning and rock dove and small mammal tissue residue studies
Aquatic	
Protection of habitat at HPA used by the avian species California brown pelican, double-crested cormorant, Barrow's goldeneye, and great blue heron Protection of habitat for and HPA populations of California halibut and the, arrow, , and bay gobies	Mollusk and crustacean bioaccumulation and tissue residue studies Fish bioaccumulation and tissue residue studies
Protection of habitat for and HPA populations of mollusks, crustaceans and annelids at HPA	Mollusk, crustacean, and annelid toxicity, growth, and reproduction studies Mollusk and crustacean bioaccumulation and tissue residue studies

TABLE 6-2
ABUNDANT SUBTIDAL AND INTERTIDAL INVERTEBRATES

REPRESENTED SPECIES[a]	IMPORTANT LIFE HISTORY CHARACTERISTICS [b]
Annelids	
Oligochaetes *Tubificidae* spp.	· Intertidal or subtidal benthic animals live within and feed upon bottom deposits. · Larger species burrow freely in the substrate and probably feed indiscriminately on the sediment. · Very small species are meiobenthic (interstitial) worms which inhabit the interstices between substrate particles and feed on fine, organic debris. · Free-burrowing species tend to live in silts and poorly sorted, fine sands; meiobenthic worms are more restricted to coarser sands. · Particularly abundant in areas of organic enrichment.
Polychaetes *Cirrifornia spirabrancha* *Exogone lourei* *Glycinde polygnatha* *Nereis succina*	· Occur in all ocean environments. · Some are planktonic throughout life, but most species and adults are benthic, dwelling on or in the bottom at various depths · Some are carnivorous predators, some are herbivores, and others may be omnivorous, scavengers, filter feeders, or deposit feeders. · Are preyed upon by great variety of invertebrates and shorebirds.

REPRESENTED SPECIES[a]	IMPORTANT LIFE HISTORY CHARACTERISTICS [b]
Crustaceans	
Amphipods *Ampelisca abdita* *Caprella scaura* *Corphium spp.* *Grandierela japonica* *Rhincotropis spp.*	· Among the most abundant crustaceans in the intertidal zone of California. · The majority of described species are benthic. · Amphipods living on sandy beaches are active burrowers. Others living on more solid substrata in the intertidal zone build more permanent tubes of mud or debris. · Most appear to be scavengers or detritus feeders, but some consume tiny plants growing on rockweed and kelp, and a few capture and eat small animals such as copepods and bryozoans.
Decapods *Hemigrapsus oregonensis* *Hemileucon hinumensis*	· Common throughout San Francisco Bay, occurring in open mud flats, mats of green alga, and beds of eelgrass in high to low intertidal zones. · Preyed upon by shorebirds. · Are primarily herbivores, feeding on diatoms and green algae but can be carnivorous if meat is available. · Are skilled burrowers and bury themselves rapidly to escape predators.
Isopods *Cirolina harfordi* *Gnoriphaeroma spp.* *Sphaeroma pentodon*	· Predominantly benthic,though a few are planktonic. · Occur in intertidal zone where they hide in worm tubes, seaweeds, and sessile animals. · Form important intermediate links in the food chain as herbivores, prey, predators, parasites, scavengers, and detritus feeders. · Salinity, temperature, and humidity are physical factors that influence distribution.

REPRESENTED SPECIES[a]	IMPORTANT LIFE HISTORY CHARACTERISTICS [b]
Mollusks	
Bivalves *Cryptomya californica* *Gemma gemma* *Macoma balthica* *Musculista senhousia* *Mya arenaria* *Mylitus edulis* *Potanocorbula ameurensis* *Tapes japonica*	· Exist as free-living infaunal burrowers or nestlers or are epifaunal, attaching to the substrate by cementation or a byssus. · Predominantly filter feeders. · Occur in dense populations on rocky shores, pilings, or soft substrates. · Preyed upon by shorebirds as well as harvested by humans.
Gastropods *Odostomia fetella*	· Commonly found in bay mudflats. · Preyed upon by shorebirds.

Notes:

[a] Top 20 to 25 percent most abundant invertebrate taxa caught at HPA in the intertidal and subtidal surveys conducted by BioSystems Analysis, Inc., and the shrimp and crab surveys conducted by California Department of Fish and Game.

[b] References:

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Smith R. and J. Carlton. 1975. "Light's Manual: Intertidal Invertebrates of the Central California Coast." University of California Press. Berkeley, California. Third Edition.