

ATTACHMENT U-2

SCREENING VALUES AND TOXICITY REFERENCE VALUES

TABLE OF CONTENTS

1.0	INTRODUCTION	U2-1
2.0	GUIDELINES FOR SELECTING OR DEVELOPING TOXICITY VALUES	U2-3
3.0	UNCERTAINTY FACTORS	U2-6
4.0	SCREENING VALUES FOR ECOLOGICAL COMMUNITIES AND AMPHIBIANS	U2-8
4.1	Soil Screening Values	U2-9
4.1.1	Inorganic Compounds	U2-11
4.1.2	Volatile Organic Compounds	U2-11
4.1.3	Semi-Volatile Organic Compounds	U2-12
4.1.4	Organochlorine Pesticides	U2-12
4.1.5	Polychlorinated Biphenyls	U2-16
4.1.6	Organophosphate Herbicides/Pesticides	U2-16
4.1.7	Dioxins/Furans	U2-16
4.1.8	Polycyclic Aromatic Hydrocarbons	U2-17
4.2	Sediment Screening Values	U2-19
4.2.1	Inorganic Compounds	U2-21
4.2.2	Volatile Organic Compounds	U2-22
4.2.3	Semi-Volatile Organic Compounds	U2-22
4.2.4	Organochlorine Pesticides	U2-22
4.2.5	Polychlorinated Biphenyls	U2-22
4.2.6	Organophosphate Herbicides/Pesticides	U2-22
4.2.7	Dioxin/Furans	U2-22
4.2.8	Polycyclic Aromatic Hydrocarbons	U2-23
4.3	Surface Water Screening Values	U2-23
4.3.1	Inorganic Compounds	U2-25
4.3.2	Volatile Organic Compounds	U2-27
4.3.3	Semi-Volatile Organic Compounds	U2-28
4.3.4	Dioxin/Furan Toxic Equivalent	U2-29
4.3.5	Polycyclic Aromatic Hydrocarbons	U2-30
5.0	TOXICITY REFERENCE VALUES FOR WILDLIFE	U2-32
5.1	Inorganic Compounds	U2-35
5.2	Volatile Organic Compounds	U2-35
5.2.1	1,1,1-Trichloroethane	U2-36
5.2.2	1,1-Dichloroethane	U2-36
5.2.3	1,2-Dibromoethane	U2-37
5.2.4	1,2-Dichlorobenzene	U2-37
5.2.5	1,2-Dichloroethene	U2-37
5.2.6	1,2-Dichloropropane	U2-37
5.2.7	1,2,3-Trichlorobenzene	U2-38
5.2.8	1,3-Dichlorobenzene	U2-38
5.2.9	1,4-Dichlorobenzene	U2-38
5.2.10	Acetonitrile	U2-38
5.2.11	Acrolein	U2-39
5.2.12	Carbon Disulfide	U2-39
5.2.13	Chlorobenzene	U2-39
5.2.14	Diisopropyl Ether	U2-40
5.2.15	Diethyl Ether	U2-40

5.2.16	Ethylbenzene	U2-40
5.2.17	Ethylene Glycol	U2-41
5.2.18	Isopropanol	U2-41
5.2.19	Tert-Butyl Alcohol	U2-41
5.2.20	Tetrahydrofuran	U2-42
5.3	Semi-Volatile Organic Compounds	U2-42
5.3.1	Benzoic Acid	U2-42
5.3.2	Bis (2-chloroethyl) ether	U2-43
5.3.3	Diethylphthalate	U2-43
5.3.4	N-nitrosodimethylamine	U2-44
5.3.5	N-nitrosodiethylamine	U2-44
5.3.6	N-nitrosodipropylamine	U2-44
5.3.7	N-nitrosomethylethylamine	U2-44
5.4	Pesticides	U2-45
5.4.1	Aldrin	U2-45
5.4.2	Hexachlorobenzene	U2-46
5.4.3	Heptachlor epoxide	U2-46
5.4.4	Kepone	U2-46
5.4.5	Methoxychlor	U2-47
5.4.6	Mirex	U2-47
5.5	Polychlorinated Biphenyls	U2-47
5.5.1	Aroclor 1260	U2-48
5.6	Herbicides	U2-48
5.6.1	2,4-Dichlorophenoxyacetic Acid (2,4-D)	U2-48
5.6.2	2,4-Dichlorophenoxybutyric Acid (2,4-DB)	U2-48
5.6.3	2,4,5-Trichlorophenoxyacetic Acid (2,4,5-T)	U2-49
5.6.4	2,4,5-Trichlorophenoxypropanoic Acid (Silvex)	U2-49
5.6.5	2-sec-Butyl-4,6-dinitrophenol (Dinoseb)	U2-50
5.6.6	Dalapon	U2-50
5.6.7	Dichlorprop	U2-51
5.6.8	2-Methyl-4-Chlorophenoxyacetic Acid	U2-51
5.6.9	2-(2-Methyl-4-Chlorophenoxy)Propionic Acid	U2-52
5.7	Dioxin/Furan Toxicity Equivalent	U2-52
5.8	Polycyclic Aromatic Hydrocarbons	U2-52
6.0	REFERENCES	U2-54

TABLES

U.A2-1.	Soil Screening Values for Ecological Communities
U.A2-2.	Sediment Screening Values for Ecological Communities
U.A2-3.	Surface Water Screening Values for Ecological Communities
U.A2-4.	Toxicity Reference Values for Wildlife

1.0 INTRODUCTION

This attachment describes the development of toxicity values for the protection of ecological receptors at the former Casmalia Hazardous Waste Management Facility located in Casmalia, California (the Site). In an ecological risk assessment (ERA), the toxicity of a chemical of potential ecological concern (CPEC) to ecological receptors is assessed by identifying a toxicity value or screening value specific to the CPEC and receptor(s) being evaluated the U.S. Environmental Protection Agency (USEPA, 1999a).

For ecological communities, consisting of plants, soil invertebrates, sediment-dwelling invertebrates, reptiles, and aquatic life, and also amphibians, effects are assessed using screening values. Screening values are threshold concentrations expressed in milligrams per kilogram (mg/kg) or milligrams per liter (mg/L) that are effect levels or benchmarks for organisms inhabiting/exposed to that matrix (soil, sediment, surface water). Although more than one exposure route/pathway is considered potentially complete for ecological communities; generally, route-specific doses are not quantified for these groups of receptors. Exposures of ecological communities to site media are expressed as concentrations rather than doses, and generally encompass all potential exposure routes.

For mammals and birds, effects are assessed using toxicity reference values (TRVs). A TRV is defined as a daily dose of a chemical expressed in milligrams of chemical per kilogram of body weight per day (mg/kg bw-day) and may be represented as a dose associated with no-observed adverse effect level (NOAEL) or lowest-observed adverse effect level (LOAEL). Both NOAELs and LOAELs represent doses affecting receptors at the individual level. If risks (i.e., hazard quotients over 1) are predicted at this level (i.e., when the estimated exposure dose exceeds the LOAEL) effects may be evident at the population level. Because there is a higher level of concern, NOAEL-based TRVs are considered in making risk management decisions for protected (threatened and endangered) species. As mammal and bird TRVs are route-specific, they are used to evaluate effects from exposure via specific pathways (e.g., ingestion pathways).

Screening values and TRVs were used in the effects assessment and risk characterization phases of the ERA. The general equation used in risk characterization can be summarized as follows:

$$HQ = \frac{Exposure}{ToxicityValue}$$

Where:

HQ = Hazard quotient (unitless)

Exposure = Exposure concentration in mg/kg or mg/L or exposure dose in mg/kg bw-day

Toxicity Value = Screening value (mg/kg or mg/L) or toxicity reference value (mg/kg bw-day).

If chemical exposure exceeds the screening value or TRV (i.e., HQ is greater than 1), there is a potential for unacceptable risk. However, toxicity values are conservative literature-derived toxicity values and are biased toward protection of the individual. Therefore, exceedances of TRVs do not necessarily indicate adverse effects to populations of receptor species at a site. Additionally, because of the many conservative assumptions used in an ERA, exceedances of toxicity values are not necessarily predictive of effects to individuals. Risk managers are advised to consider other factors in evaluating ecological risks including spatial distribution of contaminants, the magnitude of the estimated hazards, and other lines of evidence from the ERA.

Screening values and TRVs were presented in the Remedial Action/Feasibility Study (RI/FS) Work Plan for Casmalia (CSC, 2004). However, some values have been updated since the completion of the Work Plan and additional toxicity values were developed for onsite CPECs for this ERA based on the guidelines described below.

2.0 GUIDELINES FOR SELECTING OR DEVELOPING TOXICITY VALUES

The hierarchy for selecting toxicity values is based on the following (USEPA, 1999a):

- Toxicity values developed and/or adopted by federal and/or state regulatory agencies generally provided in the form of standards, criteria, guidance, or benchmarks;
- Toxicity values published in scientific literature;
- Toxicity values calculated for sediment using equilibrium partitioning approach; and
- Toxicity values from surrogate compounds.

For this ERA, toxicity values commonly used in ERAs and those reported by the USEPA (e.g., USEPA Ecological Soil Screening Levels [EcoSSLs], (2007c) and the California Environmental Protection Agency (CalEPA) (e.g., Biological Technical Assistance Group [BTAG] TRVs (CalEPA, 2002a) were utilized whenever available. Additional published TRVs were prioritized for use as described below. For CPECs where no toxicity values were available, suitable empirical data as published in literature were used to develop screening values and TRVs. Screening values derived from empirical data were preferred over those derived from modeling approaches, such as equilibrium partitioning, or statistical extrapolation.

General guidance for the development of toxicity values is available from literature sources (USEPA, 1997; 1998; 1999a,b; 2000b; CalEPA, 1996). Cal/EPA and/or USEPA guidance recommends selecting appropriate data to develop toxicity values, which include the following:

- The test or indicator species used in the study should be representative of the receptor species at the site in terms of body size, feeding habits, and exposure routes;
- Studies that indicate chronic exposures are preferred. For wildlife toxicity tests, USEPA (1999a) defines chronic as greater than 90 days, subchronic as 14 to 90 days, and acute as less than 14 days. Shorter tests that are conducted during critical life stages (e.g., on young, during gestation) can be considered chronic;
- The route of exposure should be representative of expected site-related exposure pathways. For example, ingestion as the route of exposure is preferred over gavage, and exposure routes like egg injection are not recommended. For plants and invertebrates, natural soil is preferred over hydroponic or filter paper matrices;
- Adverse effects include acute, chronic, lethal, and sublethal. The BTAG TRVs were developed considering “biological effects that primarily related to growth, reproduction, and development; however, all effects deemed ecologically relevant were considered when developing TRVs.” USEPA (1999a) indicates that “Superfund risk assessments should use site-specific assessment endpoints that address chemical specific potential adverse effects to local populations and communities of plants and animals.” Thus, the preferred endpoints for protection of both individuals and populations are reproduction, growth and development, and survival. Generally, changes in organ weight and some histological, enzymatic, or hematological endpoints are considered less useful for development of TRVs. However, these endpoints may be used in TRV development if the measure is known to adversely affect the health or fitness of the organism as discussed in USEPA (2007c; Guidance for Developing Eco-SSLs); and

- Studies that show ecologically adverse effects at the lowest statistically significant concentration (i.e., LOAEL or lowest-observed-adverse-effects concentration [LOAEC]) or the highest NOAEL or no-observed-adverse-effects concentration (NOAEC) below the lowest LOAEL are preferred. Selection of these studies maintains the conservative nature of the risk assessment when extrapolating from laboratory studies to site receptors and site-specific conditions.

A toxicity value should represent the concentration or dose of a chemical that causes no observed adverse effects to an ecologically relevant endpoint of a receptor over a long-term, (chronic) exposure duration. Toxicity values can be derived from various published toxicity studies and are generally based on NOAEL and/or LOAEL values from such studies. The NOAEL is defined as the highest exposure level (i.e., dose) shown to produce no statistically significant adverse effect (e.g., reduced growth, impaired reproduction, increased mortality) in a potential receptor species compared with the study controls (USEPA, 1997). The LOAEL is defined as the lowest exposure level or mid-range effects shown to produce adverse effects. For the ERA, potential risks to wildlife were assessed at two effects levels: a NOAEL-based TRV, hereafter referred to as the low TRV, and a LOAEL-based TRV, or high TRV. NOAELs are based on no effects and LOAELs represent concentrations where effects are observed. In the case of BTAG TRVs, the low value is a NOAEL and the high value was selected to represent “mid-range adverse effects levels.” In the case of sources, such as USEPA EcoSSL Guidance (2007c), where only a NOAEL-based TRV is provided, paired LOAEL-based TRVs were selected according to the following criteria:

- If the recommended NOAEL-based TRV was bounded, the LOAEL from the same study and endpoint was selected;
- If the recommended NOAEL-based TRV was unbounded, the lowest reproduction, growth, and survival LOAEL greater than the NOAEL-based TRV was selected;
- If the recommended NOAEL-based TRV was derived from a LOAEL to which uncertainty factors were applied, the LOAEL without those uncertainty factors was used as the LOAEL-based TRV; and
- If the recommended NOAEL-based TRV was based on geometric means of endpoints, in agreement with USEPA, the LOAEL based TRVs were derived using the following step-wise approach:
 - a. Calculating geometric means of bounded LOAELs for growth and reproduction endpoints only;
 - b. Identifying the lowest bounded LOAEL for survival endpoints; and
 - c. Selecting the lowest value from steps a and b above as the proposed LOAEL TRV.

USEPA (1999a) and Sample et al. (1996) recommend the use of studies reporting the following endpoints, in order of preference:

- Chronic NOAEL;
- Subchronic NOAEL;
- Chronic LOAEL;
- Subchronic LOAEL;
- Acute median lethality point estimate (e.g. LD50); and
- Single dose toxicity value.

Due to limited available information, the selection criteria listed above may not be met entirely. Therefore, a conservative approach is used when deriving TRVs for receptor species in order to account for uncertainties associated with extrapolating across species, test conditions, and various behavioral and ecological parameters, as described below. Several studies and literature papers were reviewed for this ERA; however, only the studies considered appropriate in selecting or developing toxicity values based on the guidelines described above, are described in the following sections.

3.0 UNCERTAINTY FACTORS

When adequate data for site receptors are unavailable, it is sometimes necessary to apply one or more uncertainty factors (UFs) to the toxicity values from literature in order to determine conservative site-specific and ecologically relevant TRVs. The number and magnitude of the uncertainty factors are generally based on the number, quality, duration, and sensitivity of the studies used to derive the TRV and on the taxonomic diversity of the surrogate species tested. The final TRV should be a chronic NOAEL and/or LOAEL specific for the receptor(s) evaluated at a given site. For each CPEC, the applied UFs are clearly described, but in general UFs greater than 1 would be considered to compensate for each of the following:

- Exposure duration (i.e., from acute or subchronic to chronic);
- Toxicological endpoint does not address sensitive indicators (i.e. reproduction, behavior);
- Toxicological endpoint (i.e., from LOAEL or LD50 to NOAEL);
- Inter-taxonomic variability (i.e., from test to indicator species); and
- Other modifying factors, as necessary.

The use of UFs for human health risk assessment is well known. Comparable protocols for the use of UFs in ERAs have been recommended (Calabrese and Baldwin, 1993). After reviewing several literature sources (Dourson and Stara, 1983, as cited in Calabrese and Baldwin, 1993; Sloof et al., 1986; Sample et al., 1996; CalEPA, 1996; USEPA, 1997; U.S. Army Center for Health Promotion and Preventive Medicine [USACHPPM], 2000), a conservative set of UFs were selected to account for differences in study duration and endpoint. The objective of the extrapolations is to normalize the study doses to chronic NOAELs, although LOAELs may be preferable or useful for some ERAs.

The UFs selected for this ERA are as follows:

- LOAEL/C to NOAEL/C = 10
- Subchronic LOAEL/C to chronic NOAEL/C = 10
- Acute Lethal Value (LC50) to chronic NOAEL/C = 100 (personal communication with Michael Anderson of DTSC and in USACHPPM, 2000).

For toxicity studies that report LOAEL/C values only, NOAEL/Cs were extrapolated using a UF of 0.1

The UFs may be modified, as appropriate, on a case-by-case basis. For protection of the ecological community on a population-level basis, sensitive endpoints (EC50s) determined over a chronic exposure duration may be considered upper-bound criteria for more serious effects. Additional extrapolations may also be needed to account for differences between the site-specific receptor and laboratory animals used in the study selected to develop the TRV. For inter-taxonomic extrapolations and other modifying factors (such as safety factors to account for special status species), UFs may be applied with a maximum combined UF of 500.

An alternate approach is to use scaling factors or allometric models to account for uncertainties associated with intraspecies or interspecies differences (Sample et al., 1996; Sample and

Arenal, 1999; Engineering Field Activity [EFA], 1997, 1998) based on differences in body weights. CalEPA does not recommend allometric adjustment of TRVs unless the body weights of the test species and receptor species differ by two orders of magnitude (CalEPA, 1999)¹.

¹ Based on studies reviewed for this attachment, body weight differences between test species and ecological receptors selected for the ERA were not greater than two orders of magnitude, and therefore, TRVs were not adjusted for body weight differences.

4.0 SCREENING VALUES FOR ECOLOGICAL COMMUNITIES AND AMPHIBIANS

Screening values were selected or developed based on the guidelines listed above (Section 2.0) for:

- Terrestrial plants – protective of plants from exposure to soil;
- Terrestrial invertebrates – protective of soil fauna from exposure to soil;
- Sediment-dwelling invertebrates – protective of sediment-dwelling organisms from exposure to sediment. ;
- Aquatic life – protective of aquatic invertebrates and fish from exposure to surface water;
- Aquatic plants – protective of aquatic plants from exposure to surface water; and
- Amphibians – protective of amphibians from exposure to surface water.

Although some toxicity studies are available (Pauli et al., 2000; James et al., 2004), in general, toxicity data as well as exposure parameters required to estimate dose are limited for amphibians exposed to terrestrial areas and reptiles. These receptors were evaluated qualitatively as described in the main ERA text (Appendix U).

In order to meet the objectives of the ERA, screening values protective of ecological communities and amphibians (in surface water) were selected or developed from the literature sources listed below. Where available, screening values for CPECs were selected based on sources recommended in ERA guidance documents (USEPA, 1999a; CalEPA, 1996); sources were referenced for these screening values and study details are not provided in this attachment. Screening values for CPECs that were not readily available and were based on literature review are described in the following sections.

Screening values recommended in the sources listed below that were derived in whole or in part to protect human health, such as for drinking water, may be considered inappropriate and overprotective for use in an ERA. In cases where generic toxicity values or values representing background concentrations or method detection limits were recommended by regulatory guidance for classes of compounds (e.g., chlorinated organics), alternate screening values from empirical data for the CPEC may supersede regulatory guidance. In these cases, the selections of final screening values are described in detail in the following sections.

To select screening values or studies to develop screening values for ecological communities and amphibians, a hierarchy of the sources was established for this ERA. The objective of the hierarchy is to ensure that appropriate, conservative, and where available, published/promulgated values are preferentially selected instead of selecting the lowest available screening value, which could be based on data with very low confidence. The hierarchy for selecting screening values for the CPECs was based on USEPA guidance (USEPA, 1999b), which gives highest priority to toxicity values developed and/or adopted by federal and/or state regulatory agencies followed by toxicity values published in the scientific literature. The hierarchy of sources used in the selection of screening values for each media is discussed in detail below.

4.1 Soil Screening Values

Soil screening values for plants and soil invertebrates selected for the ERA are presented in Table U.A2-1. As discussed with USEPA prior to the submittal of the Draft Remedial Investigation (RI) Report (CSC, 2008), preference would be given to the EcoSSLs developed by USEPA (2007c). Screening values from guidance generally used in ERAs (e.g. Oak Ridge National Laboratory's [ORNL]) and deemed appropriate would be next in the hierarchy, followed by data from other documents/databases listed below, several of which have lower overall data quality and confidence and some of which have limited peer review as far as their uses in quantitative ERAs.

The EcoSSL derivation process is considered sound and current as it was developed by a multi-stakeholder workgroup consisting of federal, state, consulting, industry, and academic participants led by the USEPA's Office of Superfund Remediation and Technology Innovation (OSRTI) and included extensive and relatively recent literature searches. The Eco-SSLs are defined as "concentrations of contaminants in soil that are protective of ecological receptors that commonly come into contact with soil or ingest biota that live in or on soil." One of the objectives of the EcoSSLs was to conserve resources by limiting the need for USEPA and other risk assessors to perform the repeated exercise of literature searches for toxicity data to develop toxicity values for the same contaminants at every site. By developing standards, it would also allow risk assessors to focus their resources on other key site-specific issues needed for critical decision-making. Additionally, USEPA also expects that the EcoSSLs will increase consistency among screening risk analyses and decrease the possibility that potential risks from soil contamination to ecological receptors will be overlooked. The general approach for deriving the EcoSSLs for plants and soil invertebrates included four steps: (1) conduct literature searches; (2) screen identified literature with exclusion and acceptability criteria; (3) extract, evaluate, and score test results for applicability in deriving an EcoSSL; and (4) derive the value. This process is described in detail in the EcoSSL guidance (USEPA, 2007c). The EcoSSLs are endorsed by USEPA and therefore, have the highest preference in the hierarchy of TRVs.

The ORNL guidance documents provide ecotoxicological screening benchmarks for surface water, sediment, and surface soil applicable to a range of aquatic organisms, soil invertebrates, and terrestrial plants. Toxicity data for the screening values reported in the ORNL documents were obtained from searches of bibliographic data bases (BIOSIS, POL TOX I, current contents), review articles, and conventional literature searches. The general criteria used for including a study in a dataset included: (1) methodology was clearly stated (especially concentrations of applied chemicals) and followed in the experiment; (2) results were quantified as measures of survivorship, growth, respiration, reproduction, substrate transformation, or enzyme activity; (3) results were presented in numeric form, or graphical presentations of data were clearly interpretable; and (4) an unambiguous reduction existed in the measured parameter within the range of applied concentrations of the chemical of interest.

The approach used by ORNL in deriving screening values is similar to the approach used by the National Oceanographic and Atmospheric Administration's (NOAA) method for deriving the Effects Range Low (ER-L) (Long and Morgan, 1990) benchmarks. The ER-L is the tenth percentile of the distribution of various toxic effects thresholds for various organisms in sediments. ORNL assumes that the "toxicity of a chemical in soil is a random variate, the toxicity of contaminated soil at a particular site is drawn from the same distribution, and the assessor should be 90% certain of protecting organisms growing in the site soil." The toxicity

benchmarks were derived by ranking the LOAEC values and then selecting the approximate 10th percentile value. Similar to ER-Ls, statistical fitting was not used because sufficient data were not always available and also because these benchmarks are to be used as screening values and do not require the consistency and precision of regulatory criteria. If there were 10 or less toxicity values for a chemical, the lowest LOAEC was selected as a benchmark. If there were more than 10 toxicity values, the 10th percentile LOAEC value was used as the benchmark. If the 10th percentile fell between LOAEC values, a value was selected by interpolation. These process is described in detail in the ORNL guidance documents (Efroymson et al., 1997a,b).

Screening values reported in other guidance or documents are generally a compilation of values from other sources based on similar approaches and hierarchies and are not discussed in detail herein.

Based on the discussion above, soil screening values for soil invertebrates and plants were selected or developed from the following sources listed in order of preference following the guidelines described in Section 2:

- USEPA EcoSSLs Guidance (USEPA, 2007c);
- ORNL: Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Terrestrial Plants (Efroymson et al., 1997a) and Toxicological Benchmarks for Contaminants of Potential Concern for Effects on Soil and Litter Invertebrates and Heterotrophic Processes (Efroymson et al., 1997b);
- Lowest of CPEC-specific, non-background values for:
 - USEPA Region 6 Screening Level Ecological Risk Assessment Protocol, Appendix U (USEPA, 1999b);
 - USEPA Supplemental Guidance to Risk Assessment Guidance for Superfund (RAGS): Region 4 Bulletins, Ecological Risk Assessment (USEPA, 2001); and
 - US Fish and Wildlife Service Evaluating Soil Contamination (Beyer, 1990);
- Canadian Council of Ministers of the Environment (CCME, 2006a); Soil Quality Guidelines;
- Dutch “Maximum Permissible Concentrations” (MPCs) or derived values from toxicity data presented in these documents (Crommentuijn et al., 2000a, 1997, Van de Plassche et al., 1994);
- Empirical data from the ECOTOX Database (USEPA, 2007a); and
- Toxicity values from surrogate compounds.

Priority was given to screening values derived from empirical data; values derived through modeling or statistical extrapolation were given lower priority than listed in the above hierarchy.

The U.S. Army Center for Health Promotion and Preventive Medicine (USACHPPM) Terrestrial Toxicity Database (USACHPPM, 2007), Netherlands Ministry of Housing, Planning, and Environment (MHSPE) Target and Intervention Values (MHSPE, 1994), and other Dutch MPCs (Crommentuijn et al., 2000b) were also consulted, however no screening values were selected from these sources.

USACHPPM TRVs (USACHPPM, 2006) were also consulted for amphibian screening values; however, no screening values were selected for use because none were available for the Site CPECs.

4.1.1 Inorganic Compounds

Plant and soil invertebrate screening values for all the inorganic CPECs (i.e., metals) were obtained from the sources listed above as presented in Table U.A2-1. A plant screening value was not available for total cyanide. Although, amenable cyanide was selected as a CPEC, there are no plant or soil invertebrate screening values available for this chemical. Potential risks to ecological communities from exposure to amenable cyanide are assumed to be included in the risks from exposure to total cyanide.

4.1.2 Volatile Organic Compounds

Soil screening values developed for volatile organic compounds (VOCs) are described below.

4.1.2.1 Plant Screening Values

Plant screening values or toxicity studies that could be used in developing screening values were not available for most of the VOCs except toluene, 1,1-dichloroethylene, isopropanol, and tetrachloroethylene. The plant screening value for toluene was obtained from Efrogmson et al. (1997a).

For 1,1-dichloroethylene, a single study reporting empirical data was available in the ECOTOX database (USEPA, 2007a). Pestemer and Auspurg (1989) report 14-day effects concentration (EC50s) of greater than 1000 mg/kg for growth reduction in 15 common crops. Conservatively, assuming the lowest EC50 of 1000 mg/kg equivalent to a LOAEC for more serious adverse effects, a UF of 10 was applied to extrapolate to a NOAEC resulting in a value of 100 mg/kg, which was used as the plant screening value for 1,1-dichloroethylene.

For isopropanol, empirical data were available in the ECOTOX database (USEPA, 2007a). The study reporting ecologically relevant adverse effects at the lowest concentration was used to develop a plant screening value. Hulzebos et al. (1989) report a 14-day EC50 of greater than 1000 mg/kg for lettuce germination when exposed to soil amended with isopropanol. Because plants that do not germinate do not survive, the endpoint was considered lethal. Conservatively assuming an EC50 of 1000 mg/kg as an acute concentration, a UF of 100 was applied to extrapolate to a NOAEC resulting in a value of 10 mg/kg, which was used as the plant screening value for isopropanol.

For tetrachloroethylene, empirical toxicity data were available in the ECOTOX database (USEPA, 2007a). The study reporting ecologically relevant adverse effects at the lowest concentration was used to develop a plant screening value. Hulzebos et al. (1993) report a 14-day EC50 of greater than 1000 mg/kg for reduced lettuce biomass when exposed soil amended with tetrachloroethylene. Conservatively assuming an EC50 of 1000 mg/kg equivalent to a LOAEC for more serious adverse effects, a UF of 10 was applied to extrapolate to a NOAEC resulting in a value of 100 mg/kg, which was used as the plant screening value for tetrachloroethylene.

Toxicity studies for effects of 1,1,1-trichloroethane, 1,1-dichloroethane, 1,2-dichloroethene, acetone, acetonitrile, acrolein, benzene, carbon disulfide, diisopropyl ether, ethyl benzene, Freon 113, methylcyclopentane, methyl ethyl ketone, methyl isobutyl ketone, methylene

chloride, propanal, tert-butyl alcohol, tetrahydrofuran, and trichloroethylene on plants were not available in literature, and therefore, plant screening values could not be developed for these CPECs.

4.1.2.2 Soil Invertebrate Screening Values

Soil invertebrate screening values for VOCs were obtained from the sources listed above as presented in Table U.A2-1. Soil invertebrate screening values or toxicity studies for effects of 4-methyl-2-pentanone (methyl isobutyl ketone), acetone, acetonitrile, acrolein, carbon disulfide, diisopropyl ether, Freon 113, isopropanol, methylcyclopentone, methyl ethyl ketone (MEK), propanal, and TBA on soil invertebrates were not available, and therefore, soil invertebrate screening values could not be developed for these CPECs.

4.1.3 **Semi-Volatile Organic Compounds**

Plant and soil invertebrate screening values for the semi-volatile organic compounds (SVOCs), bis(2-ethylhexyl)phthalate, diethylphthalate, and di-n-butylphthalate were obtained from the sources listed above as presented in Table U.A2-1. Only soil invertebrate screening values were available for n-nitrosodimethylamine, n-nitrosodipropylamine, and n-nitrosomethylethylamine from these sources as well. Plant and soil invertebrate screening values and empirical toxicity data were not available for benzoic acid and n-nitrosopyrrolidine.

4.1.4 **Organochlorine Pesticides**

Soil screening values for organochlorine pesticides are described below.

4.1.4.1 Plant Screening Values

Plant screening values were not readily available for most of the pesticides from the sources listed above except for heptachlor (USEPA, 1999a). Plant screening values were developed for the rest of the pesticides based on toxicity studies available in literature as described below. Toxicity studies for adverse effects of endrin, kepone, methoxychlor, and mirex on plants were not available in literature, and therefore, plant screening values could not be developed for these CPECs.

For aldrin, USEPA Region 5 guidance (USEPA, 2003) recommends a soil screening value of 0.0033 mg/kg based on unreported effects on plants. No additional empirical studies were available in the ECOTOX (USEPA, 2007a) database for aldrin; however empirical data were available for dieldrin. Therefore, due to close similarities between aldrin and dieldrin and lack of any specific toxicological data for effects of aldrin on plants, dieldrin was used as a surrogate. The screening value of 1.0 mg/kg for dieldrin is based on a 21 day growth LOAEC for four plants species as described below (Rajanna and De la Cruz, 1977). This value was also selected for aldrin for the ERA.

For DDT, empirical data were available in the ECOTOX database (USEPA, 2007a). A study by Urzua et al. (1986) was selected for developing a plant screening value according to the guidelines discussed above. In this study, Dutch clover was grown for 10 to 14 weeks in soil amended with DDT. Statistically significant reductions in plant biomass and mycorrhizal colonization were first observed at 9 mg/kg. Although effects on root mycorrhizal colonization

were observed at soil concentrations of 1 mg/kg, these effects were beneficial to the clover. Therefore, the concentration of 9 mg/kg was assumed as a LOAEC and a UF of 10 was applied to extrapolate to a chronic NOAEC, resulting in a value of 0.9 mg/kg, which was used as the plant screening value for total DDT and all its metabolites: 4,4'-DDD and 4,4'-DDE and for total DDT.

For benzene hexachloride (BCH), a screening value of 0.004 mg/kg is recommended by USEPA Region 5 guidance (USEPA, 2003), based on unreported effects on plants. Empirical data are available from the ECOTOX database (USEPA, 2007a). A study by Hulzebos et al. (1989), reports a 14-d EC50 of greater than 1000 mg/kg for lettuce germination. Failure to germinate was considered a lethal and ecologically significant endpoint and thus, preferred over the USEPA Region 5 guidance value (USEPA, 2003). Conservatively assuming the EC50 of 1000 mg/kg from Hulzebos et al. (1989) as an acute effect, following the guidelines, an UF of 100 was applied to extrapolate to a NOAEC resulting in a value of 10 mg/kg, which was used as the plant screening value for BCH.

For chlordane, a screening value of 0.224 mg/kg is recommended by USEPA Region 5 guidance (USEPA, 2003) based on unreported toxicological effects. Additional empirical data for chlordane suggesting much higher screening values were also available. However, the most conservative value of 0.224 mg/kg was selected as the plant screening value for chlordane.

For dieldrin, no published screening values were available for plants. However, toxicity data were available in the ECOTOX database (USEPA, 2007a). A study by Rajanna and De la Cruz (1977) reported ecologically adverse effects at the lowest concentration and was used to develop screening values for plants. In this study, cotton, soybean, bread wheat, and corn seeds were grown in soil amended with dieldrin for 21 days. The seeds were heat-treated at 40°C for 5 days, 2 days, or not treated. Heat-treated seeds showed reductions in biomass and plant height when grown in the amended soil, while non-treated seeds showed no effect. The lowest effect concentration reported was 10 mg/kg for bread wheat, which was considered a LOAEC for dieldrin. A UF of 10 was applied to extrapolate to a chronic NOAEC of 1 mg/kg which was used as the plant screening value for dieldrin.

For endosulfan (I) or endosulfan sulfate (II), empirical toxicity data for technical endosulfan, a mixture of the I and II isomers, were available in the ECOTOX database (USEPA, 2007a). The study reporting ecologically relevant adverse effects at the lowest concentration was used to develop screening values for endosulfan and endosulfan sulfate, a breakdown product of endosulfan with no available compound-specific data. Hulzebos et al. (1989) report a 14-day EC50 of greater than 1000 mg/kg for lettuce germination when exposed to soil amended with endosulfan. Because plants that do not germinate do not survive, the endpoint was considered lethal. Conservatively assuming an EC50 of 1000 mg/kg as an acute concentration, an UF of 100 was applied to extrapolate to a NOAEC resulting in a value of 10 mg/kg, which was used as the plant screening value for endosulfan I, endosulfan II, and endosulfan sulfate.

For hexachlorobenzene, empirical toxicity data were available in the ECOTOX database (USEPA, 2007a). The study reporting ecologically relevant adverse effects at the lowest concentration was used to develop a screening value for hexachlorobenzene. Hulzebos et al. (1993) report a 14-day EC50 of greater than 1000 mg/kg for reduced lettuce biomass when exposed soil amended with hexachlorobenzene. Conservatively assuming an EC50 of 1000 mg/kg equivalent to a LOAEC for more serious adverse effects, a UF of 10 was applied to

extrapolate to a NOAEC resulting in a value of 100 mg/kg, which was used as the plant screening value for hexachlorobenzene.

4.1.4.2 Soil Invertebrate Screening Values

Soil invertebrate screening values were not readily available for most of the pesticides from the sources listed above except for hexachlorobenzene (CCME, 2006b). Values from Crommentuijn et al. (2000a) were used for aldrin, dieldrin, and endosulfan isomers (I and II). Soil invertebrate screening values were developed for the rest of the pesticides based on toxicity studies available in literature as described below.

For aldrin and dieldrin, the Dutch MPC of 0.05 mg/kg is based on the lowest dieldrin NOAEC data of 0.5 mg/kg for *Onychiurus armatus* (collembola species) (Crommentuijn, 2000a; Van de Plassche, 1994). Due to lack of other published screening values, the study used for the Dutch MPC was selected as the most appropriate value for dieldrin. Following the guidelines, no UFs are applied to NOAECs in the ERA, therefore a screening value of 0.5 mg/kg was used for soil invertebrates for dieldrin. This value was used as a surrogate for aldrin.

For DDT, USEPA Region 4 (USEPA, 2001) recommends a soil screening value of 0.0025 mg/kg. This value is based on the MHSPE Target value, or typical ambient concentration (MHSPE, 1994). As discussed above, background values are not based on toxicological effects to site receptors, and therefore, not preferred as screening values for this ERA despite being recommended by regulatory guidance. The Canadian soil quality guideline (CCME, 2006a) for total DDT (DDT and metabolites) based on residential soil is 0.7 mg/kg. The Dutch MPC value of 0.01 mg/kg for DDT is based on a median LC50 or EC50 of 10 mg/kg for collembola (Crommentuijn, 2000a; Van de Plassche et al., 1994). This value was assumed to be an LC50 value, since invertebrate toxicity tests typically measure mortality. Empirical data from ECOTOX database (USEPA, 2007a) were also available for DDT. In a study by Harris (1966), 1st instar cricket larvae were exposed to soils amended with DDT for 18 hours and the mean LC50 of all soils reported was 39.4 mg/kg. Based on the guidelines and the priority of sources for developing a screening value for this ERA, the study used for the Dutch MPC was selected as the most appropriate for DDT. Following the guidelines, an UF of 100 was applied to extrapolate from an acute lethal concentration of 10 mg/kg to a chronic NOAEC, resulting in a value of 0.1 mg/kg, which was used as the soil invertebrate screening value for DDT and its metabolites.

For benzene hexachloride (BHC), also known as lindane or gamma-hexachlorocyclohexane (gamma-HCH), a Dutch MPC value of 0.005 mg/kg for gamma-BHC is available. This value is based on a NOAEC of 0.05 mg/kg for collembola (Crommentuijn, 2000a; Van de Plassche et al., 1994). Empirical data were also available from the ECOTOX (USEPA, 2007a) database. A study by Smith (1948) reports a 96 hr LC50 of approximately 200 mg/kg for the common woodlouse exposed to soil amended with BHC. Although not included in the ECOTOX database, studies by Frampton et al. (2006) and Jansch et al. (2006) were also considered. Frampton et al. (2006) reports a gamma-HCH geometric mean of LC50s of 59 mg/kg for the earthworm *Eisenia fetida*. However, this species was not considered especially sensitive to pesticides. The lowest reported LC50 for gamma-HCH appears to be approximately 0.5 mg/kg for *Onychiurus armatus*, a collembola species (value estimated from the figure). Jansch et al. (2006) cite a study by Lock et al. (2002), which reports a NOAEC of 0.03 mg/kg for chronic reproductive effects in *Folsomia candida* exposed to gamma-BHC in soil. The study by Lock et

al. (2002) was a study reporting the most conservative soil screening value and it was selected for the ERA. Following the guidelines, no UFs are applied to NOAECs. Gamma-BHC was considered a conservative representative for all BHC isomers, and therefore a value of 0.03 mg/kg was selected as the soil screening value for BHC.

For chlordane, USEPA Region 4 (2001) recommends a soil screening value of 0.1 mg/kg; however, this value is based on the "A" value or background concentration for organochlorine pesticides from Beyer (1990). As discussed above, screening values based on ambient concentrations were not selected for this ERA. The Dutch MPC value of 0.0043 mg/kg for chlordane is based on a LC50 or EC50 of 4.3 mg/kg for insects (Crommentuijn, 2000; Van de Plassche et al., 1994). This value was assumed to be a LC50 value, since invertebrate toxicity tests typically measure mortality. Empirical data were also available from the ECOTOX database (USEPA, 2007a). Goats and Edwards (1988) report a 14-day earthworm LC50 of 23.9 mg/kg. Based on the guidelines and the priority of sources for developing a screening value for this ERA, the study used for the Dutch MPC was selected as the most appropriate for chlordane. Following guidelines, an UF of 100 was applied to extrapolate from an acute lethal concentration to a chronic NOAEC, resulting in a value of 0.043 mg/kg, which was used as the soil invertebrate screening value for chlordane.

For endrin, USEPA Region 4 (2001) recommends a soil screening value of 0.001 mg/kg. This value is derived from MHSPE target value, or typical ambient concentration (MHSPE, 1994). As discussed above, screening values based on ambient concentrations were not selected for this ERA. The Dutch MPC value of 0.00095 mg/kg for endrin is based on a LC50 of 0.95 mg/kg for collembola (Crommentuijn, 2000a; Van de Plassche et al., 1994). This value was assumed to be an LC50 value, since invertebrate toxicity tests typically measure mortality. Empirical data is available from the ECOTOX database (USEPA, 2007a). A study by Cathey (1982) reported a 6-week earthworm (*Lumbriculus terrestris*) LC50 of 66 mg/kg. Based on the guidelines and the priority of sources for developing a screening value for this ERA, the study used for the Dutch MPC was selected as the most appropriate for endrin. Following guidelines, an UF of 100 was applied to extrapolate from an acute lethal concentration to a chronic NOAEC, resulting in a value of 0.0095 mg/kg, which was used as the soil invertebrate screening value for endrin.

The Dutch MPC value of 0.05 mg/kg for endosulfan is based on the LC50 or EC50 of 5 mg/kg for oligochaetes (Crommentuijn, 2000a; Van de Plassche et al., 1994). This value was assumed to be a LC50 value, since invertebrate toxicity tests typically measure mortality. The application of a UF of 100 by Crommentuijn et al. is consistent with methods for this ERA. Therefore, a screening value of 0.05 mg/kg was used for endosulfan (I) and endosulfan (II). This value was also used for endosulfan sulfate, using endosulfan as a surrogate.

The Dutch MPC value of 0.0007 mg/kg for heptachlor and heptachlor epoxide is based on the heptachlor median LC50 or EC50 of 0.7 mg/kg for insects (Crommentuijn, 2000a; Van de Plassche et al., 1994). This value was assumed to be a LC50 value, since invertebrate toxicity tests typically measure mortality. Empirical data is available from the ECOTOX database (USEPA, 2007a). In a study by Harris (1966), 1st instar cricket larvae were exposed to soils amended with heptachlor for 18 hours. The mean LC50 of all soils reported was 5.58 mg/kg. Based on the guidelines and the priority of sources for developing a screening value for this ERA, the study used for the Dutch MPC was selected as the most appropriate for heptachlor. Following guidelines, an UF of 100 was applied to extrapolate from an acute lethal concentration to a chronic NOAEC, resulting in a value of 0.007 mg/kg, which was used as the

soil invertebrate screening value for heptachlor. This value was used as a surrogate for heptachlor epoxide.

For hexachlorobenzene, USEPA Region 4 (2001) recommends a soil screening value of 0.0025 mg/kg. This value is based on a MHSPE target value, or typical ambient concentration (MHSPE, 1994). As discussed above, screening values based on ambient concentrations were not selected for this ERA. The interim remediation criteria for the protection of environmental and human health reported in CCME (2006a) is 2.0 mg/kg for residential soil. As no additional empirical data were found, the value of 2.0 mg/kg was selected as the soil invertebrate screening value for hexachlorobenzene.

Toxicity studies for effects of kepone, methoxychlor, and mirex on soil invertebrates were not available in literature, and therefore, soil screening values could not be developed for these CPECs.

4.1.5 Polychlorinated Biphenyls

Plant and soil invertebrate screening values for polychlorinated biphenyls (PCBs): Aroclor 1260 and total PCBs were obtained from the sources listed above as presented in Table U.A2-1. Plant and soil invertebrate screening soil screening values for PCB TEQ were not required.

4.1.6 Organophosphate Herbicides/Pesticides

Plant and soil invertebrate screening values were not available for organophosphate herbicides and pesticides.

MPCs were available for dichlorprop, (4-chloro-2-methylphenoxy)acetic acid (MCPA), and 2(4-chloro-2-methylphenoxy)propionic acid (MCP) in Crommentuijn et al. (2000a). However, these values were not considered appropriate as soil screening values for this ERA. MPC for dichlorprop was derived from sediment MPCs based on EqP and toxicity studies with a crustacean. The MPCs for MCPA and MCP were based on toxicity data for microbial communities and fish, respectively.

Toxicity studies for effects of organophosphate herbicides or pesticides on plants and soil invertebrates were also not available in literature. Plant data was available in the ECOTOX (USEPA, 2007a) database; however, only spray or dipped/soaked routes of exposure were available. These routes of exposure are not considered appropriate for the ERA. Publications by Frampton et al. (2006) and Jansch et al. (2006) were reviewed for soil pesticide screening values, however no Site-related CPECs were included in these studies. Therefore, plant and soil invertebrate screening values could not be developed for these CPECs.

4.1.7 Dioxins/Furans

Although dioxin/furan congeners were selected as CPECs, individual congeners were not evaluated in the ERA. Soil screening values for dioxins/furans were not available in literature and were not developed for this ERA. The mechanism of toxicity of dioxins/furans involves binding to the aryl hydrocarbon (Ah) receptors in fish and wildlife causing change in gene expression which leads to carcinogenic or teratogenic effects. However, invertebrates and plants lack the Ah receptor (Hahn et al., 1994, West et al., 1997) and therefore, dioxins/furans are not considered to cause appreciable toxicity in plant and soil invertebrate communities.

4.1.8 Polycyclic Aromatic Hydrocarbons

Frequently, available publications dealing with the toxicity of polycyclic aromatic hydrocarbons (PAHs) to soil invertebrates were not relevant. For example, Efroymson et al. (2004) reviews the toxicity of total petroleum hydrocarbons (TPH), which is not addressed in the ERA, and Cortet et al. (2006) describes an excellent mesocosm study with PAH. This experiment was conducted with a single, relatively high concentration (half the LC50) and is therefore inappropriate for developing ERA screening values. Kaputcka (2004) reviews PAH risks at 45 hazardous waste sites and concludes that there is no evidence that PAHs in soils at hazardous waste sites pose risks to invertebrates, plants, or wildlife. No screening values were presented in this review. Soil screening values developed for PAHs are described below.

4.1.8.1 Plant Screening Values

Plant screening values were not readily available for most of the PAHs from the sources listed above, except for acenaphthene (Efroymson et al., 1997a) and benzo(a)pyrene (USEPA, 1999a). Plant screening values were developed for naphthalene based on toxicity studies available in literature as described below. However, due to the lack of available toxicity data for other PAHs, the screening value for naphthalene was used as a surrogate for other low molecular weight (LMW) PAHs (2-methylnaphthalene, anthracene, acenaphthylene, fluorene, and phenanthrene) and the total LMW PAHs and the screening value for benzo(a)pyrene was used as a surrogate for other high molecular weight (HMW) PAHs (benzo(a)anthracene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, chrysene, fluoranthene, indeno(1,2,33-c,d)pyrene, and pyrene) and the total HMW PAHs.

For naphthalene, empirical data for naphthalene were available in the ECOTOX database (USEPA, 2007a). The study reporting ecologically relevant adverse effects at the lowest concentration was used to develop screening values for naphthalene. Hulzebos et al. (1993) reported a 7-day EC50 of 100 mg/kg for reduced biomass in lettuce. This study tested nearly 40 organic contaminants in both soil and a nutrient solution to determine the relationship between toxicity thresholds in both matrices. Conservatively assuming an EC50 of 100 mg/kg equivalent to a LOAEC for more serious adverse effects, a UF of 10 was applied to extrapolate to a NOAEC resulting in a value of 10 mg/kg, which was used as the plant screening value for naphthalene, the individual LMW PAHs, and total LMW PAHs. For HMW PAHs, the screening value of 1.2 mg/kg reported by USEPA (1999b) for benzo(a)pyrene was selected as the screening value for all individual HMW PAHs and total HMW PAHs.

As requested by the agencies, other PAH toxicity studies were reviewed; however, they were not used in this ERA as the screening values selected were considered more conservative. Toxicity studies cited in Jensen and Sverdrup (2003) reported EC50 concentrations ranging from 25 mg/kg to over 1,000 mg/kg for LMW PAHs such as acenaphthene, anthracene, naphthalene, and phenanthrene. HMW PAHs were generally not found to be phytotoxic even at test concentrations of 25,800 mg/kg. Sverdrup et al. (2003) reported EC50 concentrations ranging from 150 mg/kg to over 1000 mg/kg for LMW PAHs such as phenanthrene and fluorine and EC50 concentrations ranging from 640 mg/kg to over 1000 mg/kg for HMW PAHs such as fluoranthene. HMW PAHs were generally not found to be phytotoxic even at test concentrations of 25,800 mg/kg. Sverdrup et al. (2007) also found low toxicity to plants exposed to

benzo(a)pyrene and reported the lowest estimated NOAEC and LOAEC as 86 mg/kg and 470 mg/kg, respectively for *Brassica alba* (mustard).

4.1.8.2 Soil Invertebrate Screening Values

Soil invertebrate screening values were not readily available for most of the PAHs from the sources listed above except for pyrene (Beyer, 1990). Soil invertebrate screening values were developed for naphthalene and benzo(a)pyrene based on toxicity studies available in literature as described below. However, due to the lack of available toxicity data for other PAHs, the screening value for naphthalene was used as a surrogate for other LMW PAHs, and the screening value for benzo(a)pyrene was used as a surrogate for other HMW PAHs.

For naphthalene, USEPA Region 4 (USEPA, 2001) recommends a screening value of 0.1 mg/kg. This value is based on U.S. Fish and Wildlife Services (USFWS) "A" value, or typical ambient or "background" concentration in the absence of any local sources of contamination (Beyer, 1990). As discussed above, screening values based on ambient concentrations were not selected for this ERA. Empirical data are not available for naphthalene, however toxicity data for other LMW PAHs (phenanthrene and fluorene) are available from Sverdrup et al. (2001) and Sverdrup et al. (2002). NOAECs for decreases in reproductive success were reported for springtail and oligochaetes. The lowest NOAEC of these LMW PAHs is 14 mg/kg for springtail exposed to fluorene. This value is nearly three times the USFWS "B" value of 5 mg/kg for naphthalene. A Dutch MPC of 0.12 mg/kg was reported for anthracene (as cited in Kalf et al., 1995) based on toxicological data. However, on reviewing the original source, this value could not be confirmed (a toxicity study based on plants with a NOAEC of 150 mg/kg was reported) and therefore, not considered appropriate for the ERA. Dutch MPCs are available for other LMW PAHs (naphthalene and phenanthrene), however, these are derived from MPCs for water and therefore, not considered appropriate for the ERA. No additional empirical data were available, and therefore, the USFWS "B" value of 5 mg/kg was selected as the soil invertebrates screening value for naphthalene, other LMW PAHs, and total LMW PAHs.

For benzo(a)pyrene, empirical data from ECOTOX database (USEPA, 2007a) were used to develop a soil invertebrate screening value. In a study by Achazi et al., (1995), a LOAEC of 10 mg/kg was reported based on adverse reproductive effects in earthworms. The Dutch MPC for benzo(a)pyrene was also based on this same study. A Dutch MPC is also available for benzo(a)anthracene but this value is based on a crustacean study and therefore, not considered appropriate for the ERA. USFWS (Beyer, 1990), report a "B" value of 1.0 mg/kg based on the level of contamination that requires additional study to determine if unacceptable risks are possible. Empirical data were available for other HMW PAHs (pyrene and fluoranthene) from Sverdrup et al. (2001) and Sverdrup et al. (2002). NOAECs for decreases in reproductive success were reported for springtail and oligochaetes. The lowest NOAEC of these HMW PAHs is 13 mg/kg for springtail exposed to pyrene. This value is more than 10 times the USFWS "B" value of 1.0 mg/kg for benzo(a)pyrene. USEPA Region 4 (USEPA, 2001) recommends a soil screening value of 0.1 mg/kg; however, this value is based on USFWS's "A" value, representative of ambient concentrations (Beyer, 1990). As discussed above, screening values based on ambient concentrations were not selected for this ERA. USEPA (1999a) recommends a screening value of 25 mg/kg for soil invertebrates. Based on the guidelines for developing a screening value for this ERA, the study by Achazi et al., (1995), was selected as the most conservative and appropriate for benzo(a)pyrene. Following guidelines, a UF of 10 was applied

to extrapolate from the LOAEC to a chronic NOAEC of 1.0 mg/kg which was used as the soil invertebrate screening value for benzo(a)pyrene, other HMW PAHs, and total HMW PAHs.

4.2 Sediment Screening Values

Sediment screening values for sediment-dwelling invertebrates selected for the ERA are presented in Table U.A2-2.

The selection of sediment screening levels is not a simple or straightforward task, as “appropriate” sediment screening levels or guidelines are a source of great debate. A number of approaches have been developed and used as screening and assessment tools at sediment sites across the country. While there is no clear consensus on which approach is most appropriate for sediment screening, it is clear that an approach with a sound technical basis and adequate conservatism (i.e., margin of safety) is needed to support the screening step so that that defensible decisions can be made that are protective of environmental resources. An empirical approach resulting in consensus-based sediment quality guidelines (SQGs) for freshwater ecosystem (MacDonald et al., 2000) was given preference as these SQGs were designed to predict toxicity to benthic invertebrates in freshwater sediments and are recommended by USEPA and state guidance.

Approaches developed based on empirical data relationships include effects levels (Smith et al., 1996), screening level concentrations (Persuad et al., 1993), the effects range approach (Long and Morgan, 1991), and the apparent effects threshold approach (Cubbage et al., 1997). Screening levels or guideline values developed using these approaches can vary by several orders of magnitude depending on the intent of their use and the derivation procedure (MacDonald et al., 2000; Smith et al., 1996). Additionally, previously developed empirical approaches do not provide a reliable means to derive criteria or screening levels that reflect contaminant specific response thresholds (due to un-addressed co-contaminant and chemical mixture issues), and they don't incorporate site-specific factors that influence bio-availability (MacDonald et al., 2000; DiToro et al., 1991), and therefore exposure. Numerous studies have demonstrated the lack of correlation between sediment dry weight concentrations and toxicity (DiToro et al 1991).

The empirically based TECs and PECs were developed as “consensus-based” screening concentrations (MacDonald et al., 2000). Consensus-based SQGs were developed to provide a unifying synthesis of existing sediment guidelines and to account for chemical mixtures (MacDonald et al, 2000). The two sets of consensus-based SQGs developed are the TEC (below which adverse effects are not expected to occur) and the PEC (above which adverse effects are expected to occur). These levels were derived using an averaging approach based on similar thresholds from the following published sources:

- Effects-Level SQGs (TELs and PELs; Smith et al., 1996);
- *Hyalella azteca* Effects-Level SQGs (TEL-HA28 and PEL-HA28; Ingersoll et al., 1996 and USEPA, 1996);
- Effects-Range SQGs (ER-Ls and ER-Ms; Long and Morgan, 1991);
- Screening-Level Concentration SQGs (LELs and SELs; Persuad et al., 1993); and

Since the development of TECs and PECs involved the consideration of other previously derived empirically based sediment screening levels, TECs and PECs are based on associations observed between measures of adverse biological effects and the concentrations

of potential chemicals of concern in sediments. The consensus-based SQGs were evaluated by MacDonald (MacDonald et al. 2000) to determine if they would be effective tools for predicting sediment quality in freshwater ecosystems. Based on the evaluation criteria, TECs and PECs for most of the individual chemicals and mixtures were considered reliable as predictive tools (i.e. predictive ability was greater than 75 percent). This reliability was associated with the narrative intent of TECs and PECs (i.e., sediment samples were predicted to be not toxic if the measured concentration of a chemical was less than its corresponding TEC and similarly, sediment samples were predicted to be toxic if the measured concentration of a chemical was greater than its corresponding PEC).

The ORNL guidance document (Jones et al., 1997) provides ecotoxicological screening benchmarks for sediment applicable to a range of aquatic organisms. The approach used to develop sediment screening values by ORNL is summarized in Section 4.1.

Screening values reported in other guidance or documents are generally a compilation of values from other sources based on similar approaches and hierarchies and are not discussed in detail herein.

Low and high sediment screening values for the protection of sediment-dwelling invertebrates were selected or developed from the following sources listed in order of preference following the guidelines described in Section 2:

- Consensus-Based Sediment Quality Guidelines (SQGs) for Freshwater Ecosystem (MacDonald et al., 2000);
- ORNL: Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Sediment-Associated Biota (Jones et al., 1997);
- USEPA Supplemental Guidance to RAGS: Region 4 Bulletins, Ecological Risk Assessment (USEPA, 2001);
- NOAA Sediment Quality Goals (NOAA, 2006);
- USEPA Region 5 Ecological Screening Levels (USEPA, 2003);
- USEPA Region 6 Screening Level Ecological Risk Assessment Protocol, Appendix U (USEPA, 1999b);
- Canadian Council of Ministers of the Environment (CCME, 2002); Freshwater Sediment Quality Guidelines;
- Guidelines for the Protection and Management of Aquatic Sediment Quality in Ontario, Lowest Effects Levels (Persaud, 1993); and
- Dutch Environmental Quality Standards (EQS) MPCs, cited in Crommentuijn et al. (2000a).

Priority was given to screening values derived from empirical data; values derived through modeling or statistical extrapolation were given lower priority than listed in the above hierarchy. Additionally, Effects Range-Low and Effects Range-Median values from Long and Morgan (1991) and Long et al. (1995) for marine sediments were reviewed, however these values were not required because preferred freshwater values were already available for each CPEC in these sources.

MPCs for metals (Crommentuijn et al., 2000b) and Target and Intervention values from the Netherlands (MHSPE, 1994) were also consulted. No screening values were selected from these sources because metals screening values were not strictly toxicity based. Screening

values for metals from these sources included consideration of background metal concentrations as well as toxicity data, and were derived in a manner inconsistent with the approach taken in the ERA.

As requested by the agencies, USEPA's Procedures for the Derivation of Equilibrium Partitioning Sediment Benchmarks (ESBs) for the Protection of Benthic Organisms Compendium of Tier 2 Values for Nonionic Organics (USEPA, 2008) was also reviewed; however, no sediment screening values from this source were selected as the confidence in these values was considered low due to the lack of a comprehensive toxicity database.

Additionally, the Naval Facilities Engineering Service Center (NFESC) Amphibian Ecological Risk Assessment Guidance Manual was consulted for sediment screening levels for amphibians (ENSR International, 2004). Four screening values for metals (cadmium, copper, lead, and zinc) were available, but were not selected for use. The Manual explicitly recommends that generic published screening levels for benthic invertebrates should be used preferentially over values developed in the NFESC study. The values developed in the NFESC study were in some case three orders of magnitude higher than current sediment screening values. However, it must be noted that screening values for benthic invertebrates are considered very conservative for the protection of amphibians; but as this is a Tier 1 risk assessment, in addition to the lack of toxicity values protective of amphibians, screening values protective of benthic invertebrates will also be considered protective of amphibians.

The USACHPPM Wildlife Toxicity Reference Values (USACHPPM, 2006) were also consulted for amphibian screening values; however, no screening values were selected for use at the Site. Additionally, no values were selected from the Port Hueneme Amphibian Risk Assessment Guidance Manual (ENSR, 2004).

All of the screening criteria guidance listed above provide screening values that represent levels at or below which no significant adverse effects to sediment-dwelling biota are likely to occur. Screening values are available for most inorganic and organic chemicals likely to be present at the Site, with the exception of a few VOCs (1,2-dichloroethene, diisopropyl ether, Freon 113, methylcyclopentane, propanal, and tetrahydrofuran). However, VOCs are not usually of concern in sediments because, in general, they neither bind appreciably to particulates nor persist in aquatic environments. Additionally, VOCs which are evaluated via the inhalation pathway for wildlife; TRVs were developed separately for VOCs as presented in Attachment 4.

4.2.1 Inorganic Compounds

Sediment screening values for most of the inorganic CPECs (i.e., metals) were obtained from MacDonald et al. (2000). Low screening values were based on threshold effects concentrations (TECs) and high screening values were based on probable effects concentrations (PECs). See below in Section 4.2.8 for a summary of consensus-based SQGs for PAHs. Screening values for inorganic compounds were selected for this ERA based on similar reasoning.

Toxicity studies for effects of barium, molybdenum, selenium, thallium, and tin on sediment-dwelling invertebrates were not available in literature, and therefore, sediment screening values could not be developed for these CPECs.

4.2.2 Volatile Organic Compounds

Sediment screening values for most of the VOCs (1,1-dichloroethane, 4-methyl-2-pentanone, acetone, benzene, acetonitrile, carbon disulfide, ethylbenzene, MEK, methylene chloride, and trichloroethylene [TCE]) were obtained from USEPA Region 5 ecological screening levels (USEPA, 2003) as presented in Table U.A2-2. Additional high screening values were available for benzene, ethylbenzene, methylene chloride, and TCE as toxicity-based MPCs (Crommentuijn et al., 2000a).

Toxicity studies for effects of 1,2-dichloroethene, diisopropyl ether, Freon 113, methylcyclopentane, propanal, and tetrahydrofuran on sediment-dwelling invertebrates were not available in literature, and therefore, sediment screening values for these CPECs could not be developed.

4.2.3 Semi-Volatile Organic Compounds

No SVOC compounds were selected as CPECs for sediment; therefore no sediment screening values for SVOCs were derived for the ERA.

4.2.4 Organochlorine Pesticides

Sediment screening values for all the organochlorine pesticide CPECs were obtained from the sources listed above as presented in Table U.A2-2. Where available, TECs and PECs (MacDonald et al., 2000), were preferentially used as low and high screening values, respectively. If consensus-based SQGs were not available, other sources listed above were used as low and high screening values.

4.2.5 Polychlorinated Biphenyls

Sediment screening values for all the polychlorinated biphenyls (PCBs) were obtained from the sources listed above as presented in Table U.A2-2. Where available, TECs and PECs (MacDonald et al., 2000), were preferentially used as low and high screening values, respectively. ERA. Sediment screening values for PCB toxicity equivalent (TEQ) were not required.

4.2.6 Organophosphate Herbicides/Pesticides

No toxicity studies were available for dichlorprop and MCP. However, sediment MPCs based on EqP were available for these compounds from Crommentuijn et al. (2000a). No screening values or toxicity studies on the effects of 4-(2,4-dichlorophenoxy)butyric acid (2,4-DB) were available in literature, and therefore, sediment screening values could be developed for this CPEC.

4.2.7 Dioxin/Furans

There are no fish in any of the ponds onsite; therefore, values protective of fish were not developed. As mentioned earlier, dioxins/furans bind to the Ah receptors in fish and wildlife causing change in gene expression which leads to carcinogenic or teratogenic effects.

However, aquatic invertebrates lack the Ah receptor (Hahn et al., 1994, West et al., 1997) and therefore, it is considered inappropriate to use fish dioxin/furan screening values for the protection of aquatic invertebrates. A study by West et al., (1997), confirms previous investigations on the insensitivity of aquatic invertebrates to 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) exposures. Although amphibians have the Ah receptor, the affinity to bind to TCDD has been found to be low and therefore, considered relatively insensitive to TCDD (Lavine et al., 1995).

4.2.8 Polycyclic Aromatic Hydrocarbons

Sediment screening values for all the PAHs were obtained from the sources listed above as presented in Table U.A2-2. Where available, TECs and PECs (MacDonald et al., 2000), were preferentially selected as low and high screening values, respectively. If TECs and PECs were not available, ER-Ls and ER-Ms (Long et al., 1995) were selected as low and high screening values, respectively. If consensus-based SQGs or effects range SQGs were not available, other sources listed above were used as low screening value.

Several numerical SQGs have been developed for PAHs in freshwater sediments, which can vary by several orders of magnitude depending on the intent of their use and the derivation procedure (MacDonald et al., 2000; Smith et al., 1996). Consensus-based SQGs were developed to provide a unifying synthesis of existing sediment guidelines and to account for chemical mixtures (MacDonald et al., 2000). In this evaluation, consensus-based TECs and PECs were used to screen individual PAHs in site sediment data; TECs and PECs were developed from the following published numerical SQGs:

- Effects-Level SQGs (TEs and PELs; Smith et al., 1996);
- *Hyalella azteca* Effects-Level SQGs (TEL-HA28 and PEL-HA28; Ingersoll et al., 1996 and USEPA, 1996);
- Screening-Level Concentration SQGs (LEs and SELs; Persaud et al., 1993); and
- Sediment Quality Advisory Level SQGs (METs and TETs; EC and MENVIQ, 1992).

TECs and PECs for the individual PAHs detected were derived by calculating the geometric mean of the TEC or PEC type values from the SQGs listed above.

The sediment screening values described above were available from the literature sources and are presented in Table U.A.2-2. However, in order to be consistent with the risk assessment approach for ecological communities in soil and surface water and to comply with a request by USEPA, risks were estimated for the total LMW PAHs and total HMW PAHs using surrogate screening levels for these mixtures. For the total LMW PAHs, naphthalene was selected as a surrogate. For total HMW PAHs, the screening level for benzo(a)pyrene was used.

4.3 Surface Water Screening Values

Surface water screening values for aquatic life, aquatic plants, and amphibians selected for the ERA are presented in Table U.A2-3. Preference in selecting surface water screening values was given to promulgated numeric water quality criteria for priority toxic pollutants and other water quality standards provisions for the waters in the State of California (USEPA, 2000b). In 2000, USEPA promulgated this rule to fill a gap in California water quality standards which was

created in 1994 when the State's water quality control plans containing water quality criteria for priority toxic pollutants was overturned (USEPA, 2000b). The State of California has lacked numeric water quality criteria for many priority toxic pollutants as required by the Clean Water Act, which warranted this action by USEPA. As stated by USEPA (2000b) the federal criteria are legally applicable in the State of California for inland surface waters, enclosed bays and estuaries for all purposes and programs under the Clean Water Act.

Similarly, surface water screening values selected from USEPA's national recommended water quality criteria (NAWQC) for the protection of aquatic life pollutants. These criteria are published pursuant to Section 304(a) of the Clean Water Act (CWA) and are meant to provide guidance for states and tribes to use in adopting water quality standards.

The ORNL guidance document (Suter and Tsao, 1996) provides ecotoxicological screening benchmarks for surface water applicable to a range of aquatic organisms. ORNL guidance presents alternate surface water screening values for chemicals based on estimating concentrations causing significant effects. The upper-bound screening values are based on the acute NAWQC and the Secondary Acute Values (SAVs). The SAV concentrations were estimated with 80% confidence not to exceed the unknown acute NAWQC for those chemicals with no NAWQC. The lower-bound or chronic screening values were based on the chronic NAWQC, the Secondary Chronic Value (SCV), the lowest chronic values for fish and daphnids, the lowest effects concentrations (EC20) for fish and daphnids from chronic toxicity tests, the estimated EC20 for a sensitive species, and the concentration estimated to cause a 20% reduction in the recruit abundance of largemouth bass. ORNL guidance also presents Tier II screening values for chemicals with no NAWQC. The methods for Tier II screening values are described in the EPA's *Proposed Water Quality Guidance for the Great Lakes System* was applied (USEPA 1993). Tier II values were developed so that aquatic benchmarks could be established with fewer data than are required for the NAWQC. The Tier II values are concentrations that would be expected to be higher than NAWQC in no more than 20% of cases.

Screening values reported in other guidance or documents are generally a compilation of values from other sources based on similar approaches and hierarchies and are not discussed in detail herein.

Surface water screening values were selected or developed from the following sources listed in order of preference following the guidelines described in Section 2:

For aquatic life:

- USEPA Federal Register Title 40 CFR Part 131 Water Quality Standards Section 38- Established Numeric Criteria for Priority Toxic Pollutants for the State of California (USEPA, 2006a). Values for cadmium, chromium, copper, lead, nickel, and zinc were adjusted using the site-specific median hardness value of 150 mg/L (Central Valley Regional Water Quality Control Board [CVRWQCB], 2007);
- USEPA NAWQC (USEPA, 2006b); Values for cadmium, chromium, copper, lead, nickel, and zinc were adjusted using the site-specific median hardness value of 150 mg/L (CVRWQCB, 2007);

- ORNL: Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Aquatic Biota (Suter and Tsao, 1996);
- USEPA Supplemental Guidance to RAGS: Region 4 Bulletins, Ecological Risk Assessment (USEPA, 2001);
- San Francisco Regional Water Quality Control Board (SFRWQCB) ESLs: Freshwater Aquatic Habitat Goals (SFRWQCB, 2005);
- CVRWQCB Recommended Numerical Limits to Translate Water Quality Objectives (CVRWQCB, 2007); and
- Water Management Policies Guidelines Provincial Water Quality Objectives of the Ontario Ministry of Environment and Energy (MOEE, 1999 revision).

To provide a range screening value for aquatic life, the criterion continuous concentrations (CCC) and the criterion maximum concentrations (CMC) from the NAWQC (USEPA, 2006b) for aquatic life are also presented in Table U.A2-3. The CCC, which is an estimate of the highest concentration of a material in surface water to which an aquatic community can be exposed indefinitely without resulting in an unacceptable effect (USEPA, 2006b), was selected, if available, to be protective under conditions of chronic exposure. The CMC is a concentration protective of aquatic life under short-term exposure to a material (USEPA, 2006b).

The following sources were also consulted; however, no screening values were selected for use either because the study was not found to be appropriate for the ERA or data was available from a source of higher priority:

- Canadian Water Quality Guidelines for the Protection of Aquatic Life (CCME, 2006);
- Water Quality Guidelines for British Columbia (Ministry of the Environment, Lands, and Parks, 2001);
- Dutch EQS MPCs as cited in Crommentuijn et al. (2000a, b);
- World Health Organization Concise International Chemical Assessment Documents (WHO CICADs, 2007); and
- Individual toxicity studies.

For amphibians, lowest of:

- Empirical data cited in Ecotoxicology of Amphibians and Reptiles (Sparling et al., 2000);
- Empirical data from the Database of Reptile and Amphibian Toxicology Literature (RATL) (Pauli et al. 2000); and
- Empirical data from the ECOTOX database (USEPA, 2007a).

For aquatic plants:

- ORNL: As green algae are the most abundant aquatic plant life at the Site, the screening values reported by Suter and Tsao (1996) are considered appropriate for the Site as these are generally based on toxicity tests using algae; and
- Empirical data from the ECOTOX database (USEPA, 2007a).

4.3.1 Inorganic Compounds

Surface water screening values developed for inorganic CPECs (i.e., metals) are described below.

4.3.1.1 Aquatic Life

Surface water screening values protective of aquatic life for metals and inorganic compounds were obtained from the sources listed above as presented in Table U.A2-3. Many of the screening values for metals were based on ambient water quality criteria for California waters (USEPA, 2006); these values are hardness dependent. Screening values were adjusted for site-specific median hardness of 150 mg/L calcium carbonate (CaCO₃).

4.3.1.2 Amphibians

No published screening values were available for metals, and therefore, toxicity studies from the sources listed above were reviewed, and amphibian screening values were developed following the guidelines and use of uncertainty factors described in Sections 2 and 3, respectively. The most conservative and appropriate screening value was then selected for the protection of amphibians as presented in Table U.A2-3.

For antimony, arsenic, beryllium, cadmium, cobalt, silver, and zinc, the LC50 data from Pauli et al., (2000) were extrapolated to a chronic NOAEC using a UF of 100. For barium, copper, manganese, molybdenum, nickel, thallium, tin, and vanadium, the LC10 data from Sparling et al., (2000) were extrapolated to a chronic NOAEC using a UF of 100.

For chromium, lead, mercury, and selenium empirical data available from ECOTOX database (USEPA, 2007a) were used to develop surface water screening values for amphibians. Acute 7-day LC50 toxicity studies conducted by Birge et al. (1979) for effects on eastern narrow-mouthed toad were selected. Although this test was conducted from fertilization through four days post-hatch, amphibian embryos are frequently less sensitive to contaminants than emerged hatchlings or tadpoles because the egg provides a protective barrier that reduces exposure to some contaminants present in the water. Therefore, this study was not considered to be conducted on the most sensitive life stages, and a UF of 100 was applied to extrapolate the acute LC50 to a chronic NOAECs, resulting in values of 0.3 micrograms per liter (µg/L) for chromium, 0.4 µg/L for lead, 0.01 µg/L for mercury, and 0.9 µg/L for selenium. These values were used as the surface water screening value for amphibians.

Toxicity studies for the effects of cyanide on amphibians were not available in literature, and therefore, surface water screening values for this CPECs could not be developed.

4.3.1.3 Aquatic Plants

Surface water screening values protective of aquatic plants for inorganic CPECs (i.e., metals) were obtained from Suter and Tsao (1996) as presented in Table U.A2-3. Barium is not considered toxic to aquatic plants; a NOAEC-based surface water screening value of 15,000,000 µg/L protective of aquatic plants for barium was obtained from the CICAD database (WHO CICAD, 2007); no UF was required.

4.3.2 Volatile Organic Compounds

4.3.2.1 Aquatic Life

Surface water screening values protective of aquatic life for VOCs were obtained from the sources listed above as presented in Table U.A2-3. A summary of VOC toxicity data and screening levels published by Rowe et al (1997) was also reviewed. Toxicity data in this paper were compiled from the ECOTOX database, which was consulted independently for the ERA. No additional screening levels were found in this source. Toxicity studies for effects of nonanal and propanal on aquatic life were not available in literature, and therefore, surface water screening values for these CPECs could not be developed.

4.3.2.2 Amphibians

For acetone, empirical data from ECOTOX database (USEPA, 2007a) were used to develop screening value for amphibians. Bridges (2000) report a 6-day LOAEC of 198 µg/L for metamorphosis in leopard frog. A UF of 10 was applied to extrapolate the LOAEC to a NOAEL, resulting in a value of 19.8 µg/L, which was used as the amphibian surface water screening value for acetone.

For acetonitrile, no empirical data or published screening values were available. A tadpole 96 hour LC50 of 11600 µg/L for acrylonitrile was available from Pauli et al. (2000) and due to close structural similarities, acrylonitrile was selected as a surrogate for acetonitrile. A UF of 100 was applied to extrapolate from the acute LC50 to a chronic NOAEL, resulting in a value of 116 µg/L for the amphibian screening value for acetonitrile.

For carbon disulfide, empirical data from ECOTOX database (USEPA, 2007a) were used to develop screening value for amphibians. Ghate (1985) reports a 4-day LC50 of 120 µg/L for *Microhyla* (frog) embryos. A UF of 100 was applied to convert this acute median lethal concentration to a chronic NOAEC, resulting in a value of 1.2 µg/L, which was used as the amphibian surface water screening value for carbon disulfide.

For ethylene glycol, empirical data from ECOTOX database (USEPA, 2007a) were used to develop screening value for amphibians. A study by De Zwart and Slooff (1987) reports a 48-hour LC50 326,000 µg/L for *Xenopus laevis*. A UF of 100 was applied to convert this acute median lethal concentration to a chronic NOAEC, resulting in a value of 3,260 µg/L, which was used as the amphibian surface water screening value for ethylene glycol.

For trichloroethylene, the LC50 data from Pauli et al., (2000) were extrapolated to a chronic NOAEC using a UF of 100. For 1,1-dichloroethane and methylene chloride, the LC10 data from Sparling et al., (2000) were extrapolated to a chronic NOAEC using a UF of 100.

Toxicity studies for effects of 1,2-dibromoethane, 1,2-dichloroethene, acetonitrile, methyl isobutyl ketone, nonanal, and propanal on amphibians were not available in literature, and therefore, surface water screening values for these CPECs could not be developed.

4.3.2.3 Aquatic Plants

For acetone, empirical data from ECOTOX database (USEPA, 2007a) were used to develop screening value for aquatic plants. A study by Shubert et al., (1995) reports a 2-hour EC25 of 12.2 mg/L for tobacco plant germination. Due to the short exposure duration and adverse endpoint, this study was considered acute, and therefore, a UF of 100 was applied to convert the acute lethal concentration to a chronic NOAEC, resulting in a value of 0.122 mg/L or 122 µg/L which was used as the aquatic plant surface water screening value for acetone.

For ethylene glycol, empirical data from ECOTOX database (USEPA, 2007a) were used to develop screening value for aquatic plants. The only study available was for lettuce germination in agar (Reynolds, 1977). Although the agar matrix is not preferred, the 3-day EC50 of 54,600 mg/L was used for lack of alternate data. Germination was considered a lethal endpoint, and therefore, a UF of 100 was applied to extrapolate the acute lethal concentration to a chronic NOAEC, resulting in a value of 546 mg/L, or 546,000 µg/L, which was used as the aquatic plant surface water screening value for ethylene glycol.

For methyl isobutyl ketone, empirical data from ECOTOX database (USEPA, 2007a) were used to develop screening value for aquatic plants. A lettuce germination study conducted in agar was selected (Reynolds, 1977). Although the agar matrix is not preferred, the 3-day EC50 of 41 mg/L was used for lack of alternate data. Germination was considered a lethal endpoint, and therefore, a UF of 100 was applied to extrapolate the acute lethal concentration to a chronic NOAEC, resulting in a value of 0.41 mg/L, or 410 µg/L, which was used as the aquatic plants surface water screening value for methyl isobutyl ketone.

For propanal, empirical data from ECOTOX database (USEPA, 2007a) were used to develop screening value for aquatic plants. A lettuce germination study conducted in agar was selected (Reynolds, 1977). Although the agar matrix is not preferred, the 3-day EC50 of 796 mg/L was used for lack of alternate data. Germination was considered a lethal endpoint, and therefore, a UF of 100 was applied to extrapolate the acute lethal concentration to a chronic NOAEC, resulting in a value of 7.96 mg/L, or 7960 µg/L, which was used as the aquatic plants surface water screening value for propanal.

For trichloroethylene, empirical data from ECOTOX database (USEPA, 2007a) were used to develop screening value for aquatic plants. A study by Shubert et al., (1995) reports a two-hour EC50 of 31.7 mg/L for tobacco plant germination. Due to the short exposure duration and adverse endpoint, this study was considered acute, and therefore, a UF of 100 was applied to convert the acute lethal concentration to a chronic NOAEC, resulting in a value of 0.317 mg/L, or 317µg/L, which was used as the aquatic plant surface water screening value for trichloroethylene.

Toxicity studies for effects of 1,1-dichloroethane, 1,2-dibromoethane, 1,2-dichloroethene, acetonitrile, carbon disulfide, methylene chloride, and nonanal on aquatic plants were not available in literature, and therefore, surface water screening values for these CPECs could not be developed.

4.3.3 Semi-Volatile Organic Compounds

Surface water screening values for SVOCs are described below.

4.3.3.1 Aquatic Life

Surface water screening values protective of aquatic life were available for bis(2-ethylhexyl)phthalate and bis(2-chloroethyl)ether from the sources listed above.

No surface water screening values were available for n-nitrosodipropylamine, however a value was available for n-nitrosodiphenylamine (Suter and Tsao, 1996). N-nitrosodiphenylamine was selected as a surrogate based on structural and functional similarities. Therefore the surface water screening value of 7.0 µg/L was selected for n-nitrosodipropylamine.

Toxicity studies for effects of n-nitrosopyrrolidine and n-nitrosodiethylamine on aquatic life were not available, and therefore, surface water screening values for these CPECs could not be developed.

4.3.3.2 Amphibians

Surface water screening values protective of amphibians were not available for most of the SVOCs except for bis(2-ethylhexyl)phthalate. Empirical data from ECOTOX database (USEPA, 2007a) were used to develop a screening value. A study by Birge et al. (1978) reports an 8-day LC50 of 3880 µg/L for toad (*Bufo woodhouseri fowleri*). A UF of 100 was applied to extrapolate this acute median lethal concentration to a chronic NOAEC, in a value of 38.8 µg/L, which was used as the amphibian surface water screening value for bis(2-ethylhexyl)phthalate.

Toxicity studies for effects of bis(2-chloroethyl)ether, n-nitrosodiethylamine, n-nitrosodipropylamine, and n-nitrosopyrrolidine on amphibians were not available, and therefore, surface water screening values for these CPECs could not be developed.

4.3.3.3 Aquatic Plants

Surface water screening values protective of aquatic plants were not available for any of the SVOCs. However, toxicity data were available for diethylphthalate from ECOTOX database (USEPA, 2007a), which were used as a surrogate for bis(2-ethylhexyl)phthalate based on their shared phthalate chemical structure. A study by Adema and Henzen (2001) reports a 16-day NOAEC of 3200 µg/L for reduced growth in lettuce. Due to the sensitive endpoint and calculation of a NOAEC, no UFs were applied. Therefore, the NOAEC of 3200 µg/L. was used as the aquatic plant surface water screening value for bis(2-ethylhexyl)phthalate.

Toxicity studies for effects of bis(2-chloroethyl)ether, n-nitrosodiethylamine, n-nitrosodipropylamine, and n-nitrosopyrrolidine on aquatic plants were not available, and therefore, surface water screening values for these CPECs could not be developed.

4.3.4 **Dioxin/Furan Toxic Equivalent**

Surface water screening values based on water quality standards were not available for aquatic invertebrates for dioxin/furan TEQ. As explained earlier (Section 4.2.7), due to the lack of the Ah receptor, aquatic invertebrates are considered to be relatively insensitive to dioxins (Hahn et al., 1994; West et al., 1997). It must be noted that there are no fish in the ponds and it is considered inappropriate to use fish dioxin/furan TEQ screening value for the protection of aquatic invertebrates due to the mechanism of toxicity of dioxin/furan TEQ.

Toxicity studies for the effects of dioxin/furan TEQ on amphibians were not available, and therefore, surface water screening values for these CPECs could not be developed. As explained earlier (Section 4.2.7), amphibians are relatively insensitive to dioxins due to low affinity to bind to the Ah receptor (Lavine et al., 1995). For plants, no screening level was required, as dioxins/furans have not been found to be toxic to plants.

Although dioxin/furan congeners were selected as CPECs, individual congeners were not evaluated in the ERA. USEPA proposed that the CSC use 0.0006 µg/L as its TCDD surface water screening value for ecological communities. However, on review of individual congener data, TCDD was not detected in surface water and thus, the benchmark recommended by USEPA was not required in the ERA. Using the screening level for aquatic communities for TCDD without a TEF is considered inappropriate and will likely overestimate risks. As mentioned earlier, TEFs are not applicable for aquatic receptors (due to lack of Ah receptor).

4.3.5 Polycyclic Aromatic Hydrocarbons

Surface water screening values developed for PAHs are described below.

4.3.5.1 Aquatic Life

For PAHs, surface water screening values protective of aquatic life were obtained from the sources listed above as presented in Table U.A2-3. To assess total LMW PAHs, naphthalene was selected as a surrogate; for HMW PAHs, benzo(a)pyrene was used.

4.3.5.2 Amphibians

For naphthalene, empirical data from the RATL database (Pauli et al., 2000) were used to develop a screening value for amphibians following the guidelines discussed in Section 2. Edmisten and Bantle, (1982, as cited in Pauli et al., 2000), report a 4-day LC50 of 2100 µg/L for *Xenopus laevis*. A UF of 100 was applied to extrapolate the acute median lethal concentration to a chronic NOAEC, resulting in a value of 21 µg/L, which was used as the amphibian surface water screening value for naphthalene. Due to the lack of toxicity data, the screening value for naphthalene was used as a surrogate for other LMW PAHs and total LMW PAHs. As requested by the agencies, studies by Monson et al. (1999), Walker et al. (1998), and Hatch and Burton (1998) were also reviewed. However, the screening values that could be developed from these studies would be less conservative than the one already selected for the Site. Additionally, these are based on fluoranthene which is not a CPEC at the Site.

For benzo(a)pyrene, empirical data from ECOTOX database (USEPA, 2007a) were used to develop a screening value. Propst et al. (1997) report a 4-day EC50 of 8700 µg/L for developmental deformities in *Xenopus laevis*. The study was conducted on embryonic frogs and was considered a LOAEC. Therefore, a UF of 10 was applied to extrapolate to a NOAEC, resulting in a value of 870 µg/L, which was used as the amphibian surface water screening value for benzo(a)pyrene. Due to the lack of toxicity data, the screening value for benzo(a)pyrene was used as a surrogate for other HMW PAHs and for total HMW PAHs.

4.3.5.3 Aquatic Plants

For naphthalene, a NOAEC-based screening value of 33,000 µg/L reported by Suter and Tsao (1996) was used. No UFs were applied, resulting in a value of 33,000 µg/L, which was used as the aquatic plant surface water screening value for naphthalene, other LMW PAHS, and total LMW PAHs.

For benzo(a)pyrene, empirical data from ECOTOX database (USEPA, 2007a) were used to develop a screening value. Fiskesjo et al. (1981) report a 5-day EC50 of 8,500 µg/L for reduced growth in onion. The endpoint was considered LOAEC, and therefore, a UF of 10 was applied to extrapolate to a chronic NOAEC, resulting in a value of 850 µg/L, which was used as the aquatic plant surface water screening value for benzo(a)pyrene. Due to the lack of toxicity data, the screening value for benzo(a)pyrene was used as a surrogate for other HMW PAHs and total HMW PAHs.

5.0 TOXICITY REFERENCE VALUES FOR WILDLIFE

Following USEPA guidance (1997), wildlife TRVs were developed based on population-level assessment endpoints such as survival, reproductive, development, and growth endpoints for wildlife. TRVs were developed for the protection of birds and mammals from CPEC exposure at the Site following appropriate guidance (USEPA, 1999a; USEPA, 2007c) and included a review of toxicity benchmarks from standard sources, where available, and the selection of TRVs based on studies with appropriate endpoints. Differences in body weight between the site-specific wildlife receptor and the laboratory animals used in the study to develop the TRVs were not considered significant. Therefore, scaling factors were not used in developing TRVs for this ERA.

A range of TRVs were developed in order to estimate a range of risks. As mentioned earlier (Section 2), low TRVs were preferably based on chronic NOAELs, with an emphasis on studies that measured effects on survival, reproductive, development and growth endpoints, applicable to the protection of wildlife populations. If NOAELs were not available or reported, the LOAELs were extrapolated to develop NOAELs using UFs as described in Section 3. With the exception of BTAG values which are based on mid-range effects levels, high TRVs for the ERA were preferably based on chronic LOAELs for reproductive, growth, and survival endpoints

In order to meet the objectives of the ERA, TRVs protective of wildlife were selected or developed from the literature sources listed below. Where available, TRVs for CPECs were selected based on sources recommended in ERA guidance documents (USEPA, 1999a; CalEPA, 1996); sources from regulatory guidance were referenced for these TRVs and study details are not provided in this attachment. TRVs for CPECs that were not readily available but were developed based on literature review and following guidelines described in Section 2 are described in the following sections.

To select TRVs or studies to develop TRVs for mammals and birds, a hierarchy of the sources was established for this ERA. The objective of the hierarchy is to ensure that appropriate, conservative, and where available, published/promulgated values are preferentially selected instead of selecting the lowest available screening value, which could be based on varying levels of confidence. The hierarchy for selecting TRVs for the CPECs was based on USEPA guidance (USEPA, 1999b), which gives highest priority to toxicity values developed and/or adopted by federal and/or state regulatory agencies followed by toxicity values published in the scientific literature. As discussed with USEPA prior to the submittal of the Draft RI Report (CSC, 2008), preference would be given to the EcoSSL based TRVs developed by USEPA (2007c) followed by BTAG TRVs; and TRVs from other sources would follow.

As described in Section 4, the EcoSSL derivation process and thus the selection of EcoSSL TRVs, was considered sound and current as it was developed by a multi-stakeholder workgroup consisting of federal, state, consulting, industry, and academic participants led by the USEPA. The methods for deriving the oral TRVs needed for calculation of EcoSSLs for mammals and birds are included in the guidance (USEPA, 2007c) as four standard operating procedures (SOPs): (1) EcoSSL SOP #3 *Literature Search and Retrieval* (Attachment 4-2); (2) EcoSSL SOP #4 *Literature Review, Data Extraction and Coding* (Attachment 4-3); (3) EcoSSL SOP #5 *Data Evaluation* (Attachment 4-4); and EcoSSL SOP #6 *Derivation of the Oral TRV* (Attachment 4-5).

SOP #4 describes the procedures used for review and extraction of data from toxicological studies identified as a result of SOP #3 and also serves as a user's manual for the web-based data entry system used to guide the data extraction process. The extracted data are evaluated (scored) for their usefulness in establishing an oral TRV according to procedures provided in SOP #5. The extracted and scored data are then used to derive TRVs for mammals and birds, according to the procedures outlined in SOP #6.

The BTAG TRVs were developed for assessing risk to mammals and birds in 1997 (EFA, 1997). The Navy/BTAG TRV workgroup selected biological effects that primarily related to growth, reproduction, and development; however, all effects deemed ecologically relevant were considered when developing TRVs. The TRVs were selected from the published literature following a consensus effort among the Navy, Navy consultants, and several regulatory and natural resource trustee agencies including the USEPA, DTSC, Regional Water Quality Control Board (RWQCB), Office of Environmental Health Hazard Assessment (OEHHA), NOAA, USFWS, and the California Department of Fish and Game (DFG). The EcoSSL TRVs are endorsed by USEPA and therefore, have the highest preference in the hierarchy of TRVs. Additionally, the EcoSSLs included a more thorough, rigorous, and recent literature search and analysis process.

The rationale for developing BTAG TRVs is described in detail in the CalEPA guidance (EcoNote 4; CalEPA, 2000). In summary, BTAG TRVs are accepted by regulatory agencies for use in ERAs, and allow risk assessors to expedite agency review and approval of ERAs used to support remedial investigations. The BTAG TRVs allow for consistency and efficiency for conducting ERAs at different sites and limiting resources required by regulators to review alternate TRVs developed by the risk assessors. A substantial amount of effort was invested by the different parties to develop the BTAG TRVs including a comprehensive review of scientific journals by designated working groups. The USEPA, the DTSC, and the other agencies listed above endorse and recommend the use of the BTAG TRVs and these TRVs are applicable at any hazardous waste site with ecological exposure pathways to mammals and birds. Therefore, BTAG TRVs have the second highest preference in the hierarchy of TRVs.

TRVs reported in other guidance or documents are generally a compilation of values from other sources based on similar approaches and hierarchies and are not discussed in detail herein.

TRVs for mammals and birds selected for the ERA including the BTAG TRVs are presented in Table U.A2-4. Sources for TRVs selected or developed for this ERA are listed below in order of preference:

- USEPA EcoSSL Guidance (USEPA, 2007c);
- USEPA Region IX BTAG and U.S. Navy (CalEPA, 2000, 2002a,b);
- ORNL: Toxicological Benchmarks for Wildlife (Sample et al., 1996);
- Screening Level Ecological Risk Assessment Protocol, Appendix U (USEPA, 1999b);
- USEPA Supplemental Guidance to RAGS: Region 4 Bulletins, Ecological Risk Assessment (USEPA, 2001);
- Empirical data, meeting the guidelines described above (Section 2), listed hierarchically in the following databases:
 - Agency for Toxic Substances and Disease Registry (ATSDR, 2007) Toxicological Profiles. (U.S. Department of Health and Human Services);
 - USEPA's Integrated Risk Information System (IRIS) Database (USEPA, 2007b);

- Hazardous Substances Data Bank (HSDB, 2007; U.S. National Library of Medicine); and
- ECOTOX Database (USEPA, 2007a).
- Toxicity values from surrogate compounds.

The USACHPPM Wildlife TRVs (2006) were also reviewed. No values were selected from this source, as TRVs were available from sources with higher priority, as described above.

In the case of sources, such as the USEPA EcoSSL guidance (2007c), where only a NOAEL-based TRV was provided, paired LOAEL-based TRVs were selected according to the criteria described in Section 2. LOAEL-based TRVs were paired to EcoSSLs-based TRVs for birds and mammals as follows:

- A bounded NOAEL-based TRV was recommended and the LOAEL from the same study and endpoint was selected. For mammals this is the case for antimony, arsenic, DDT and metabolites), and nickel; for birds, copper, DDT (and metabolites) lead, and vanadium;
- The recommended NOAEL-based TRV was unbounded, and the lowest reproduction, growth, and survival LOAEL greater than the NOAEL-based TRV was selected. For mammals this is the case for beryllium, cadmium, copper, lead, and vanadium; for birds this is the case for arsenic;
- The recommended NOAEL-based TRV was derived from a LOAEL with a UF applied, and the LOAEL-based TRV was selected by removing the UF. For mammals and birds, this was the case for silver;
- The mammalian NOAEL-based TRV for chromium is the geometric mean of the reproduction and growth NOAELs. No bounded NOAELs or LOAELs were contained in the dataset however. Therefore, the lowest reproduction and growth LOAEL greater than mammalian low TRV for chromium was conservatively selected as the LOAEL-based TRV; and
- For CPECs in which NOAEL based TRVs were based on geometric means of endpoints, in agreement with the agencies, the LOAEL based TRVs were derived using the following step-wise approach:
 1. Calculating geometric means of bounded LOAELs for growth and reproduction endpoints only;
 2. Identifying the lowest bounded LOAEL for survival endpoints; and
 3. Selecting the lowest value from steps 1 and 2 above as the proposed LOAEL TRV.

For mammals this is the case for barium and cobalt; for birds, cadmium, chromium, cobalt, and nickel.

The TRVs described below were developed for CPECs found in soil and sediment, as these matrices constitute complete and significant exposure pathways for wildlife. For aquatic wildlife only, surface water is also a significant and complete exposure pathway. Additional TRVs for CPECs unique to deep soil (0-10 feet below ground surface [bgs]) were also developed deep burrowing mammals (e.g. badger). The following compounds were selected as CPECs in surface water, but not in soil or sediment: antimony, silver, 1,2-dibromoethane, ethylene glycol, nonanal, bis(2-chloroethyl)ether, N-nitrosodiethylamine, N-nitrosopyrrolidine, and dibenzo(a,h)anthracene. For antimony, silver, and dibenzo(a,h,)anthracene, TRVs were either

available or developed as described below. For the remaining CPECs, no TRVs were available from the sources listed above.

Note that CalEPA has requested that BTAG TRVs be considered where available. Although risks were primarily estimated based on EcoSSL TRVs in the ERA, where available, risks estimated using BTAG TRVs for those chemicals with EcoSSL TRVs are presented in Table U.A2-4 and discussed in the uncertainty analysis of the main ERA (Appendix U).

As requested by the agencies, Table U.A2-4 includes the species, endpoint, and primary study associated with each TRV proposed by the CSC and for the BTAG TRVs.

5.1 Inorganic Compounds

Low and high TRVs for mammals and birds for most the inorganic CPECs (i.e., metals) were obtained directly from the sources listed above as presented in Table U.A2-4 a. In general, metal TRVs have been thoroughly evaluated by USEPA and others (USEPA, 2007c; 1997; CalEPA, 2000, Sample et al., 1996), and therefore, not described in this attachment, except for cyanide TRV for mammals as described below.

For cyanide, a low mammalian TRV was available from Sample et al. (1996); however, a high TRV was unavailable for mammals. The ATSDR Toxicological Profile for cyanide (ATSDR, 2006a) contained several studies with more sensitive endpoints than reported in Sample et al. (1996). A 13-week National Toxicology Program (NTP) study (NTP, 1993) conducted with rats was the most sensitive ecologically relevant study and was selected to develop mammalian TRVs for the ERA. In this study, rats were exposed to sodium cyanide via drinking water. Decreased spermatogenesis in males was first observed at a dose of 12.5 mg/kg bw-day, with a NOAEL of 4.5 mg/kg bw-day. The study is considered by ATSDR to be highly reliable, with adequate replication, number of dose groups, and an exposure of chronic duration. No UFs were applied to these values, therefore the final low and high mammalian TRVs for cyanide are 4.5 and 12.5 mg/kg bw-day, respectively. Although, amenable cyanide was selected as a CPEC, no wildlife TRVs are available for this chemical. Potential risks to wildlife from exposure to amenable cyanide are assumed to be included in the risks from exposure to total cyanide and is discussed in the uncertainty section of the ERA.

5.2 Volatile Organic Compounds

Low and high TRVs for mammals and birds for most the VOCs were obtained directly from Sample et al., (1996) as presented in Table U.A2-4. For mammals, the test animal was used as a representative species.

Mammalian low and high TRVs or empirical data were not available for the following VOCs: 3,3,5-trimethylcyclohexanone, acetonitrile, Freon 113, methylcyclopentane, nonanal, and propanal. Therefore, TRVs could not be developed for these CPECs. Empirical data were not available for 1,1-dichloroethylene, 1,1-dichloroethane, 1,2-dichloroethene, and 4-methyl-2-pentanone (MIBK) to develop high TRVs for mammals.

Similarly, for birds, low and high TRVs or empirical data were not available for most of the VOCs, except 1,1-dichloroethane (Sample et al., 1996) and acetone (USEPA, 1999a). Therefore, TRVs could not be developed for most of these CPECs.

In general, TRVs from Sample et al., (1996) have been thoroughly evaluated and, therefore, not described in this attachment unless they were modified specifically for this ERA. TRVs for VOCs that were not published or readily available were developed from empirical data as described below.

5.2.1 1,1,1-Trichloroethane

For mammals, a published low mammalian TRV was available for 1,1,1-trichloroethane from Sample et al. (1996). This value of 1000 mg/kg-d for mouse was selected as the low TRV for the ERA. Additional data from the ATSDR database (ATSDR, 2006b) were available to derive a high mammalian TRV. A NTP study (2000) reports a 13-week mouse NOAEL and LOAEL of 1770 mg/kg-d and 3550 mg/kg-d, respectively. Exposure was via the food and resulted in a decrease in total body weight. Two chronic studies (NCI, 1977a; Maltoni et al., 1986) report unbounded LOAELs for mortality and growth somewhat below the NTP LOAEL of 3550; however both exposed rats and/or mice to 1,1,1-trichloroethane in oil via gavage. Feeding was considered a more ecologically realistic exposure route; therefore the NTP (2000) LOAEL was selected to derive the high TRV, according to the methods described in Section 2. No UFs were applied. Therefore the low and high TRVs for 1,1,1-trichloroethane are 1000 mg/kg-d and 3550 mg/kg-d, respectively.

For birds, no published TRVs or empirical data were available. However, published TRVs were available for 1,2-dichloroethane (Sample et al., 1996), which was used as a surrogate for 1,1,1-trichloroethane based on similar chemical structures. Therefore, the bird low and high TRVs for 1,1,1-trichloroethane were 17.2 mg/kg bw-day and 34.4 mg/kg bw-day, respectively.

5.2.2 1,1-Dichloroethane

For mammals, no published mammalian TRVs were available for 1,1-dichloroethane. However, empirical toxicity data for mammals were available from the ATSDR database (ATSDR, 1990). A National Cancer Institute (NCI, 1977b) study reports a 78-week LOAEL of 382 mg/kg bw-day for mortality in rats. Klaunig et al. (1986) report a 52-week NOAEL of 475 mg/kg bw-day for systemic organ toxicity and mortality, with exposure via drinking water in mice. In the NCI study (NCI, 1977), rats were gavaged with 1,1-dichloroethane in corn oil at concentrations six times greater than in the Klaunig study. Additionally, in the NCI study, both rats and mice were found to have no adverse organ-level effects (excluding cancer) at concentrations up to 1,442 mg/kg bw-day for mouse and 764 mg/kg bw-day for rat. Ad libitum ingestion of drinking water is considered an ecologically realistic route of exposure, therefore the Klaunig study was selected for TRV development in the ERA. No UFs were applied to this chronic NOAEL, and therefore, the mammalian low TRV developed was 475 mg/kg bw-day for 1,1-dichloroethane. A UF of 0.1 was applied to extrapolate a LOAEL from the NOAEL, resulting in a high TRV of 4750 mg/kg bw-day.

For birds, no published TRVs or empirical data were available for avian species. However, published TRVs were available for 1,2-dichloroethane (Sample et al., 1996), which was used as a surrogate for 1,1-dichloroethane based on similar chemical structures. Therefore, the bird low and high TRVs for 1,1-dichloroethane were 17.2 mg/kg bw-day and 34.4 mg/kg bw-day, respectively.

5.2.3 1,2-Dibromoethane

For mammals, no published TRVs were available from the literature for this compound. However, empirical data for mammals were available in the ATSDR (2007) and IRIS (USEPA, 2007b). Amir et al (1977) reports a LOAEL of 4 mg/kg bw-day for transient sperm anomalies in bull after exposure to 1,2-dibromoethane by gavage for 20 days. Due to the route of exposure, relatively short exposure period, reversible nature of the endpoint, and dissimilarities of bull with ERA site receptors, the Amir (1977) study was not selected for TRV development. A study by the National Cancer Institute (NCI, 1978) reports a LOAEL of 27 mg/kg bw-day for testicular atrophy and hepatic peliosis in rats exposed to 1,2-dibromoethane for one year. This study was selected for development of the human reference dose (RfD), and was considered appropriate for TRV development for the ERA as well. A NOAEL was not reported by NCI (1978), therefore a UF of 10 was applied to convert the LOAEL to a NOAEL. The resulting low mammalian TRV for 1,2-dibromoethane is 2.7 mg/kg bw-day and the high TRV is 27 mg/kg bw-day.

For birds, no published TRVs were available from the literature and no empirical data were available to develop avian TRVs for this compound.

5.2.4 1,2-Dichlorobenzene

For mammals, no published TRVs were available from the literature for this compound. There were no other suitable studies were available for this compound to develop TRVs. Due to lack of data, TRVs developed for the parent compound, chlorobenzene (see below) were used as surrogates (i.e., 19 mg/kg bw-day as low TRV and 38 mg/kg bw-day as high TRV).

For birds, no TRV was required for the ERA as this compound is a CPEC only in deeper soils (0-10 feet bgs) and birds are not exposed to soils at these depths.

5.2.5 1,2-Dichloroethene

For mammals, a low TRV for 1,2-dichloroethene was based on the published NOAEL-based TRV value of 45.2 mg/kg bw-day for mouse (Sample et al., 1996). No LOAEL-based TRV was available, therefore a UF of 0.1 was applied to extrapolate a high TRV from the low TRV. The resulting high TRV for 1,2-dichloroethene is 452 mg/kg bw-day.

For birds, no published TRVs were available from the literature and no empirical data were available to develop avian TRVs for this compound.

5.2.6 1,2-Dichloropropane

Although no published TRVs were available for 1,2-dichloropropane, empirical mammalian data were available in the IRIS and ATSDR databases (USEPA, 2007b; ATSDR, 1989c, ATSDR, 2007). Several studies suggest a subchronic LOAEL of 125 mg/kg bw-day for 1,2-dichloropropane. Kirk et al. (1989) found decreased body weight gain and decreased movement in pregnant rats and delayed ossification in the offspring after exposure to 1,2-dichloropropane by gavage during gestation. Multi-generational exposures via drinking water were found to result in decreased offspring weights and maternal toxicity, with estimated NOAEL and LOAELs of 100 mg/kg bw-day and 200 mg/k, respectively (Kirk et al, 1990). Anemia, anorexia, and

decreased body weight gain were observed in pregnant rabbits at 150 mg/kg bw-day, with delayed ossification in offspring at this dose as well. (Hanley et al., 1989). A NTP study with rats and mice given 1,2-dichloropropane by gavage for 13 or 105 weeks showed decreased body weight in rats at 125 mg/kg bw-day in the chronic study (NTP, 1986). The various NOAELs associated with these studies range from 30-60 mg/kg bw-day. The Kirk et al (1989) study reporting maternal and fetotoxic effects at 125 mg/kg bw-day, with a LOAEL of 30 mg/kg bw-day was selected to derive TRVs for the ERA. No UFs were applied, therefore the low and high TRVs for 1,2-dichloropropane are 30 mg/kg bw-day and 125 mg/kg bw-day, respectively.

For birds, no TRV was required for the ERA as this compound is a CPEC only in deeper soils (0-10 feet bgs) and birds are not exposed to soils at these depths.

5.2.7 1,2,3-Trichlorobenzene

For mammals, no published TRVs were available from the literature for this compound. There were no other suitable studies were available for this compound to develop TRVs. Due to lack of data, TRVs developed for the parent compound, chlorobenzene (see below) were used as surrogates (i.e., 19 mg/kg bw-day as low TRV and 38 mg/kg bw-day as high TRV).

For birds, no TRV was required for the ERA as this compound is a CPEC only in deeper soils (0-10 feet bgs) and birds are not exposed to soils at these depths.

5.2.8 1,3-Dichlorobenzene

For mammals, no published TRVs were available from the literature for this compound. There were no other suitable studies were available for this compound to develop TRVs. Due to lack of data, TRVs developed for the parent compound, chlorobenzene (see below) were used as surrogates (i.e., 19 mg/kg bw-day as low TRV and 38 mg/kg bw-day as high TRV).

For birds, no TRV was required for the ERA as this compound is a CPEC only in deeper soils (0-10 feet bgs) and birds are not exposed to soils at these depths.

5.2.9 1,4-Dichlorobenzene

For mammals, no published TRVs were available from the literature for this compound. There were no other suitable studies were available for this compound to develop TRVs. Due to lack of data, TRVs developed for the parent compound, chlorobenzene (see below) were used as surrogates (i.e., 19 mg/kg bw-day as low TRV and 38 mg/kg bw-day as high TRV).

For birds, no TRV was required for the ERA as this compound is a CPEC only in deeper soils (0-10 feet bgs) and birds are not exposed to soils at these depths.

5.2.10 Acetonitrile

For mammals, no published TRVs were available from the literature and no appropriate empirical data was found. A published TRV of 0.46 mg/kg bw-day is available for acrylonitrile from USEPA (1999b). Due to structural and functional similarities between the two compounds, acrylonitrile was selected as a surrogate for acetonitrile. The published TRV is based on a chronic LOAEL of 4.6 mg/kg bw-day for lesions and organ-level effects in rat (Quast, 1980).

USEPA (1999b) applied a UF of 10 to extrapolate from the LOAEL to a NAOEL. This approach is consistent with the UFs applied in the ERA. Therefore, the low and high TRVs for acetonitrile are 0.46 mg/kg bw-day and 4.6 mg/kg bw-day, respectively.

For birds, no published TRVs were available from the literature and no empirical data were available to develop avian TRVs for this compound.

5.2.11 Acrolein

For mammals, no published TRVs were available for acrolein from literature. However, empirical data for mammals were available in the ATSDR (2007), IRIS (USEPA, 2007b), and ECOTOX databases (USEPA, 2007a). The study reporting adverse reproduction, growth, or mortality effects at the lowest concentration according to the guidelines discussed above was selected for developing mammalian TRVs. A 2-year study with Norway rats resulted in a chronic LOAEL of 0.5 mg/kg bw-day for decreased survival after 1 year (Parent et al., 1992). No UFs were applied and the chronic LOAEL of 0.5 mg/kg bw-day was selected as the mammalian high TRV for acrolein. A UF of 10 was applied to extrapolate from the chronic LOAEL to a chronic NOAEL, resulting in a value of 0.05 mg/kg bw-day, which was used as the mammalian low TRV for acrolein.

For birds, no published TRVs were available from the literature and no empirical data were available to develop avian TRVs for this compound.

5.2.12 Carbon Disulfide

For mammals, no published TRVs were available for carbon disulfide from literature. However, limited empirical data were available. Jones-Price et al. (1984) report a LOAEL of 25 mg/kg bw-day for fetal resorption and hindlimb paralysis. In this 14-day gestational study, rabbits were gavaged with carbon disulfide dissolved in oil. Hardin et al. (1981) report an unbounded NOAEL of 11 mg/kg bw-day for fetal toxicity and teratogenic effects in rabbit. However, exposure in this study was via inhalation, and the NOAEL dose was calculated by IRIS (USEPA, 2007b) assuming standard inhalation rates and body weight for this species. The Hardin et al. (1981) study was used to calculate the RfD and was considered acceptable for use developing TRVs for the ERA. No UFs were applied, and therefore, the NOAEL of 11 mg/kg bw-day was used as the mammalian low TRV for carbon disulfide. No LOAEL was reported in the Hardin et al. (1981) study. Therefore, the LOAEL of 25 mg/kg bw-day from Jones-Price et al. (1984) was used as the mammalian high TRV.

For birds, no published TRVs were available from the literature and no empirical data were available to develop avian TRVs for this compound.

5.2.13 Chlorobenzene

For mammals, no TRVs were available in the literature for chlorobenzene. Empirical data for this compound were found in the IRIS (USEPA, 2007b) and ATSDR (2007) databases, and a study was selected for TRV derivation according to the criteria discussed above. A 13 week study by the Monsanto Company reports pathological changes in liver in dogs exposed to chlorobenzene by capsule at doses of 38 mg/kg bw-day. (Monsanto, 1967; Knapp, 1971). The NOAEL for these effects was 19 mg/kg bw-day. Higher doses resulted in progressive hepatic

degeneration and death. This study was selected to derive the human RfD, and was used to derive mammalian TRVs for the ERA. No UFs were applied to the reported NOAEL and LOAEL, therefore the low and high TRVs for chlorobenzene are 19 mg/kg bw-day and 38 mg/kg bw-day, respectively.

For birds, no TRV was required for the ERA as this compound is a CPEC only in deeper soils (0-10 feet bgs) and birds are not exposed to soils at these depths.

5.2.14 Diisopropyl Ether

For mammals, no published TRVs were available for diisopropyl ether from literature. Empirical data for mammals were also unavailable. However, as diisopropyl ether is structurally similar to methyl tert-butyl ether (MTBE); toxicity data for MTBE were used as a surrogate to develop mammalian TRVs for diisopropyl ether. ATSDR used an acute rat study by Bioresearch Labs (1990) to derive the oral minimal risk level for humans. A NOAEL and LOAEL of 40 and 400 mg/kg bw-day, respectively, were reported for lethargy following a single gavage dose. This endpoint was not considered adverse, however, and the exposure duration and route were not preferred. Chun et al. (1992) report a chronic rat NOAEL and LOAEL of 259 milligrams per cubic meter per day (mg/m³-day) and 1946 mg/m³-day, respectively, for increased liver and kidney weight and renal lesions. The 2-year exposure was via inhalation, and progressively severe effects were noted at higher concentrations. The NOAEL and LOAEL values were converted to doses of 165 mg/kg bw-day and 1240 mg/kg bw-day, respectively, by multiplying by the daily inhalation volume for rat (0.223 m³, IRIS/USEPA, 2007) and dividing by the mean rat body weight (0.35 kg, IRIS/USEPA, 2007). No UFs were applied, and therefore, for diisopropyl ether, the mammalian low TRV used was 165 mg/kg bw-day and the mammalian high TRV used was 1240 mg/kg bw-day.

For birds, no published TRVs were available from the literature and no empirical data were available to develop avian TRVs for this compound.

5.2.15 Diethyl Ether

For mammals, no published TRVs were available for diethyl ether from literature. Empirical data for mammals were also unavailable. However, as diisopropyl ether is structurally similar to diisopropyl ether and MTBE, toxicity data for MTBE were used as a surrogate to develop mammalian TRVs for diethyl ether.

For birds, no TRV was required for the ERA as this compound is a CPEC only in deeper soils (0-10 feet bgs) and birds are not exposed to soils at these depths.

5.2.16 Ethylbenzene

The TRV for ethylbenzene was based on a toxicity study conducted by Wolf et al. in 1956 (as cited in IRIS/USEPA, 2007a,b) where rats were administered oral doses of ethylbenzene five days a week for 182 days. The chronic LOAEL reported was 408 mg/kg-bw/day (NOAEL was not available) based on histopathologic changes in liver and kidney in rats which was further adjusted for the dosing schedule resulting in a value reported as 291 mg/kg bw-day. To develop ethylbenzene TRVs for use at the Site, the chronic LOAEL of 291 mg/kg bw-day was extrapolated to a chronic NOAEL using a UF of 10 resulting in a value of 29.1 mg/kg bw-day.

Therefore, the NOAEL value of 29.1 mg/kg bw-day was selected as the mammalian low TRV, and the LOAEL of 291 mg/kg bw-day was selected as the mammalian high TRV for ethylbenzene.

For birds, no published TRVs were available from the literature and no empirical data were available to develop avian TRVs for this compound.

5.2.17 Ethylene Glycol

There are no published mammalian TRVs for ethylene glycol in the literature. The IRIS (USEPA, 2007b) and ATSDR (2007) databases both cite a study by Blood et al. (1965) that was selected to derive TRVs for the ERA. In this study, rats were exposed to ethylene glycol in the diet for 2 years; however ATSDR reports that no test animals survived longer than 18 months. The IRIS database (USEPA, 2007b) assumes a food intake rate of 5% body weight per day for rat, resulting in the reported NOAEL and LOAEL of 100 mg/kg bw-day and 250 mg/kg bw-day, respectively. ATSDR (2007) reports a LOAEL of 500 mg/kg bw-day. For the ERA, the values reported in IRIS (USEPA, 2007b) was considered appropriate. Therefore, the NOAEL of 100 mg/kg bw-day was selected as the low mammalian TRV and the LOAEL of 250 mg/kg bw-day was selected as the high mammalian TRV for ethylene glycol.

For birds, no published TRVs were available from the literature and no empirical data were available to develop avian TRVs for this compound.

5.2.18 Isopropanol

For mammals, no published TRVs were available for isopropanol from literature. Empirical data for mammals were also unavailable. However, limited data for butanol and isobutyl alcohol were available in the IRIS database (USEPA, 2007b). These compounds are structurally similar to isopropanol. Therefore, TRVs developed for butanol were used as a surrogate for isopropanol. For butanol, a 13-week rat study reported a NOAEL and LOAEL for hypoactivity and ataxia of 125 mg/kg bw-day and 500 mg/kg bw-day, respectively (USEPA, 1986). The same study reported a NOAEL and LOAEL for isobutanol of 316 mg/kg bw-day and 1000 mg/kg bw-day, respectively, for those endpoints. The butanol NOAEL and LOAEL were conservatively chosen as surrogates for isopropanol. No UFs were applied, therefore the low TRV used was 125 mg/kg bw-day and the high TRV used was 500 mg/kg bw-day for isopropanol.

For birds, no published TRVs were available from the literature and no empirical data were available to develop avian TRVs for this compound.

5.2.19 Tert-Butyl Alcohol

No published mammalian TRVs for TBA were available from the literature. Empirical data for mammals were also unavailable. However, limited data for butanol and isobutyl alcohol were available in the IRIS database (USEPA, 2007b). These compounds are quite similar to TBA in that they are all structural isomers; each are alcohols attached to a four-carbon chain. For butanol, a 13-week rat study reported a NOAEL of 125 mg/kg bw-day and a LOAEL of 500 mg/kg bw-day for hypoactivity and ataxia (USEPA, 1986). The same study reported a NOAEL and LOAEL for isobutanol of 316 mg/kg bw-day and 1000 mg/kg bw-day, respectively, for the same endpoints. Isobutanol and TBA share greater structural similarity in that both have a

branched carbon chain. To maintain the conservative approach, butanol NOAEL and LOAEL were selected as surrogates for TBA. No UFs were applied, therefore the mammalian low TRV used was 125 mg/kg bw-day and the mammalian high TRV used was 500 mg/kg bw-day for TBA.

For birds, no published TRVs were available from the literature and no empirical data were available to develop avian TRVs for this compound.

5.2.20 Tetrahydrofuran

For mammals, no published TRVs were available for tetrahydrofuran. However, limited relevant empirical data were available. Mast et al. (1992) report a LOAEL of 1800 parts per million (ppm) for teratogenic effects and increased resorptions for mice exposed during gestation. A 4-week rat study by Komsta et al (1988) was conducted with tetrahydrofuran in the drinking water, predominantly enzymatic and biochemical endpoints were studied and only mild histological changes were observed. Hellwig et al. (2002) conducted a two-generation rat study with tetrahydrofuran in drinking water. This study reports systemic parental effects and mild developmental toxicity at 9000 ppm, with a NOAEL of 3000 ppm, equivalent to 700 mg/kg-d and 300 mg/kg-d, respectively, as reported by the authors. Due to the ingestion route of exposure and sensitive endpoints, the Hellwig study was selected to derive TRVs for the ERA. No UFs were applied, therefore the mammalian low TRV used was 300 mg/kg bw-day and the mammalian high TRVs used was 700 mg/kg bw-day for tetrahydrofuran.

For birds, no published TRVs were available from the literature and no empirical data were available to develop avian TRVs for this compound.

5.3 Semi-Volatile Organic Compounds

For mammals, published TRVs were available from Sample et al. (1996) for three SVOC compounds: bis(2-ethylhexyl)phthalate, diethylphthalate, and di-n-butylphthalate. For diethylphthalate, only a low TRV was available from this source and additional data were used to derive TRVs as described in the following sections. TRVs were unavailable for n-nitrosopyrrolidine.

For birds, published TRVs were available only for bis(2-ethylhexyl)phthalate and di-n-butylphthalate (Sample et al., 1996). Due to lack of toxicity data structural similarity, TRVs for di-n-butylphthalate were used as surrogate TRVs for diethylphthalate. TRVs were unavailable for benzoic acid, bis(2-chloroethyl) ether, and the five nitroso-compounds (n-nitrosodiethylamine, n-nitrosodimethylamine, n-nitrosodipropylamine, n-nitrosomethylethylamine, and n-nitrosopyrrolidine.)

5.3.1 Benzoic Acid

For mammals, published TRVs were unavailable for benzoic acid. Abundant empirical data from the IRIS database (USEPA, 2007b) were available for humans, and the RfD is based on human toxicity data. The limited animal studies presented show no clear evidence that benzoic acid causes adverse effects in rats and mice. Ignat'ev (1965) found no effects on body weight, survival, or gross or microscopic pathology in rats exposed to 80 mg/kg bw-d for 18 months. Although decreased water intake and "stress resistance" were observed at lower

concentrations, these endpoints were not considered adverse. Marquardt (1960) observed decreased food intake and body weights in chronic tests with rats exposed to 1.5% benzoic acid (750 mg/kg bw-d) in the diet. These relatively mild endpoints were the most severe found in rodents exposed to benzoic acid, therefore the Marquardt (1960) study was selected to develop TRVs for the ERA. The NOAEL for these effects was 1% benzoic acid (500 mg/kg bw-d), therefore the low and high TRVs for benzoic acid are 500 mg/kg bw-d and 750 mg/kg bw-d, respectively.

For birds, no published TRVs were available from the literature and no empirical data were available to develop avian TRVs for this compound.

5.3.2 Bis (2-chloroethyl) ether

No published TRVs are available for mammals for bis (2-chloroethyl) ether. Extremely limited empirical data (ATSDR, 1989b) are available from the ATSDR (2007) database with which to develop TRVs for the ERA. Weisburger et al (1981) report a LOAEL of 25 mg/kg bw-d for decreased body weight in rats exposed to this compound by gavage for 78 weeks. No NOAEL was reported, therefore a UF of 10 was applied to convert the LOAEL to a NOAEL. The resulting low mammalian TRV is 2.5 mg/kg bw-d and the high TRV for bis (2-chloroethyl) ether is 25 mg/kg bw-d.

For birds, no published TRVs were available from the literature and no empirical data were available to develop avian TRVs for this compound.

5.3.3 Diethylphthalate

For mammals, a low TRV of 4953 mg/kg bw-day is available for diethylphthalate (Sample et al., 1996). However, a high TRV was not available from this source, and therefore, additional paired data were considered for development of TRVs for the ERA. Empirical data were available for diethylphthalate in the ATSDR (2007) and IRIS (USEPA, 2007b) databases and from Pereira et al. (2006). Pereira et al (2006) exposed rats to diethylphthalate in the diet for three generations. Biochemical changes and alterations in liver histology were detected at doses as low as 0.57 mg/kg-d; biochemical endpoints were not considered adverse and hepatic effects were not observed at higher concentrations. Additionally, doses were changed in each generation, so clear effect thresholds are uncertain. Field et al. (1993) report a LOAEL of 3210 mg/kg bw-day for increased incidence of supernumary ribs in rat developmental exposure via ingestion. The NOAEL for this effect is 1910 mg/kg bw-day. Additionally, a 16-week rat study reports systemic changes in organ weight, with a NOAEL and LOAEL of 750 mg/kg bw-day and 3160 mg/kg bw-day, respectively (Brown et al, 1978). Although changes in overall organ weight were not necessarily considered adverse, teratogenic effects were considered ecologically relevant. Therefore, the Field et al. (1993) NOAEL and LOAEL values were selected for TRV development. No UFs were applied, and therefore, the mammalian low TRV used was 1910 mg/kg bw-day and the mammalian high TRV used was 3210 mg/kg bw-day for diethylphthalate.

For birds, no published TRVs were available for diethylphthalate in literature. Empirical data for this compound were also unavailable. Due to lack of data, di-n-butylphthalate was selected as a surrogate for diethylphthalate due to the similarity in chemical structure and activity of the two chemicals. Published bird low TRV of 0.11 mg/kg bw-day and bird high TRV of 1.11 mg/kg bw-day for di-n-butylphthalate were selected (Sample et al., 1996).

5.3.4 N-nitrosodimethylamine

For mammals, no published TRVs were available for this compound. Empirical studies were located in the ATSDR database (2007) for this compound (ATSDR, 1989a) that were used to derive TRVs for the ERA. A mouse gestation study with exposure to n-nitrosodimethylamine in the drinking water reports a LOAEL of 0.02 mg/kg bw-day for increased perinatal death (Anderson et al., 1978). No NOAEL was reported in this study, therefore a UF of 10 was applied to convert the LOAEL to a NOAEL. The resulting low and high TRVs for mammals are 0.002 mg/kg bw-day and 0.02 mg/kg bw-day, respectively.

For birds, no published TRVs were available from the literature and no empirical data were available to develop avian TRVs for this compound.

5.3.5 N-nitrosodiethylamine

No published TRVs or empirical studies were located for mammals for n-nitrosodiethylamine in any of the sources listed above. Due to close structural similarities, TRVs for n-nitrosodimethylamine were selected as surrogates for the ERA. Therefore, the low mammalian TRV is 0.002 mg/kg bw-day and the high mammalian TRV is 0.02 mg/kg bw-day.

For birds, no published TRVs were available from the literature and no empirical data were available to develop avian TRVs for this compound.

5.3.6 N-nitrosodipropylamine

Published TRVs for this compound were unavailable for mammals, however, empirical studies (ATSDR, 1989d) were available from the ATSDR database (2007). Lijinsky and Taylor (1978, 1979) report a LOAEL of 5.1 mg/kg bw-day for decreased longevity in rats exposed to n-nitrosodipropylamine in drinking water. The exposures were conducted for 30 weeks. However, the endpoints in these studies were carcinogenic and therefore, not considered to be appropriate for the ERA.

Due to close structural similarities, TRVs for n-nitrosodimethylamine were selected as surrogates for n-nitrosodipropylamine. Therefore, the low mammalian TRV is 0.002 mg/kg bw-d and the high mammalian TRV is 0.02 mg/kg bw-day.

For birds, no published TRVs were available from the literature and no empirical data were available to develop avian TRVs for this compound.

5.3.7 N-nitrosomethylethylamine

No published TRVs or empirical studies were located for mammals for n-nitrosomethylethylamine in any of the sources listed above. Due to close structural similarities, TRVs for n-nitrosodimethylamine were selected as surrogates for the ERA. Therefore, the low mammalian TRV is 0.002 mg/kg bw-day and the high mammalian TRV is 0.02 mg/kg bw-day.

For birds, no published TRVs were available from the literature and no empirical data were available to develop avian TRVs for this compound.

5.4 Pesticides

For mammals and birds, published TRVs were available for all of the pesticides from the sources listed above and as presented in Table U.A2-4. TRVs for DDD, DDE, DDT, and total DDT were based on EcoSSLs (USEPA, 2007c) TRVs for heptachlor, aldrin, BHC (benzene hexachloride; hexachlorocyclohexane) methoxychlor (mammals only) were based on BTAG TRVs (CalEPA, 2002b). TRVs for chlordane, dieldrin, endosulfan I, endosulfan II, endosulfan sulfate, endrin, kepone (for mammals only), and BHC (birds only) were from Sample et al., (1996). There were no published mammalian or avian TRVs for hexachlorobenzene, heptachlor epoxide, and mirex and no published avian TRVs for aldrin, kepone and methoxychlor. TRVs were developed for these CPECs as described below.

The agencies suggested that the USACHPPM (2006) NOAEL and LOAEL TRVs for chlordane (2.1 and 21 mg/kg-d, respectively) be incorporated, as the proposed ORNL chlordane NOAEL TRV (4.6 mg/kg-d) for mammals exceeds the USACHPPM NOAEL TRV. However, the ORNL TRVs were based on a long term study and include a bounded NOAEL/LOAEL pair. The ORNL recommended study was considered more appropriate and was used in this ERA than the study cited in the USACHPPM because that study was over a relatively short duration and the endpoints were not bounded.

5.4.1 Aldrin

For mammals, BTAG TRVs were selected for the ERA. The low TRV of 0.1 mg/kg and the high TRV of 1 mg/kg bw-d are based on neurobehavioral changes in rat.

For birds, no published TRVs were available from the literature sources listed above. Empirical studies from the ECOTOX database (USEPA, 2007a) were obtained and a study by Hill et al. (1975) was selected to develop TRVs for the ERA according to the guidelines described above. In this study, birds were exposed to aldrin via the diet, and an 8-day Japanese quail LC50 of 34 mg/kg was reported. The dietary concentration was converted to a dose by multiplying by the food intake rate as a percentage of body weight. Since no food intake rate data is available specifically for Japanese quail, a value of 0.078 for northern bobwhite was assumed. The acute LC50 of 2.7 mg/kg bw-d was converted to a LOAEL by applying a UF of 10, and a UF of 100 was applied to convert the LC50 to a NOAEL. The resulting low mammalian TRV for aldrin is 0.027 mg/kg bw-d and the high TRV is 0.27 mg/kg bw-d.

As requested by the agencies, the study by Hall et al. (1971) reported in USACHPPM (2006) was reviewed. The endpoint selected as the basis of the aldrin TRV reported by USACHPPM (2005) for avian species was a subchronic no-effect of decreased growth seen in pheasants fed encapsulated aldrin (Hall et al. 1971). Using the guideline described above, the subchronic NOAEL of 0.07 mg/kg bw-d can be converted to a chronic NOAEL of 0.035 mg/kg bw-d using a UF of 2. A chronic LOAEL of 0.35 mg/kg bw-d can be extrapolated from the estimated chronic NOAEL by using a UF of 10. Although the study reported by USACHPPM is considered appropriate, the resulting TRVs are less conservative than the study discussed above. In addition, USACHPPM (2005) reports a low confidence to this avian. Therefore, the avian TRV for aldrin was not based on study reported in USACHPPM (2005), but on the study by Hill et al. (1975) discussed above.

5.4.2 Hexachlorobenzene

For mammals, published TRVs are available for hexachlorobenzene (USEPA, 1999b). However, the TRVs from this source were derived from chronic rat toxicity data (Grant et al., 1977). More conservative data were available from the ATSDR and IRIS databases (ATSDR, 2007; USEPA, 2007b), however. The additional data include a two-generation rat study that reports a LOAEL for liver fibrosis of 0.016 mg/kg bw-day and a LOAEL for reduced pup survival and chronic nephrosis of 1.5 mg/kg bw-day (Arnold et al, 1985). Although regulatory guidance recommends higher TRVs, to maintain the conservative nature of the risk assessment, the Arnold et al. (1985) study was selected for TRV derivation. Although liver fibrosis may not be considered adverse on a population basis, reduced pup survival is considered an appropriate adverse endpoint. Therefore the LOAEL of 1.5 mg/kg bw-day was selected as the mammalian high TRV for hexachlorobenzene. A UF of 10 was applied to extrapolate from the chronic LOAEL to a chronic NOAEL, resulting in a mammalian low TRV of 0.15 mg/kg bw-day for hexachlorobenzene.

For birds, published low and high TRVs of 0.225 and 22.5 mg/kg bw-day, respectively, were available from USEPA (1999a). These values were derived from acute toxicity data for quail (Hill and Camardese, 1986).

5.4.3 Heptachlor epoxide

Published mammalian or avian TRVs for heptachlor epoxide were unavailable from the literature. Since heptachlor is rapidly metabolized to the more toxic epoxide in animals, toxicity values for heptachlor are derived in part from the toxicity of heptachlor epoxide. Therefore, published TRVs for heptachlor were used as surrogates for this compound. For mammals, BTAG values of 0.13 and 6.8 mg/kg bw-day for heptachlor were selected as low and high mammalian TRVs respectively. For birds, a LOAEL-based high TRV of 6.5 mg/kg bw-day is available from USEPA (1999a). A UF of 10 was applied to calculate a low TRV of 0.65 mg/kg bw-day for birds.

5.4.4 Kepone

For mammals, as mentioned above, the low TRV of 0.08 mg/kg bw-day and high TRV of 0.4 mg/kg bw-day were obtained from Sample et al. (1996) based on exposure to rat.

For birds, no published TRV values were available for kepone from literature. However, empirical data were available in the ECOTOX database (USEPA, 2007a) and were used to develop avian TRVs following the guidelines outlined in Section 2. Eroschenko (1979) reports an 8-month LOAEL of 80 mg/kg in the feed for changes in organ weight. This endpoint was not considered adverse, therefore a chronic LOAEL for eggshell cracking in Japanese quail was used (Eroschenko and Place, 1977). This 8-month study resulted in a LOAEL of 200 mg/kg in the feed. This value was converted to a dose of 15.6 mg/kg bw-day by multiplying the food concentration by the daily food intake rate as a percentage of body weight (USEPA, 1993). Since no intake rate or mean body weight were available specifically for Japanese quail, the intake rate of 7.8% for northern bobwhite was used instead. No UFs were applied to this chronic LOAEL, and therefore, the avian high TRV was based on the LOAEL of 15.6 mg/kg bw-day. A UF of 10 was applied to extrapolate from the chronic LOAEL to a chronic NOAEL, resulting in an avian low TRV of 1.56 mg/kg bw-day for kepone.

5.4.5 Methoxychlor

For mammals, as mentioned above, the low TRV of 2.5 mg/kg bw-day and high TRV of 50 mg/kg bw-day were obtained from CalEPA (2000). These BTAG values are rat developmental studies.

For birds, no published TRV values were available for methoxychlor from literature. However, empirical data were available in the ECOTOX database (USEPA, 2007a) and were used to develop avian TRVs following the guidelines outlined in Section 2. Cecil et al. (1974) report a 2-mo LOAEL for changes in organ weight in Japanese quail. This endpoint was not considered adverse, therefore an 8-d LC50 for mallard was used to derive low and high TRVs for the ERA. USEPA (2000) reports an 8-d LC50 of greater than 5620 mg/kg. An LC50 of 5620 mg/kg was conservatively assumed, and this value was converted to a dose of 320 mg/kg bw-day by multiplying the food concentration by the daily food intake rate of 5.7% as a percentage of body weight of approximately 1 kilogram (kg) (USEPA, 1993). A UF of 10 was applied to extrapolate from the acute lethal dose to a chronic LOAEL, resulting in an avian high TRV of 32 mg/kg bw-day. A UF of 100 was applied to extrapolate from the acute lethal dose to a chronic NOAEL, resulting in an avian low TRV of 3.2 mg/kg bw-day for methoxychlor.

5.4.6 Mirex

For mammals, published TRVs for mirex were unavailable. However, empirical studies were located in the ATSDR Toxicological Profile for Mirex and Chlordecone (1995). Chu et al. (1981a) reported histological changes in the liver and thyroid at concentrations of 0.07 mg/kg bw-day in rats exposed to mirex in the food for 21 months. In an associated reproductive study, Chu et al. (1981b) found increased incidence of cataracts in rat pups at 0.25 mg/kg bw-day. However, this endpoint was not considered ecologically relevant to derive TRVs for the ERA. The first study by Chu et al., (1981a) was considered more appropriate and was selected to develop TRVs for the ERA. A NOAEL was not reported by Chu et al (1981b); therefore, a UF of 10 was applied to convert the LOAEL of 0.07 mg/kg bw-day to a NOAEL of 0.007 mg/kg bw-day. The resulting low TRV for mirex is 0.007 mg/kg bw-day and the high TRV for this compound is 0.07 mg/kg bw-day.

Published TRVs for mirex were also unavailable for birds. However empirical data was available from the ECOTOX database (USEPA, 2007a) for mirex. A study by Stickel et al. (1973) was selected for TRV development according to the guidelines described above. In this study juvenile wild common grackle were exposed to mirex in the diet for 38 days, resulting in an LC50 of 250 mg/kg. The dietary mirex concentration was converted to a dose by multiplying by the food intake rate as a percentage of body weight. Since data specific to grackle is unavailable, the food intake rate of 7.8% for northern bobwhite was assumed. A UF of 10 was applied to the resulting LC50 of 19.5 mg/kg bw-day, yielding a high TRV of 2.0 19.5 mg/kg bw-day for mirex. A UF of 100 was applied to convert the LC50 to a NOAEL yielding a low TRV 0.2 mg/kg bw-day for mirex.

5.5 Polychlorinated Biphenyls

For mammals and birds, TRVs for total PCB congeners and total PCB TEQ (based on dioxin/furan TEQ TRVs) were obtained from sources listed above and as presented in Table U.A2-4. TRVs for Aroclor 1260 were based on surrogates as described below.

5.5.1 Aroclor 1260

For mammals, USEPA Region 9 BTAG (CalEPA, 2002b) recommends a low TRV of 0.36 mg/kg bw-day and a high TRV of 1.28 mg/kg bw-day for total PCBs. These values are based on mouse reproductive studies with Aroclor 1254 (Simmons and McKee, 1992; Linzey, 1987). Because Aroclor 1260 is a mixture of PCB congeners, the BTAG TRVs for total PCBs were selected. No UFs were applied, and therefore, the mammalian low TRV used was 0.36 mg/kg bw-day and the mammalian high TRV used was 1.28 mg/kg bw-day for Aroclor 1260.

For birds, USEPA Region 9 BTAG (CalEPA, 2002b) recommends a low TRV of 0.09 mg/kg bw-day and a high TRV of 1.27 mg/kg bw-day for total PCBs. These values are based on chicken reproductive studies (Platonow and Reinhart, 1973; Britton and Huston, 1973). Because Aroclor 1260 is a mixture of PCB congeners, the BTAG TRVs for total PCBs were selected. No UFs were applied, and therefore, the avian low TRV of 0.09 mg/kg bw-day and the avian high TRV of 1.27 mg/kg bw-day was used for Aroclor 1260.

5.6 Herbicides

For mammals and birds, published TRVs were not available in literature for any of the herbicides. However, empirical data were available and TRVs were developed for mammals and birds as described below.

5.6.1 2,4-Dichlorophenoxyacetic Acid (2,4-D)

For mammals, empirical data were available from the IRIS database (USEPA, 2007b). The human RfD was derived from a 90-d rat study showing effects on blood chemistry, and liver and renal enzyme levels and organ weight at 5 mg/kg bw-day (Dow Chemical Company, 1983). These endpoints were not considered adverse however. Additional chronic and reproductive studies reported no adverse effects at higher doses: no effects were reported in dogs at dietary levels up to 500 ppm (approximately 14.5 mg/kg bw/day), up to 1250 ppm in rats (approximately 62.5 mg/kg bw/day) (Hansen et al., 1971), or at levels of 1000 ppm in drinking water (50-100 mg/kg bw/day) in pregnant rats or their offspring (Bjorklund and Erne, 1966). A secondary reference to a personal communication reported an increase in mortality among rats whose dams received approximately 50 mg/kg bw/day of 2,4-D in the diet for 3 months before mating and throughout gestation and lactation (Gaines and Kimbrough, 1970). Due to lack of empirical data and evidence of adverse effects in mammals, 2,4-DB was selected as a surrogate to derive TRVs for the ERA. Therefore, the low and high TRVs for 2,4-DB of 8 mg/kg bw-day and 25 mg/kg bw-day were also selected for 2,4-D.

For birds, no TRV was required for the ERA as this compound is a CPEC only in deeper soils (0-10 feet bgs) and birds are not exposed to soils at these depths.

5.6.2 2,4-Dichlorophenoxybutyric Acid (2,4-DB)

For mammals, no published TRVs were available for 2,4-DB from literature. However, limited empirical studies were available; a small number of mammalian toxicological studies with the chlorophenoxy herbicide, 2,4-DB, were available from the IRIS database (USEPA, 2007b). A 90-day study (Rhodia, Inc., 1969) with dogs was selected for developing the human RfD; the same study was used in this ERA to develop mammalian TRVs. Dogs exposed to 2,4-DB at 25 mg/kg bw-day experienced severe internal hemorrhaging and death. The NOAEL for this endpoint was 8 mg/kg bw-day. No UFs were applied, resulting in a mammalian low TRV of 8 mg/kg bw-day and a mammalian high TRV of 25 mg/kg bw-day for 2,4-DB.

For birds, published TRVs for 2,4-DB were unavailable. However, empirical data for this herbicide were available in the ECOTOX database (USEPA, 2007a), and the study reporting adverse reproduction, growth, or mortality effects at the lowest concentration was selected for developing avian TRVs. Based on the guidelines described above, a USEPA study (2000a) from the Office of Pesticide Programs was selected where a 21-day northern bobwhite LC50 of 1536 mg/kg was reported with exposure to 2,4-DB via food. The acute LC50 was converted to a daily dose of 120 mg/kg bw-day by multiplying the food concentration by the daily food intake rate of 7.8% for northern bobwhite as a percentage of body weight (USEPA, 1993). A UF of 10 was applied to extrapolate from the acute lethal dose to a chronic LOAEL, resulting in an avian high TRV of 12 mg/kg bw-day. A UF of 100 was applied to extrapolate from the acute lethal dose to a chronic NOAEL, resulting in an avian low TRV of 1.2 mg/kg bw-day for 2,4-DB.

5.6.3 2,4,5-Trichlorophenoxyacetic Acid (2,4,5-T)

For mammals, no published TRVs were available in the literature. However, toxicity data were available from the IRIS database (USEPA, 2007b). A three-generation study with rats exposed to 2,4,5-T in the diet showed reduced neo-natal survival at 10 mg/kg bw-day (Smith et al., 1981). The same LOAEL was demonstrated for other sublethal effects in rats, and reproductive effects were well documented in mice, hamsters, and monkeys at higher concentrations. To extrapolate a NOAEL for the low TRV, a UF of 10 was applied to the LOAEL of 10 mg/kg bw-day. The resulting low and high TRVs for the ERA are 1.0 mg/kg bw-day and 10 mg/kg bw-day for 2,4,5-T.

For birds, no TRV was required for the ERA as this compound is a CPEC only in deeper soils (0-10 feet bgs) and birds are not exposed to soils at these depths.

5.6.4 2,4,5-Trichlorophenoxypropanoic Acid (Silvex)

For mammals, no published TRVs were available for 2,4,5-Trichlorophenoxypropanoic Acid (2,4,5-TP or Silvex) from literature, however limited data were available from the IRIS database (USEPA, 2007b). A study with dogs exposed to silvex in the diet for two years reported histopathological changes in the liver at concentrations of 2.5 mg/kg bw-day (Gehring and Betso, 1978). This study was selected for developing the human RfD, but the authors note that dogs may be especially sensitive to silvex because of their relatively poor capacity for renal excretion of organic acids. Additionally, the changes were not considered necessarily adverse. In a NAS study (1977), reduced pup weights and incomplete skull ossification occurred in rats at dosages of 50 mg/kg bw-day, with a NOAEL of 25 mg/kg bw-day for these fetotoxic and teratogenic effects. The NAS study (1977) was selected for TRV development and no UFs were applied. Therefore the low mammalian TRV for silvex is 25 mg/kg bw-d and the high mammalian TRV is 50 mg/kg bw-d.

For birds, no published TRVs were available from the literature. The lowest avian toxicity value for reproduction, growth or mortality effects was selected from the ECOTOX database (USEPA, 2007a) based on the guidelines described above. Hill et al. (1975) report an 8-d LC50 of 3031 mg/kg for northern bobwhite. The acute LC50 was converted to a daily dose of 236 mg/kg bw-day by multiplying the food concentration by the daily food intake rate of 7.8% for northern bobwhite as a percentage of body weight (USEPA, 1993). A UF of 10 was applied to extrapolate from the acute lethal dose to a chronic LOAEL, resulting in an avian high TRV of 23.6 mg/kg bw-day. A UF of 100 was applied to extrapolate from the acute lethal dose to a chronic NOAEL, resulting in an avian low TRV of 2.36 mg/kg bw-day for silvex.

5.6.5 2-sec-Butyl-4,6-dinitrophenol (Dinoseb)

No published TRVs were available for mammals from the literature. Empirical data were found in the ATSDR and IRIS (USEPA, 2007b) databases however. Dow Chemical (1981) conducted a three-year rat study with exposure to dinoseb through the diet. Reduced fetal weights were reported at doses of 1mg/kg bw-day; a NOAEL was not reported. This study was selected for TRV development for the ERA. A UF of 10 was applied to convert the LOAEL to a NOAEL resulting in a low mammalian TRV of 0.1 mg/kg bw-d and a high mammalian TRV of 1 mg/kg bw-day for dinoseb.

For birds, no published TRVs were available. Limited empirical studies for this herbicide were available in the ECOTOX database (USEPA, 2007a). The study reporting adverse reproduction, growth, or mortality effects at the lowest concentration was selected for developing avian TRVs. An 8-day Japanese quail LC50 of 409 mg/kg was reported by Hill et al. (1975) with exposure through the diet. This acute LC50 was converted to a dose of 31.9 mg/kg bw-d by multiplying by the daily food intake rate. (Since no intake rate or mean body weight were available specifically for Japanese quail, the intake rate of 7.8% for northern bobwhite was assumed). A UF of 10 was applied to convert the acute LC50 to a LOAEL, resulting in a high TRV of 3.2 mg/kg bw-d. A UF of 100 was applied to convert the LC50 to a NOAEL. Therefore, the avian low and high TRVs for dinoseb are 3.2 mg/k and 0.32 mg/kg bw-d, respectively.

5.6.6 Dalapon

For mammals, no published TRVs for the herbicide dalapon were available from literature. However, limited empirical toxicity data for mammals were found in the IRIS database (USEPA, 2007b). Paynter et al. (1960) report a NOAEL 8.5 mg/kg bw-day and a LOAEL of 28.2 mg/kg bw-day for changes in rat kidney weight after a 2-year dietary exposure to dalapon. The Paynter study (1960) was conducted under rigorous statistical design, over a significant proportion of a rat lifespan, and reported a highly sensitive endpoint. Alterations in kidney weight are often precursors to more severe kidney damage, and this endpoint was considered a conservative marker protective of organ-level failure. For these reasons, the Paynter study (1960) was used to develop mammalian TRVs for this ERA. No UFs were applied, and therefore, the mammalian low TRV was based on the NOAEL of 8.5 mg/kg bw-day, and the mammalian high TRV was based on the LOAEL of 28.2 mg/kg bw-day for dalapon.

For birds, published TRVs for dalapon were unavailable. However, limited empirical data for this herbicide were available in the HSDB database (HSDB, 2007). Two sources of toxicity data were reported: a chicken LD50 of 5660 mg/kg (Tomlin, 2002) and 5-day mortality NOAELs for

pheasant, Japanese quail, and mallard duck all greater than 5000 mg/kg (USFWS, 1975). Because the USFWS study was conducted using mallard, a Site receptor, and pheasant, which is the same order (Passeriformes) as the model Site receptor (western meadowlark) and other bird species noted to utilize the Site, it was selected to develop avian TRVs for this ERA. The reported NOAEL of greater than 5000 mg/kg was converted to a daily dose of 285 mg/kg bw-day by conservatively assuming a NOAEL of 5000 mg/kg and multiplying the food concentration by the daily food intake rate (5.7%) as a percentage of body weight, based on allometric equations for food intake and mean body weight for mallard ducks (USEPA, 1993). No UFs were applied, resulting in an avian low TRV of 285 mg/kg bw-day. Because a LOAEL for the USFWS study was not reported, no high TRV was derived for dalapon.

5.6.7 Dichlorprop

For mammals, no published TRVs were available for dichlorprop (2,4-DP) from literature. Additionally, no empirical toxicity data for mammals were found. Due to the lack of toxicity data for 2,4-DP and the structural similarity with the herbicide, 2,4-DB, TRVs developed for 2,4-DB were used as surrogates 2,4-DP. Also, both 2,4-DB and 2,4-DP have similar biological activity as chlorophenoxy herbicides. Therefore, the mammalian low TRV used was 8 mg/kg bw-day, and the mammalian high TRV used was 25 mg/kg bw-day for 2,4-DP.

Similarly, for birds, published TRVs and empirical data were not available for 2,4-DP, and therefore, the avian TRVs developed for 2,4-DB were used for this ERA. Therefore, the avian low TRV of 1.2 mg/kg bw-day and the avian high TRV of 12 mg/kg bw-day were used for 2,4-DP.

5.6.8 2-Methyl-4-Chlorophenoxyacetic Acid

For mammals, no published TRVs were available for the herbicide 2-methyl-4-chlorophenoxyacetic acid (MCPA) from literature. However, empirical toxicity data for mammals were available in the IRIS database (USEPA, 2007b). A two-generation rat study conducted by the Industry Task Force on MCPA Research (1986a) reports small decreases in pup weight and weight gain at 22.5 mg/kg bw-day. However, a 1-year study by the same Task Force (1986b) found beagle dogs to be more sensitive than rats. Histopathological changes in the kidney and liver of beagle dogs and changes in clinical chemistry at 0.75 mg/kg-day were reported, with no adverse effects at 0.15 mg/kg bw-day, and progressively serious effects with a LOAEL for mortality of 48 mg/kg bw-day. The beagle study was selected to develop TRVs for this ERA to protect Site species that may be more sensitive to MCPA than rats. No UFs were applied, and therefore, the low mammalian TRV was based on the NOAEL of 0.15 mg/kg bw-day, and the mammalian high TRV was based on the LOAEL of 0.75 mg/kg bw-day for MCPA.

For birds, published TRVs were not available for MCPA from literature. However, empirical data for this herbicide were available in the ECOTOX database (USEPA, 2007a). The study reporting adverse reproduction, growth, or mortality effects at the lowest concentration was selected for developing avian TRVs for this ERA. Based on the guidelines described in Section 2, a USEPA study from the Office of Pesticide Programs was used (USEPA, 2000a). Acute (LC50) 8-day studies for northern bobwhite, mallard, and pheasant were all reported greater than 2000 mg/kg in the feed. Because mallards have a lower daily food intake rate (5.7%) than bobwhite (7.8%; USEPA, 1993), and mallards were selected as Site receptors, values for mallard were used to convert the LC50 to a dose of 114 mg/kg-day by multiplying the food

concentration by the daily food intake rate as a percentage of body weight (USEPA, 1993). A UF of 10 was applied to extrapolate from the acute lethal dose to a chronic LOAEL resulting in an avian high TRV of 11.4 mg/kg bw-day. A UF of 100 was applied to extrapolate from the acute lethal dose to a chronic NOAEL, resulting in an avian low TRV of 1.14 mg/kg bw-day for MCPA.

5.6.9 2-(2-Methyl-4-Chlorophenoxy)Propionic Acid

For mammals, no published TRVs were available for the herbicide 2-(2-methyl-4-chlorophenoxy)propionic acid (MCPA) from literature. However, empirical toxicity data for rats and mice were available in the IRIS (USEPA, 2007b), ECOTOX (USEPA, 2007a), and HSDB (2007) databases. Based on the guidelines described in Section 2, the lowest LOAEL was reported by the BASF Group (BASF, 1985) in a chronic 90-day rat exposure. In this study (BASF, 1985), the LOAEL reported for increased kidney weight was 9 mg/kg bw-day, and the NOAEL reported was 3 mg/kg bw-day. Due to the sensitive, sub-lethal endpoint and the chronic duration of the study, no UFs were applied. Therefore, the mammalian low TRV was based on the NOAEL of 3 mg/kg bw-day, and the mammalian high TRV was based on the LOAEL of 9 mg/kg bw-day for MCPA.

For birds, published TRVs were not available for MCPA from literature. Empirical data for this herbicide were also not available. However, due to the lack of toxicity data and the structural and toxicological similarity with the herbicide MCPA, avian TRVs developed for MCPA were used for MCPA. Therefore, the avian low TRV of 1.14 mg/kg bw-day and the avian high TRV of 11.4 mg/kg bw-day were used for MCPA.

5.7 Dioxin/Furan Toxicity Equivalent

For wildlife, potential risks from individual dioxin/furan congeners were not evaluated; instead dioxin/furan TEQ were evaluated. For mammals, TRVs for dioxin/furan TEQ were based on the most toxic of the dioxin and furan congener, 2,3,7,8-TCDD for the rat from Sample et al. (1996). The mammalian low TRV used was 0.000001 mg/kg-day, and the mammalian high TRV used was 0.00001 mg/kg-day as presented in Table U.A2-4.

Similarly, avian TRVs for 2,3,7,8-TCDD from Sample et al. (1996) were used for this ERA. The avian low TRV used was 0.000014 mg/kg-day, and the avian high TRV used was 0.00014 mg/kg-day.

5.8 Polycyclic Aromatic Hydrocarbons

In general, PAHs have been thoroughly evaluated for toxicity by the USEPA and others (USEPA, 2006a; 1997; CalEPA, 2000; Sample et al., 1996). As discussed earlier, due to lack of data for all PAHs and the overall similarity in terms of chemical reactivity, environmental fate, and toxicological effects for LMW PAHs, TRVs developed for a LMW PAH such as naphthalene were used as a surrogate for all LMW PAHs. Similarly, TRVs developed for a HMW PAH such as benzo(a)pyrene or 7,12-dimethylbenz(a)anthracene were used as surrogates for all HMW PAHs.

For mammals, the BTAG TRVs for naphthalene and benzo(a)pyrene from CalEPA (2002b) were used for this ERA. The mammalian low TRV of 50 mg/kg bw-day and the mammalian high TRV

of 150 mg/kg bw-day for naphthalene were used for all LMW PAHs and total LMW PAHs. Similarly, the mammalian low TRV of 1.31 mg/kg bw-day and the mammalian high TRV of 32.8 mg/kg bw-day for benzo(a)pyrene were used for all HMW PAHs and total HMW PAHs.

For birds, published TRVs were available in USEPA Region 6 guidance (USEPA, 1999b). However, the study (Brunstrom et al., 1991) was based on egg injection tests that are not considered appropriate for developing TRVs (USEPA, 2006a). Several studies were reviewed, and the most appropriate study was selected to develop avian TRVs for this ERA.

For LMW PAHs, Patton and Dieter's study (1980) evaluating the effect of PAH mixtures on hepatic function in mallard duck livers using a mixture of paraffins and aromatic hydrocarbons was selected. There were visible signs of toxicity, indicated by significant increase in liver weight for the group administered 4,000 mg/kg PAH mixture, but livers appeared normal in texture and color. No effects were observed for the 400 mg/kg treatment group. Therefore, 400 mg/kg was converted to a NOAEL assuming an average body weight of approximately 1 kg (USEPA, 1993) for the mallard ducks and estimating an ingestion rate of 0.059 kilograms per day (kg/day; calculated from allometric equation in USEPA, 1993 for food ingestion rate in dry weight) resulting in a avian low TRV of 22.8 mg/kg bw-day. Similarly, the 4,000 mg/kg was converted to a LOAEL resulting in an avian high TRV of 228 mg/kg bw-day for individual LMW PAHs and total LMW PAHs.

For HMW PAHs, a study by Trust et al. (1994) reporting a NOAEL of 10 mg/kg bw-day and a LOAEL of 100 mg/kg bw-day for overt signs of toxicity, such as decreased body mass in European starlings exposed to 7,12-dimethylbenz(a)anthracene, was selected to develop TRVs for this ERA. Immunosuppression was observed at higher doses. The exposures were via oral gavage, and the study was conducted on nestlings, a sensitive life-stage. No UFs were applied, and therefore, an avian low TRV of 10 mg/kg bw-day and an avian high TRV of 100 mg/kg bw-day were used for individual HMW PAHs and total HMW PAHs.

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