8.0 STAGE II ERC – ANALYSIS - EXPOSURE ASSESSMENT

8.1 Purpose

The purpose of the exposure assessment phase is to evaluate exposure of receptors to chemical stressors. This phase involves collection and integration of information on toxicity of the chemical(s) of potential ecological concern (COPECs), COPEC concentrations and spatial distribution, and exposure conditions (temporal and spatial patterns). Exposure point concentrations of COPECs will be determined and later (in the section on Effects Assessment) compared to toxicity reference values in order to calculate the potential for adverse effects.

8.2 Exposure Point Concentrations

The MCP (310 CMR 40.0926) provides guidance for the determination of exposure point concentrations and states that the use of upper percentiles or maximum concentrations is appropriate when conducting a screening-level assessment and/or when insufficient data are available to adequately characterize the site. However, in a Stage II ERA, the MCP states that the use of an arithmetic mean is appropriate (MCP Chapter 9 Guidance, section 9.5.3.3). For environmental concentration data that is lognormally distributed (as is the case with most environmental data), a geometric mean may be a more appropriate statistic to represent the central tendency of the data and is less sensitive to a single elevated value in a dataset than the arithmetic mean. Furthermore, the sampling density at this site for wetland sediments and soils is sufficiently great that it is highly unlikely that the site mean has been underestimated. Said another way, it is highly unlikely that additional sampling would result in a greater exposure point concentration. Thus, both the arithmetic and geometric means are presented as exposure point concentrations to provide a range of exposures. To provide a conservative estimate of chemical concentrations in plant tissue and surface water, arithmetic mean and maximum concentrations are presented to provide a range of exposures since the spatial coverage of these sample types are the most limited.

8.2.1 Exposure Point Concentrations For Surface Water Exposures

Aquatic receptors (fish and aquatic invertebrates) are potentially exposed to COPECs in surface water and sediments via direct contact. However, the exposure from direct contact from wetland sediments and soils at this site to these aquatic receptors is difficult to quantify and is expected to be minimal since the site is inundated for only a few months and the presence of emergent vegetation would be expected to minimize direct contact. Unlike lake and river bottoms where there is permanent flooding and substantial exposure via sediments, the situation is considerably different in a temporarily flooded wetland. Approaches such as Effects Range-Low (ERLs) and Effects Range Median (ERMs) and similarly derived benchmark levels for sediment exposures are usually limited to screening level assessments and are thus, not applicable in a Stage II assessment. For more refined equilibrium partitioning approaches, which are used to assess exposure and effects from sediment exposure pathways, MCP guidance states that such benchmarks "are derived using sediment/water partioning coefficients … [and] are only appropriate for wetland sediments that are permanently flooded" (MCP Chapter 9, p. 9-94). Thus, any estimate of potential exposure of aquatic organisms via sediments at this site would be greatly uncertain due to the temporary nature of the expoure duration. Furthermore, this route of expsoure is likely to be minimal when compared to exposure through surface water exposure pathways.

Exposure point concentrations of COPECs in surface water are presented for the overall site for two seasons since the concentrations are dramatically different depending on site-specific seasonal conditions

(Table 8-1). In addition, exposure point concentrations for surface water were presented earlier for low flow and inundated conditions (Tables 4-5 and 4-7).

Table 8-1. Exposure point concentrations of COPECs outside the "Area of Readily Apparent Harm" that exceeded water quality criteria or screening benchmarks.

	Exposure Point Concentration (µg/L)			
	Low water		Inundation	
Chemical	Mean	Maximum	Mean	Maximum
Aluminum	45.5	66.0	17.1	22.9
Cadmium	1.16	1.60	0.079	0.093
Copper	30.0	39.0	3.85	4.50
Iron	475	580	168	170
Zinc	190	210	18.0	23.4

8.3 Exposure Characteristics of Wetland Plants

Exposure pathways to plants include uptake of COPECs from deposition onto plant surfaces and from soil and water by the roots and, for some COPECs, translocation throughout the plant to the stems, leaves, seeds, and fruiting bodies. The amount of COPEC that is actually biologically available depends on a number of COPEC-specific, plant-specific, and site-specific parameters, including the speciation and complexation of the COPEC (for metals), species-specific differences in plant uptake and retention of COPECs, and several soil characteristics such as cation exchange capacity, organic carbon content, pH, grain size, and percent sand, silt, and clay. The primary exposure pathway for plants at this site is assumed to be from soil. It is recognized that uptake from water, particularly from saturated soil, is also a potential exposure pathway to plants. However, the exposures from these pathways are assumed to be proportional to the concentrations of COPECs in soil, since there is likely an equilibrium between soil and the water in the soil spaces.

One of the measurement endpoints for wetland plants includes an evaluation of the concentrations of COPECs in the wetland soil that are located within the area of stunted vegetation. These concentrations will then be compared to concentrations of COPECs in locations outside of the "Area of Readily Apparent Harm". To determine an exposure point concentration that is associated with the field-measured area of stunted vegetation, a statistical analysis of these samples was conducted for all COPECs for which the literature-based phytotoxicity benchmark levels were exceeded. In this way, the phytotoxicity benchmarks acted as a sort of screen for which chemicals are potentially responsible for the stunted growth of the wetland vegetation.

8.4 Exposure Characteristics of Avian and Mammalian Wildlife Receptors

Characteristics of key receptors are presented in Tables 8-2 and 8-3, including exposure assumptions for body weight, ingestion rate, dietary composition, area use factor, *etc.* The primary source of exposure assumptions is the USEPA Exposure Factors Handbook (USEPA 1993). Additional sources of information include primary peer-reviewed scientific literature, site surveys (Appendix A), and professional judgment, and other compendia of region-specific (DeGraaf and Rudis, 1986) and species-specific information (Sample and Suter, 1994). Whenever available, region-specific exposure information was utilized. The selected species might be exposed to COPECs through contact with and/or ingestion of contaminated media (*e.g.*, primarily through dietary exposure) and through contaminants accumulated and retained in tissues of the receptor itself (*e.g.*, internal body burdens, transfer to eggs, *etc.*). Exposure estimates for all species were calculated for COPECs detected in dietary items, incidental soil ingestion, and water ingestion. In cases where minor or uncommon dietary items were reported, these were incorporated into categories for which data are available or are readily modeled. For example, almost 15% of the diet of red-tailed hawks can be composed of reptiles. Since COPECs were not measured in reptiles and since there are no adequate models for uptake into reptiles, fraction of the diet for reptiles was added to the dietary fraction consisting of mammals.

8.5 Exposure Point Concentrations for Avian and Mammalian Wildlife Receptors

Exposure calculations were conducted with some exposure concentrations derived from either measured concentrations of chemical stressors or concentrations predicted from models when no measured concentrations were available. Bioaccumulation models are often fraught with uncertainty because bioavailability depends upon highly variable site-specific considerations such as soil type, pH, moisture, clay content, organic carbon, cation exchange capacity, and receptor-specific considerations such as uptake mechanisms. In particular, available data demonstrate very limited assimilation and accumulation of COPECs into wetland vegetation and small mammals. Thus, to minimize uncertainty, concentrations of COPECs were measured in the dietary exposure and water ingestion pathways of herbivorous and carnivorous wildlife utilizing the site. Exposure to COPECs from inhalation and dermal contact were considered negligible for the purposes of this Stage II ERC for wildlife (*e.g.*, muskrats, mallards, red-tailed hawk, *etc.*). Direct ingestion of soil or sediment was considered for certain wildlife since wildlife may experience significant contaminant exposure through direct ingestion of soil or sediment. Exposure point concentrations for incidental soil ingestion and plants are presented in Table 8-4. Exposure point concentrations for year-round surface water ingestion by wildlife are assumed to be same as the concentrations observed during conditions of low flow and are presented in Table 8-5.

Terrestrial small mammals are the primary dietary items of the red-tailed hawk, one of the receptors considered in this ERC. In 1989, USFWS collected small mammal samples at multiple locations along the Sudbury River, including two small mammal samples from the wetland adjacent to the former Raytheon facility. COPEC concentrations in these samples are presented in Table 8-6. The exposure point concentration for small mammals in the diet of red-tailed hawks is the maximum value for each detected COPEC.

Table 8-2. Key characteristics of ecological receptors.

Receptor	Genus/Species	Body Weight BW (kg)	Food Ingestion Rate IR_{feed} (kg/day) [a]	Water Ingestion Rate IR_{water} (L/day)	Sediment Ingestion Rate IR_{sed} (kg/day) [b]	Soil Ingestion Rate IR_{soil} (kg/day) [b]
Meadow Vole	*Microtus pennsylvanicus*	0.044[1]	0.005[1]	0.006[1]	NA	0.00012[1]
White-tailed Deer	*Odocoileus virginianus*	56.7[1]	1.7[1]	3.7[1]	NA	0.0066[1]
Red-tailed Hawk	*Buteo jamaicenisis*	1.126	0.109[c]	0.064	NA	NA
Muskrat	*Ondatra zibethicus*	1.25[1]	0.37[1]	0.0625[1]	0.012[1]	NA
Mallard duck	*Anas platyrhynchos*	1.0[1]	0.100[1]	0.064[2]	0.00031[2]	NA

[a]kg/d wet weight
[b]kg/d dry weight
[c]IR_{food} was estimated using allometric model for all birds, USEPA, 1993
[1]Sample, B.E. and G.W. Suter III, 1994. Estimating exposure of terrestrial wildlife to contaminants. ORNL.
[2]USEPA, 1993. Wildlife exposure factors handbook. EPA/600/R-93/187

Table 8-3. Dietary fraction for ecological receptors.

		Dietary Fraction (% of total diet on a wet weight basis)[1]				
Receptor	**Genus/Species**	**Plants Above Soil[2]**	**Plants Below Soil[3]**	**Small mammals**	**Birds**	**Aquatic Invertebrates**
Meadow Vole	*Microtus pennsylvanicus*	75	25			
White-tailed Deer	*Odocoileus virginianus*	100				
Red-tailed Hawk	*Buteo jamaicenisis*			90	10	
Muskrat	*Ondatra zibethicus*	25	75			
Mallard duck	*Anas platyrhynchos*	60	25			15[4]

[1]Dietary fraction is based on USEPA (1993), DeGraaf and Rudis (1986), and scientific judgment.
[2]Plants above soil include grasses, forbs, leaves, seeds, *etc.*
[3]Plants below soil include emergent roots, shoots, tubers, *etc.*
[4]Concentrations of COPECs in aquatic invertebrates were not measured and were assumed to be similar to concentrations of COPECs in plants below soil (refer to section 8.5.1 for more discussion).

Table 8-4. Exposure point concentrations of COPECs in wetland soil and vegetation outside of the "Area of Readily Apparent Harm".

	Concentration (mg/kg, dry weight for soil and wet weight for vegetation)					
	Wetland Soil[1]		Cattail Root[2] (*Typha latifolia*)		Buttonbush Seeds[3] (*Cephalanthus occidentalis*)	
COPEC	**Mean**	**Geometric mean**	**Mean**	**Maximum**	**Mean**	**Maximum**
Antimony	4.33	3.85	0.02	0.05	0.001	0.003
Arsenic	14.9	9.43	0.26	0.33	0.005	0.008
Cadmium	3.34	2.39	0.29	0.39	0.005	0.009
Chromium (Cr3+)	551	183	1.09	1.34	0.35	0.50
Chromium (Cr6+)	50.9	20.3	NA	NA	NA	NA
Copper	585	243	5.74	11.33	1.77	1.89
Lead	267	199	3.30	5.80	0.032	0.054
Manganese	311	236	59.14	68.33	33.270	46.961
Mercury	1.68	0.97	0.010	0.02	0.001	0.001
Silver	23.7	5.23	0.07	0.11	0.023	0.047
Tin	22.5	12.2	0.80	0.88	0.14	0.15
Vanadium	39.1	33.5	0.47	0.75	0.004	0.005
Zinc	130	105	29.44	41.66	3.22	3.90
Total PAHs	12.0	4.27	NA	NA	NA	NA
Total PCBs	2.92	1.42	0.009	0.014	0.0003	0.0004

[1]Concentrations in soil are from Table 4-13.

[2]Concentrations in cattail roots are from Table 4-17 except the values were corrected to wet weight concentrations using a value of percent moisture of 84.9% (mean of 7 samples collected on-site).

[3]Concentrations in buttonbush seeds are from Table 4-19 except the values were corrected to wet weight concentrations using a value of percent moisture of 10.8% (mean of 7 samples collected on-site).

Table 8-5. Exposure point concentrations of COPECs in surface water outside the "Area of Readily Apparent Harm" for calculating chemical exposure to wildlife via water ingestion.

	Concentration (μg/L)[1]	
COPEC	**Mean**	**Max**
Antimony	2.50	2.50
Arsenic	1.80	2.00
Cadmium	0.93	1.60
Chromium (Cr3+)	5.05	5.40
Chromium (Cr6+)	2.5	2.5
Copper	30.0	39.0
Lead	0.82	1.10
Manganese	770	1000
Mercury	0.10	0.10
Silver	0.25	0.25
Tin	10.0	10.0
Vanadium	1.25	1.25
Zinc	190	210
	NA	NA
	NA	NA

[1]Concentrations in surface water are from Table 4-5.

Table 8-6. Concentrations of COPECs in small mammals collected and analyzed by USFWS[1].

	Concentration (mg/kg, ww)[2]	
COPEC	**RAY1**[3]	**RAY2**[3]
Cadmium	0.0549	0.07905
Chromium	0.854	2.958
Copper	4.514	4.131
Lead	0.2745	0.255
Manganese	4.0	24.6
Mercury	0.0305	0.00765
Zinc	33.855	32.13
Total PCBs	0.005	0.005

[1]Data transcribed from Eaton and Carr, 1991.

[2]These samples were analyzed as whole body samples.

[3]The sample designations are RAY1, which is a meadow jumping mouse (*Zapus hudsonius*), and RAY2, which is a meadow vole (*Microtus pennsylvanicus*).

8.5.1 Estimation of oral exposure for avian and mammalian wildlife receptors

Estimates of daily contaminant exposure experienced by individual receptor species were calculated using a modification of the generalized exposure model presented by Sample and Suter (1994). The generalized exposure model is depicted (Eq. 8-1):

$$ADD_{pot} = \frac{[(IR_{prey} x C_{diet}) + (IR_{soil} x C_{soil}) + (IR_{sed} x C_{sed}) + (IR_{wat} x C_{wat})] x \; SUF}{BW} \qquad \text{Eq. 8-1}$$

Where:

ADD_{pot} = potential average daily dose (*e.g.*, mg/kg-d)

IR_{diet} = Amount of prey or vegetation ingested (kg/d)

C_{diet} = Concentration of chemical in prey or vegetation (mg/kg)

IR_{soil} = Amount of soil ingested (kg/d)

C_{soil} = Concentration of chemical in soil (mg/kg)

IR_{sed} = Amount of sediment ingested (kg/d)

C_{sed} = Concentration of chemical in sediment (mg/kg)

IR_{wat} = Amount of water ingested (kg/d)

C_{wat} = Concentration of chemical in water (mg/kg)

SUF = Site use factor (unitless) (foraging area/site area)

BW = Body weight (kg)

An area use factor was included in the exposure model for some species since some wildlife species were assumed to forage only a portion of the time at the site. For example, based on professional judgment, white-tailed deer would not likely use the site more than 20% of the time because there is not a resident population on the site and their foraging range is much larger than the size of the site. Likewise, red-tailed hawks would not be expected to forage entirely at the site because their foraging range is much larger than the site and they would not be expected to forage at this site during flooded conditions. Therefore an area use factor of 50% was assumed for the red-tailed hawks based on professional judgment.

In addition, a fractional absorption value was included in the exposure model to account for the fraction of the oral dose that is absorbed through the gastrointestinal tract. The fractional absorption values were determined from the scientific literature and were specific to each class of COPECs (*e.g.*, metals, PAHs, and PCBs) and to each exposure medium (*e.g.*, soil and sediment ingestion, normal diet, water ingestion). This fractional absorption factor is especially important for incidental ingestion of sediments and soils. It has been shown that short-term contact of soil with a compound that can be sorbed reduces its bioavailability.

For metals, the percent absorption was assumed to be 100% from water and 50% from vegetation. Support for the use of 50% absorption for metals in the diet comes from EPA reports on the toxicological reviews of metals for the Integrated Risk Information System (IRIS) and other sources in which the oral and gastrointestinal absorption was reported to be 0.4-3% for chromium, 2.5% for cadmium, 10% for silver (Eisler 1996), 20-30% for zinc (Eisler 1993). The percent absorption of metals from sediment and soil was assumed to be 1% for meadow voles (Pascoe et al., 1994a) and 2.5% for muskrats, white-tailed deer, and mallards (Pascoe et al., 1994a). The assumed value of 1% bioavailability from soil for meadow voles is actually 10-fold greater than the measured value by these same authors (Pascoe et al., 1994b) to account for potential uncertainty due to differences in soil parameters for the site described in this ERC. The value of 2.5% reflects additional conservatism to account for potential species-specific differences in bioavailability (absorption) of metals from soils.

Little information is available on the absorption factor or matrix effect for organic chemicals in soils, especially aged chemicals in soils. There is considerable evidence that demonstrates a reduction in bioavailability for persistent organics with increasing time (*e.g.*, aging) in soil (Alexander, 1995). For PAHs (benzo(a)pyrene is used in this ERC as a surrogate for total PAHs for mammals), the percent absorption is assumed to be 100% from water and 25% from soil. Support for the use of 25% absorption from soil is from a study that utilized a digestive tract model to measure the percent mobilization of PAHs from soils (Hack and Selenka 1996). For PCBs, the percent absorption is assumed to be 100% from water, 100% from normal diet, and 85% from sediment and soil (Fries et al. 1989).

For some of the receptors, concentrations of COPECs were not measured in all possible dietary items. While there are models available to estimate the concentrations of COPECs in these dietary items, the assumptions of most of these models are not appropriate to apply to this site. For example, most of the available food web models are designed for systems at equilibrium (usually fully aquatic systems or fully terrestrial systems) in which biota are exposed over their lifetime through exposure pathway(s) that are not expected to differ greatly over the lifetime of the biological organism. Since this wetland site is transitional between aquatic and terrestrial (refer to section 3.5 for more details), it is unlikely that any of the available models can predict the concentrations in dietary items accurately. Therefore, for minor items such as birds in the diet of red-tailed hawk, the relative proportion of dietary items for which data are available (small mammals) was increased. Likewise, for invertebrates in the diet of mallards, it was assumed that the concentrations of COPECs are similar between invertebrates and the roots of cattails since they are both exposed to COPECs in the interstitial water of sediments and soils. In an extensive investigation of biota from a contaminated wetland in Montana, similar concentrations of COPECs were observed for both invertebrates (aquatic and terrestrial) and below ground vegetation such as roots cattails and other emergent vegetation (Pascoe *et al.*, 1996).

The ADD_{pot} for the receptors of concern are presented in the Risk Characterization section. For each receptor, two levels of exposure were calculated as a low ADD_{pot} and a high ADD_{pot} to provide a range of potential exposures. The low ADD_{pot} is based on the following exposure point concentrations of COPECs: arithmetic mean concentrations in surface water and wetland vegetation, geometric mean concentrations in soil and sediment. The high ADD_{pot} is based on the following exposure point concentrations of COPECs: maximum concentrations in surface water and wetland vegetation, arithmetic mean concentrations in soil and sediment.

9.0 STAGE II ERC – ANALYSIS - EFFECTS ASSESSSMENT

9.1 Purpose

The purpose of this section is to summarize available toxicological data and establish toxicity reference values and benchmarks for COPECs for the ERC. It is beyond the scope of this ERC to provide a comprehensive review of COPEC toxicity data. In this section, only summary data relating to aquatic organisms, plants, birds and mammals are discussed. Specifically, the availability of both dietary exposure and tissue residue-based toxicological data was evaluated for COPECs. The limitations of these toxicity data are discussed. The information in this section will be utilized with data from the section on Exposure Assessment (Section 8.0) to conduct the risk characterization (Section 10.0).

9.2 Toxicity Reference Values (TRVs)

The toxicity reference value (TRV) is the concentration of a chemical in water, food, or the tissues of a receptor that will not cause toxicity to receptors of concern. Ideally, TRVs are derived from chronic toxicity studies in which an ecologically relevant endpoint was assessed in the species of concern, or a closely related species. While TRVs can be expressed or defined as no observable adverse effect levels (NOAELs), the use of lowest observable adverse effect levels (LOAELs) is generally preferred for a Stage II ERC or baseline ERA as NOAELs by definition incorporate greater uncertainty than LOAELs (Sample et al., 1996). Alternatively, TRVs can be expressed as the geometric mean of the NOAEL and LOAEL to provide a conservative estimate of a threshold of effect (Tillitt *et al.,* 1996). For this ERC, values are presented for both NOAELs and LOAELs for comparison.

MCP guidance recommends selection of the lowest available LOAEL and the highest available NOAEL. Furthermore, the MCP states that, "if an appropriate state or federal agency has proposed a toxicity value as a criterion for the protection of wildlife, that value should be used unless it is clearly not applicable for the specific case, and a more appropriate value is identified." Therefore, some of the TRVs that were established in this ERC are from the recently issued USEPA Draft Ecological Soil Screening Level Guidance (USEPA, 2000a) in which TRVs were developed for a number of COPECs. Additionally, USEPA water quality criteria (WQC) were utilized since they are federally promulgated criteria and are specifically adopted by MADEP. However, USEPA WQC for copper does not account for site-specific conditions that indicate reduced bioavailability. Thus, additional sources of related data from the peer-reviewed literature were evaluated.

Sources of toxicological data that were reviewed to develop TRVs included primary peer-reviewed scientific literature, pertinent reviews of individual COPECs (*e.g.*, journal review articles, USFWS Contaminant Hazard Reviews, *etc.*), Draft Ecological Soil Screening Level Guidance (USEPA, 2000a), Oak Ridge National Laboratory report on benchmarks for wildlife, miscellaneous USEPA reports, and other relevant sources of information. In this ERC, endpoints such as effects on reproductive and developmental toxicity and reduced survival were evaluated and used whenever possible. However, for some COPECs, studies reporting systemic effects were utilized if the preferred endpoints were not available. In such situations, appropriate uncertainty factors were applied (as described below).

It is, therefore, essential to perform a critical evaluation of the applicability of the toxicological data to the site-specific receptors of concern and exposure pathways. TRVs derived in the same species are not available for the majority of wildlife receptors and, therefore, it is necessary to derive TRVs using toxicological data for surrogate species in combination with uncertainty factors. Uncertainty concerning interpretation of the toxicity test information among different species, different laboratory endpoints, and differences in experimental design, age of test animals, duration of test, *etc.*, are addressed by applying uncertainty factors (UFs) to the toxicology data to derive the final TRV. For this ERC, general recommendations of Sample *et al.*, (1996), USEPA (1995), and USEPA Region 8 (Henningsen and Hoff, 1997) were considered for the derivation and use of uncertainty factors.

9.2.1 Exposure Duration Extrapolation (UF_A)

This factor is used to estimate the LOAEL or dose of a chemical when only acute (short-term) toxicity test data are available. In situations where a chronic NOAEL value for an ecologically relevant endpoint was not available, the NOAEL can be estimated by a LOAEL to NOAEL uncertainty factor of between 1 and 10 (USEPA, 1995). In this ERC, NOAEL and LOAEL values were identified for most of the COPECs for avian and mammalian receptors. However, in some cases, a LOAEL was not available and needed to be estimated from a NOAEL and *vise versa*. Thus, a factor of 3 was utilized to convert NOAELs to LOAELs and *vice versa*. As for duration and timing of exposure, the definition of chronic exposure was adapted from Sample *et al.*, (1996). In this definition, exposures that occurred during critical lifestages, such as reproductive and developmental time points, were considered to represent chronic exposures since these lifestages are very sensitive to adverse effects because of the coordination of multiple pathways of differentiation and proliferation of cells occurring within the embryo. However, for calculating toxicity values from data for sub-chronic tests, the resultant NOAEL or LOAEL values were divided by an additional factor of 10 (as recommended in the MCP Chapter 9 Guidance, p. 9-84).

9.2.2 Intertaxon Variability Extrapolation (UF_B)

This factor is used to estimate an effects concentration or dose for a receptor from laboratory test species and extrapolate this data to the receptor of concern. Obviously, mammalian test species were never extrapolated to avian species and *vice versa*. While every attempt was made to identify studies conducted with the receptors of concern and COPECs for this site, there were relatively few studies that were conducted with both the receptors of concern and the COPECs for this site. While the uncertainty factor for the extrapolation between species was determined for each receptor and the available toxicological data on similar species, the range of these uncertainty factors used in this ERC were consistent with those of USEPA (Henningsen and Hoff, 1997). The uncertainty factor for species within the same family were set to 1 (as recommended in the MCP Chapter 9 Guidance, p. 9-84), whereas the uncertainty factor for species of the same order but different family were set to 5 (Henningsen and Hoff, 1997).

9.2.3 Toxicologic Endpoint Extrapolation (UF_C)

This factor is used to estimate NOAEL and/or LOAEL values from studies that report other endpoints. An effort was made to identify studies for which ecologically relevant endpoints were assessed. However, in some cases, this information was unavailable and an extrapolation needed to be made. The uncertainty factor for this endpoint extrapolation was between 1 and 3 in this ERC.

9.2.4 Other Modifying Factors (UF_D)

These factors incorporate other sources of uncertainty, including: relevance of endpoint to ecological health, extrapolation from laboratory to field, study conducted with relevant co-contaminants, endpoint is mechanistically unclear (vs. clear), study species is either highly sensitive or highly resistant, ratios used to estimate whole body burden from tissue, intraspecific variability, other applicable modifiers. None of the TRVs determined in this ERC required modification with this uncertainty factor. After consideration of the available data and necessary uncertainty factors, the TRV is calculated using the equation:

$$TRV = \frac{\text{Study Dose}}{(UF_A * UF_B * UF_C * UF_D)} \qquad \text{Eq. 9-1}$$

9.2.5 Effects Benchmarks for Aquatic Organisms

The primary effects concentrations for surface water that were evaluated for COPECs at this site were the national ambient water quality criteria (WQC). Although not entirely aquatic, the potential adverse effects to amphibians were assessed by comparison to WQC since surface water is likely to be a major exposure pathway for amphibians and other aquatic organisms (refer to section 8.2.1 for more details). While it may be possible to estimate exposure through dietary and dermal routes of exposure for amphibians, comparable effects data are insufficient in the available toxicological literature. Furthermore, early developmental stages are often the most sensitive and would occur in water for amphibians. It is recognized that these surface water quality criteria were developed with the intention of protecting 95% of species 95% of the time. Thus, they are quite conservative values and do not indicate that actual effects are occurring to most organisms at these concentrations. Furthermore, for some metals, USEPA is considering adoption of criteria that will account for the ameliorating effect of dissolved organic matter (DOM) on the toxicity of metals (USEPA, 2000c and 2000d). Thus, wherever available, literature-derived studies were utilized that were performed on sensitive aquatic organisms under conditions of similar bioavailability to those expected in the surface waters of this site.

9.2.5.1 Weight-of-Evidence Approach – Assigning Weight to Measurement Endpoints

There are potentially two ways of evaluating measurement endpoints for assessment endpoint #1 that is concerned with aquatic organisms: (A) ambient WQC, and (B) site-specific measures of bioavailability. As such, it was necessary to determine the relative weights of each of these measurement endpoints since a qualitative weight-of-evidence approach will be utilized (refer to Section 5.4 for more details). In the qualitative approach, the first step was to assign each measurement endpoint a qualitative score of high, medium, or low (refer to the section on Problem Formulation for more details). The overall score of the two measurement endpoints, based on the attribute-specific scores are as follows:

Measurement Endpoint	Overall Weight
A. Ambient WQC	Medium
B. Site-specific measures of bioavailability	High

Only those COPECs which exceed their respective WQC will be further evaluated with the second measurement endpoint, since the WQC are designed to be conservative. Specifically, the toxicity of copper has been clearly shown to decrease as the concentration of DOM increases (Figure 9-1). The 24 h LC50 data for copper toxicity to *Ceriodaphnia dubia* are very similar to the actual water quality criteria for copper (Kim et al., 1999) because daphnids are among the most sensitive aquatic organisms to the effects of copper toxicity and thus have a strong influence on the resulting WQC. Note also that the acute to chronic ratio for copper toxicity is very low (approximately a factor of 2). If one assumes that DOM has the same ameliorating effect on the toxicity of copper to aquatic organisms in general, which is supported by multiple species tests (Pommery et al., 1983; Playle et al., 1993; Kim et al., 1999), then it should be possible to extrapolate such laboratory results to field measurements of DOM. In this case, with a mean DOM concentration of 7.6 mg/L in the surface waters of the wetland near the former Raytheon facility, the water quality criteria would be increased by approximately 10-fold, reflecting the decreased bioavailability and decreased toxicity of copper in the presence of DOM. Thus, the concentrations of copper in water that would be protective of aquatic organisms, after accounting for site-specific concentrations of DOM, would be 162 μg/L and 106 μg/L (during low flow conditions and hardness of 122.4 mg/L) for acute and chronic conditions, respectively. The concentrations of copper in water that would be protective of aquatic organisms, after accounting for site-specific concentrations of DOM, would be 69 μg/L and 49 μg/L (during high flow conditions and hardness of 49.4 mg/L) for acute and chronic conditions, respectively.

Although not as much is known about the effects of DOM on zinc toxicity (compared to that of copper), the available evidence indicates that DOM ameliorates the toxicity of zinc to aquatic organisms through precipitation, complexation, and adsorption. For example, humic acid (one predominant type of DOM) has been shown to decrease the toxicity of zinc to phytoplankton communities (Hongve et al., 1980), *Daphnia magna* (Paulaskis and Winner, 1988), and *Ceriodaphnia dubia* (Polonsky and Clements, 1999). Several other studies have similarly reported decreased toxicity of zinc in the presence of organic material (Pommery et al., 1983; Wong et al., 1991; and Feoktistov et al., 1991). For the purposes of this ERC and given the site-specific mean concentration of DOM of 7.6 mg/L, it is assumed that the water quality criteria for zinc would be increased by approximately 2-fold, reflecting the decreased bioavailability and decreased toxicity of zinc in the presence of DOM. Thus, the concentrations of zinc in water that would be protective of aquatic organisms, after accounting for site-specific concentrations of DOM, would be 556.28 μg/L and 280.42 μg/L (during low flow conditions and hardness of 122.4 mg/L) for acute and chronic conditions, respectively. The concentrations of zinc in water that would be protective of aquatic organisms, after accounting for site-specific concentrations of DOM, would be 258 μg/L and 125.7 μg/L (during high flow conditions and hardness of 49.4 mg/L) for acute and chronic conditions, respectively.

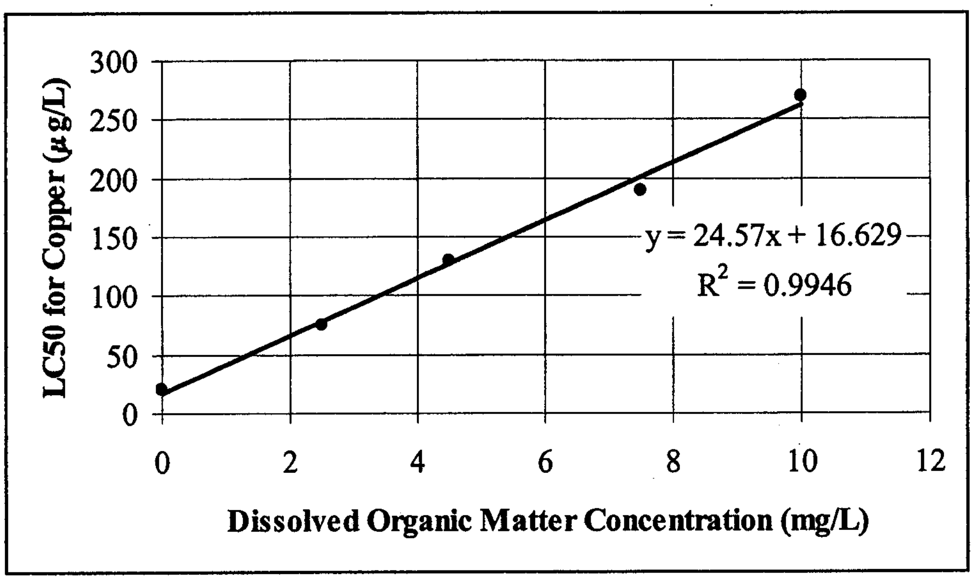

Figure 9-1. Ameliorating effect of increasing concentrations of dissolved organic matter (DOM) on the the 24 h LC_{50} endpoint in *Ceriodaphnia dubia*, which is one of the most sensitive species to copper toxicity. Data from Kim et al., (1999). Note that the DOM concentrations in surface water samples range from 5.4 to 11 mg/L (mean = 7.6 mg/L) from the wetlands near the former Raytheon facility.

9.2.6 Effects Benchmarks for Plants - Summary

Three different types of effect concentrations were determined to assess the potential for phytotoxicity: soil-based effect levels from the literature, plant tissue residues, and field-measured soil-based effect levels for phytotoxicity. These three effect concentrations represent three different measurement endpoints for assessment endpoint #2 that is concerned with wetland plants. As such, it was necessary to determine the relative weights of each of these measurement endpoints since a qualitative weight-of-evidence approach will be utilized. In the qualitative approach, the first step was to assign each measurement endpoint a qualitative score of high, medium, or low (refer to the section on Problem Formulation for more details). The overall score of the two measurement endpoints, based on the attribute-specific scores are as follows:

Measurement Endpoint	Overall Weight
A. Soil-based effect levels from the literature	Medium
B. Plant tissue-based effect levels from the literature	Medium
C. Field-measured soil-based effect levels for phytotoxicity	High

The first type of measurement endpoint, or phytotoxicity effect concentration, is based on laboratory studies conducted with plants grown in soil spiked at one or more concentrations of a COPEC (Table 9-1). The values from these literature studies are given a medium weight in the weight-of-evidence approach because of the tendency of these studies to depend on spiking studies, which are overestimating bioavailability and thus are not readily comparable to sites with historical, "aged" deposits of COPECs. In addition, there is a tendency of these studies to use soils with low CEC and low organic carbon content, which is not applicable to this site which has relatively great CEC and organic carbon content. Additional literature-based studies are available for plants grown in nutrient solution but were not considered because there is such a dramatic difference in the effect concentrations between the two types of exposure media.

The second type of measurement endpoint, or phytotoxicity effect concentration, is based on published diagnostic criteria for concentrations of metals that are associated with observed toxicity in plants (Chapman, 1966; Adriano, 1986). With tissue-based effect levels, different values were utilized for different parts of the plant (roots versus tops) because the roots typically contain at least one order of magnitude greater concentrations than the tops of vegetation (Table 9-2; Adriano, 1986). The effects benchmarks for the literature-based types of benchmarks are presented in more detail for each COPEC in section 9.4.

Table 9-1. Soil-based phytotoxicity effect concentrations for COPECs from the scientific literature.

	Soil-based Phytotoxicity Effect Concentration (mg/kg, dry weight)	
COPEC	**Low**	**High**
Antimony	5	NA
Arsenic	37	NA
Cadmium	29	NA
Chromium	50	100
Copper	100	200
Lead	196	494
Mercury	5	50
Silver	10	100
Tin	50	500
Vanadium	150	500
Zinc	190	NA
PAHs (acenaphthene)	2.5	25
PCBs	40	100

NA – not available

Table 9-2. Tissue based phytotoxicity effect concentrations for COPECs.

	Tissue-based Phytotoxicity Effect Concentration (mg/kg, dry weight)	
COPEC	**Typha[1]**	**Cephalanthus[2]**
Antimony	0.1	0.1
Arsenic	11	1.7
Cadmium	3	3
Chromium	5-175	5-175
Copper	100	20
Lead	300	50
Mercury	3	0.5
Silver	1760	4
Tin	2	2
Vanadium	170	2
Zinc	100	100

[1]*Typha latifolia* (cattail roots)
[2]*Cephalanthus occidentalis* (buttonbush seedheads)

Table 9-3. Field-measured soil-based phytotoxicity effect concentrations for COPECs.

COPEC[1]	Soil-Based Phytotoxicity Effect Concentration (mg/kg, dry weight) 95% LCL
Chromium	6495
Copper	4295
Lead	661
Silver	172

[1]Only those COPECs for which soil concentrations exceeded literature-based phytotoxicity screening values are presented.

The third type of phytotoxicity effect concentration is based on field-measured concentrations of COPECs (only those that exceed literature-based effect levels described below) in soil that are found within the area of stunted vegetation (Table 9-3). The values from this type of evaluation are given the greatest weight in a weight-of-evidence approach since they are based on values determined in the field. As such, these values take into account site-specific differences in bioavailability due to "aging" of the COPECs in soil, elevated concentrations of cation exchange capacity (CEC), elevated concentrations of organic carbon, multiple speciations and complexations of the COPECs that are metals. In a sense, this measurement endpoint is analogous to an *in situ* toxicity test for phytotoxicity. Thus, this set of effect concentrations is the most highly realistic and predictive for other areas of the site. To maintain conservatism, the 95% lower confidence limit (LCL) of the arithmetic mean within the area of stunted vegetation was determined to be a representative threshold for field-measured phytotoxicity. This value was then compared to the sitewide 95% upper confidence limit (UCL) of the arithmetic mean soil concentration for areas outside of the "Area of Readily Apparent Harm".

Support for the ameliorating effects of CEC and organic matter content on the bioavailability and toxicity of some metals in soil are available from a number of studies and reports (Reuther *et al.*, 1953; Demayo *et al.*, 1982; Beyer, 1990; USEPA, 2000a; Brun *et al.*, 2001). For example, copper has been shown to be toxic to citrus seedlings at concentrations of copper that are greater than 40 mg/kg per milli-equivalent (mEq) of CEC, whereas concentrations of copper that are less than 20 mg/kg per mEq are not toxic (Reuther et al., 1953). Thus, using this relationship of phytotoxicity effect concentrations with the site-specific mean CEC of 153 mEq, the corresponding CEC-normalized effect concentrations are 3,060 mg/kg (no toxicity) and 6,120 mg/kg (toxicity). This is very similar to the observed concentrations of copper in wetland soil at this site within the area of stunted vegetation, which are approximately 4300 to 8600 mg/kg (based on the lower and upper confidence limits of the arithmetic mean). Thus, normalization to site-specific CEC data appears to be a valid and predictive tool for addressing issues of bioavailability of metals for plants.

9.2.7 Mammalian and Avian Wildlife Dietary Toxicity Reference Values (TRVs) - Summary

Dietary TRVs were determined by evaluating high quality toxicological studies. Whenever possible, to minimize the use of uncertainty factors, an attempt was made to focus on studies that evaluated test species that were as similar as possible to the avian and mammalian wildlife receptors of concern in this ERC. In addition, an attempt was made to identify studies that evaluated sensitive ecologically relevant endpoints (primarily effects on reproduction), similar exposure routes (primarily dietary), exposure duration (chronic and during sensitive life stages), and forms of chemicals as those expected at this site. Dietary TRVs for mammalian and avian receptors of concern at this site are summarized in Tables 9-4 and 9-5. A description of the toxicological studies that were selected as most appropriate for the derivation of TRVs are discussed for each specific COPEC later in this section, including derivation of TRVs from the original toxicological studies with the use of uncertainty factors.

Table 9-4. Summary of Mammalian Toxicity Reference Values (TRVs) for all COPECs[1].

	White-Tailed Deer TRV (mg/kg/d)		Meadow Vole TRV (mg/kg/d)		Muskrat TRV (mg/kg/d)	
COPEC	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Antimony	4.4	NA	4.4	NA	4.4	NA
Arsenic	0.084	0.252	0.42	1.26	0.42	1.26
Cadmium	0.2	2	1.0	10	1	10
Chromium	24.5	NA	24.5	NA	24.5	NA
Copper	1.28	2.58	6.40	12.9	6.4	12.9
Lead	1.6	16	8	80	8	80
Mercury	2.64	7.92	13.2	40	13.2	40
Silver	4.93	14.80	24.67	74	24.67	74
Tin	NA	NA	NA	NA	NA	NA
Vanadium	0.33	1	1.67	5	1.67	5
Zinc	32	64	160	320	160	320
PAHs	3.05	9.16	15.27	45.8	15.27	45.8
PCBs	0.06	0.3	0.32	1.5	0.32	1.5

[1]Refer to document sections on specific chemical stressors for calculations of species-specific TRVs from generic mammalian TRVs.
NA= Not Available

Table 9-5. Summary of Avian Toxicity Reference Values (TRVs) for all COPECs[1].

	Mallard TRV (mg/kg/d)		Red-Tailed Hawk TRV (mg/kg/d)	
COPEC	NOAEL	LOAEL	NOAEL	LOAEL
Antimony	NA	NA	NA	NA
Arsenic	10	40	2	8
Cadmium	1.45	20	0.29	4.0
Chromium	1.6	NA	1.6	NA
Copper	9.4	12.3	9.4	12.3
Lead	0.23	2.26	0.23	2.26
Mercury	0.09	0.18	0.09	0.18
Silver	0.42	0.83	0.08	0.17
Tin	NA	NA	NA	NA
Vanadium	11.4	34.2	2.28	6.8
Zinc	2.9	26.2	2.9	26.2
PAHs	40	400	8	80
PCBs	0.12	0.36	0.12	0.36

[1]Refer to document sections on specific chemical stressors for calculations of species-specific TRVs from generic avian TRVs.
NA= Not Available

9.3 Non-Chemical Stressors

It is possible that there is potential impact from non-chemical stressors at this site, including habitat suitability, siltation, and urbanization. Since these natural and anthropogenic stressors can have potential impact on certain receptor populations and productivity, the effects of these stressors are likely confounding factors in this ERC.

9.4 Chemical Stressors

Chemicals of potential ecological concern (COPECs) are each generally discussed below for toxicity reference values for phytotoxicity, avian toxicity, and mammalian toxicity. Physical and chemical properties and natural abundance, bioavailability, mechanisms of action, and toxic properties of COPECs are discussed elewhere (Appendix H).

9.4.1 Toxicity Reference Values (TRVs) and Effect Benchmarks for Antimony

9.4.1.1 Plants

No primary reference data were located that describe a soil-based phytotoxic level for antimony. The benchmark is based on a report of unspecified toxic effects on plants grown in a surface soil with the addition of 5 ppm Sb (Kabata-Pendias and Pendias 1984 as cited in Efroymson *et al.*, 1997).

In general, little is known about plant uptake of antimony and its phytotoxicity. However, there are indications that normal concentrations in most terrestrial plants should be around 0.1 mg/kg (dry weight; Adriano, 1986). Thus, the tissue based benchmark level for both cattails and buttonbush is set to 0.1 mg/kg. There is considerable uncertainty regarding an actual threshold concentration at which phytotoxicity would be observed although it is likely to be greater than 0.1 mg/kg.

9.4.1.2 Mammals

The recent USEPA Draft Ecological Soil Screening Level Guidance contains a toxicity reference value (TRV) of 4.4 mg/kg-d (dry weight) for antimony to mammals. The benchmark is based on ten studies.

Antimony NOAEL - Mammalian: TRV = 4.4 mg/kg-d

9.4.1.3 Birds

No studies were located that examined antimony toxicity to avian wildlife.

9.4.2 Toxicity Reference Values (TRVs) and Effect Benchmarks for Arsenic

9.4.2.1 Plants

The recent USEPA Draft Ecological Soil Screening Level Guidance contains a soil-based phytotoxic benchmark level of 37 mg/kg (dry weight) for arsenic to plants. The benchmark is based on nine records that were obtained from two papers and six species. All of the toxicity data were based on growth effects, a chronic endpoint. The experiments were conducted with natural soils under conditions of high or relatively high bioavailability.

Normal concentrations of arsenic in most terrestrial plants are between 0.01-5 mg/kg (dry weight; Adriano, 1986). The tissue based benchmark level for cattails is 11 mg/kg, based on the lowest number in a range of critical values for barley leaves and shoots (Adriano, 1986). The tissue based benchmark for buttonbush is 1.7 mg/kg based on a no effect concentration in leaves from fruit trees (Adriano, 1986).

9.4.2.2 Mammals

Schroeder and Mitchener (1971) conducted a 3-generation reproduction study in which CD mice were exposed by diet to one dose of arsenic salts (1.26 mg/kg/d combined exposure from water and feed as calculated in Sample *et al.*, 1996) through a critical reproductive lifestage. At 1.26 mg/ks/d, adverse effects were observed including a decrease in litter size. Since the study considered dietary exposure during reproduction, the 1.26 dose was considered to be a chronic LOAEL. A chronic NOAEL of 0.42 was estimated by dividing the chronic LOAEL by an uncertainty factor of three. Arsenic TRV derivations for mammalian receptors of concern are shown in Table 9-6.

Table 9-6. Arsenic TRV derivations for mamalian receptors of concern.

Receptor of concern	White-tailed deer		Meadow Vole		Muskrat	
Study Chemical	Arsenite salt		Arsenite salt		Arsenite salt	
Reference	Schroeder & Mitchener, 1971		Schroeder & Mitchener, 1971		Schroeder & Mitchener, 1971	
	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Reference TRV (mg/kg-d)	NA	1.26	NA	1.26	NA	1.26
Test Species UCF	5 (mouse)	5 (mouse)	1 (mouse)	1 (mouse)	1 (mouse)	1 (mouse)
Duration UCF	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)
Endpoint UCF	3 (reprod.)	1 (reprod.)	3 (reprod.)	1 (reprod.)	3 (reprod.)	1 (reprod.)
Total UCF	15	5	3	1	3	1
Final TRV (mg/kg-d)	**0.08**	**0.25**	**0.42**	**1.26**	**0.42**	**1.26**

Final TRV = Reference TRV / Total UCF
NA = Not available

9.4.2.3 Birds

A reproduction study in which 1-year-old mallard ducks were exposed by diet to four doses of sodium arsenite (0, 25, 100, and 400 mg/kg in feed) from four weeks prior to pairing through multiple hatching cycles was conducted by Stanley et al (1994). Conversion of concentrations in diet to a daily dose are based on a body weight of 1 kg and a food consumption rate of 0.1 kg/d (Sample *et al.*, 1996). No adverse effects were observed at a dose level of 100 mg/kg in feed (or 10 mg/kg-d). At 400 mg/kg in feed (or 40 mg/kg/d), adverse effects were observed including a decrease in duckling growth rate and duckling production. Since the study considered dietary exposure during reproduction, the 10 and 40 mg/kg/d doses were considered to be chronic NOAELs and LOAELs, respectively. Arsenic TRV derivations for avian receptors of concern are shown in Table 9-7.

Table 9-7. Arsenic TRV derivations for avian receptors of concern.

Receptor of concern	Mallard		Red-tailed hawk	
Study Chemical	Sodium arsenate		Sodium arsenate	
Reference	Stanley et al., 1994		Stanley et al., 1994	
	NOAEL	LOAEL	NOAEL	LOAEL
Reference TRV (mg/kg-d)	10	40	10	40
Test Species UCF	1 (mallard)	1 (mallard)	5 (mallard)	5 (mallard)
Duration UCF	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)
Endpoint UCF	1 (reprod.)	1 (reprod.)	1 (reprod.)	1 (reprod.)
Total UCF	1	1	5	5
Final TRV (mg/kg-d)	**10.00**	**40.00**	**2.00**	**8.00**

Final TRV = Reference TRV / Total UCF
NA = Not available

9.4.3 Toxicity Reference Values (TRVs) and Effect Benchmarks for Cadmium

9.4.3.1 Plants

The recent USEPA Draft Ecological Soil Screening Level Guidance contains a soil-based phytotoxic benchmark level of 29 mg/kg (dry weight) for cadmium to plants. The benchmark is based on nine records that were obtained from three papers and eight species. All of the toxicity data were based on growth effects, a chronic endpoint. The experiments were conducted with natural soils under conditions of high or relatively high bioavailability.

Normal concentrations of cadmium in most terrestrial plants are between 0.05-6 mg/kg (dry weight) for roots and tops (Adriano, 1986). The tissue based benchmark concentrations for cattails and buttonbush are 3 mg/kg, based on the lowest number in a range of critical values (range = 3-80 mg/kg) for reduced yield for a number of species (Adriano, 1986).

9.4.3.2 Mammals

No high quality studies were located that examined cadmium toxicity to terrestrial mammalian wildlife. The NOAEL and LOAEL identified for Cd, therefore, are based on studies conducted with laboratory rodents. Sutuo *et al.*, (1980) conducted a reproduction study in which rats were exposed by oral gavage to four doses of cadmium chloride (0, 0.1, 1.0, and 10 mg/kg/d) for 6 weeks through the sensitive reproductive and developmental stages of mating and gestation. No adverse effects were observed at a dose level of 1 mg/kg-d. At 10 mg/kg/d, adverse effects were observed including a decrease in implantations and fetal survivorship and an increase in fetal resorptions. Since the study considered oral exposure during reproduction, the 1 and 10 mg/kg/d doses were considered to be chronic NOAELs and LOAELs, respectively. Cadmium TRV derivation for mammalian receptors of concern are shown in Table 9-8.

Table 9-8. Cadmium TRV derivation for mammalian receptors of concern.

Receptor of concern	White-tailed deer		Meadow Vole		Muskrat	
Study Chemical	Cadmium chloride		Cadmium chloride		Cadmium chloride	
Reference	Sutuo et al., 1980		Sutuo et al., 1980		Sutuo et al., 1980	
	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Reference TRV (mg/kg-d)	1	10	1	10	1	10
Test Species UCF	5 (rat)	5 (rat)	1 (rat)	1 (rat)	1 (rat)	1 (rat)
Duration UCF	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)
Endpoint UCF	1 (reprod.)	1 (reprod.)	1 (reprod.)	1 (reprod.)	1 (reprod.)	1 (reprod.)
Total UCF	5	5	1	1	1	1
Final TRV (mg/kg-d)	**0.20**	**2.00**	**1.00**	**10.00**	**1.00**	**10.00**

Final TRV = Reference TRV / Total UCF

9.4.3.3 Birds

The literature reviewed included studies examining cadmium toxicity to both laboratory birds and more environmentally relevant species, such as the mallard duck.

White and Finley (1978) conducted a reproduction study in which mallard ducks were exposed by diet to three doses of cadmium chloride (0.15, 1.45, 20 mg/kg/d) for 90 d through a critical reproductive lifestage. Conversion of concentrations in diet to a daily dose are based on a body weight of 1.153 kg and a food consumption rate of 0.11 kg/d (White and Finley, 1978). No adverse effects were observed at a dose level of 1.45 mg/kg-d. At 20 mg/kg/d, adverse effects were observed including a decrease in the number of eggs produced. Since the study considered dietary exposure during reproduction, the 1.45 and 20 mg/kg/d doses were considered to be chronic NOAELs and LOAELs, respectively. Cadmium TRV derivations for avian receptors of concern are shown in Table 9-9.

Table 9-9. Cadmium TRV derivations for avian receptors of concern.

Receptor of concern	Mallard		Red-tailed hawk	
Study Chemical	Cadmium chloride		Cadmium chloride	
Reference	White and Finley, 1978		White and Finley, 1978	
	NOAEL	LOAEL	NOAEL	LOAEL
Reference TRV (mg/kg-d)	1.45	20	1.45	20
Test Species UCF	1 (mallard)	1 (mallard)	5 (mallard)	5 (mallard)
Duration UCF	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)
Endpoint UCF	1 (reprod.)	1 (reprod.)	1 (reprod.)	1 (reprod.)
Total UCF	1	1	5	5
Final TRV (mg/kg-d)	**1.45**	**20.00**	**0.29**	**4.00**

Final TRV = Reference TRV / Total UCF

9.4.4 Toxicity Reference Values (TRVs) and Effect Benchmarks for Chromium

9.4.4.1 Plants

Few studies were identified that have evaluated phytotoxicity of chromium in soils. The soil-based high and low benchmarks are based on a study of corn, beans, and tomatoes that reported stunted growth at 100 mg/kg but not at 50 mg/kg (Schueneman, 1974 in Adriano, 1986). The recent USEPA Draft Ecological Soil Screening Level Guidance contains a soil-based phytotoxic benchmark level of 5 mg/kg (dry weight) for chromium to plants. However, this value was not used in this ERC because background concentrations of chromium at this site are greater than 5 mg/kg. The benchmark is based on seven records that were obtained from one paper and five species. All of the toxicity data were based on growth effects, a chronic endpoint. The experiments were conducted with natural soils under conditions of high or relatively high bioavailability. However, for the value of 5 mg/kg for chromium, the Draft Ecological Soil Screening Level Guidance recommends further testing on the phytotoxic effects of chromium to strengthen the data set. Since this value is less than background and much less than the actual threshold for toxicity as determined from available studies, it was not used in this ERC.

Normal concentrations of chromium in terrestrial plants are approximately 1 mg/kg (dry weight; unspecified plant tissue, Adriano, 1986). The range of tissue based benchmark concentrations for cattails and buttonbush are 5-175 mg/kg, based on the range of critical values for barley, corn, oats, citrus, and tobacco (Adriano, 1986).

9.4.4.2 Mammals

The recent USEPA Draft Ecological Soil Screening Level Guidance contains a dietary toxicity reference value (TRV) of 24.5 mg/kg (dry weight) for chromium to mammalian species.

Chromium NOAEL - Mammalian: TRV = 24.5 mg/kg-d

9.4.4.3 Birds

The recent USEPA Draft Ecological Soil Screening Level Guidance contains a dietary toxicity reference value (TRV) of 1.6 mg/kg (dry weight) for chromium to avian species.

Chromium NOAEL - Avian: TRV = 1.6 mg/kg-d

9.4.5 Toxicity Reference Values (TRVs) and Effect Benchmarks for Chromium (VI)

9.4.5.1 Plants

Adriano (1986) summarizes the available studies of the effects of hexavalent chromium on plants grown in soil. The lowest exposure of chromium that resulted in stunted growth, a chronic effect, was 50 mg/kg in soil affecting the growth of barley. Barley death resulted at an exposure of 500 mg/kg. Another study observed no toxicity to sweet orange seedlings grown in soil with 75 ppm chromium (VI). From these studies, the low and high soil-based benchmark concentrations are 50 and 500 mg/kg, respectively.

9.4.5.2 Mammals

The recent USEPA Daft Ecological Soil Screening Level Guidance contains a dietary toxicity reference value (TRV) of 22 mg/kg/d (dry weight) for hexavalent chromium to mammalian species.

9.4.5.3 Birds

No studies were located that examined hexavalent chromium toxicity to avian wildlife. However, since TRVs are available for trivalent chromium and/or total chromium the effects of chromium are being evaluated.

9.4.6 Toxicity Reference Values (TRVs) and Effect Benchmarks for Copper

9.4.6.1 Plants

Few studies were identified that have evaluated phytotoxicity of copper in soils. Kjaer and Elmegaard (1996) conducted a study in which a non-crop plant species, Black bindweed (*Polygonum convolvulus* L.), was exposed to 5 soil concentrations of copper sulfate (0, 125, 200, 315, and 500 mg/kg dry weight in soil) for 105 d. At 125 mg/kg, copper did not cause any significant adverse effects. At 200 mg/kg, copper caused 33% mortality which was statistically significant. Other more recent studies from these same authors indicate that copper becomes less bioavailable as a result of soil properties such as organic carbon content and through aging processes (Kjaer et al., 1998; Pederson et al., 2000).

In another study, Wallace *et al.* (1977) evaluated the effects of copper sulfate to a loam soil, on leaf and stem weights of bush beans grown from seed for 17 d. Leaf weight was reduced 26% by 200 mg/kg copper, while 100 mg/kg had no effect. From this study, the low and high soil-based benchmark concentrations are 100 and 200 mg/kg, respectively.

Normal concentrations of copper in terrestrial plants are between 1-28 mg/kg (dry weight; Adriano, 1986). Chino (1980 in Adriano, 1986) reported that for rice plants, toxic concentrations in tops were in the range of 20-30 mg/kg, whereas for roots, the toxic range was from 100-300 mg/kg. The tissue based benchmark level for cattails is 100 mg/kg, based on the lowest number in the range of critical values for rice plant roots (Adriano, 1986). The tissue based benchmark for buttonbush is 20 mg/kg based on the lowest number in the range of critical values for rice plant tops (Adriano, 1986).

9.4.6.2 Mammals

The literature reviewed included studies examining copper toxicity to both laboratory studies of rodents and more environmentally relevant species, such as short-tailed shrews.

A 13-week study in which male and female F344/N rats and B6C3F1 mice were exposed by diet to five dose concentrations of cupric sulfate [0, 500 (rats only), 1,000, 2, 000, 4,000, 8,000, 16,000 (mice only) mg/kg in feed] for 91 d was conducted by Hebert et al (1993). Conversion of concentrations in diet to a daily dose was not necessary because this information was provided in the journal article. The daily dose concentrations for male rats were 0, 32, 64, 129, 259, and 551 mg/kg/d. The daily dose concentrations for male mice were 0, 173, 382, 736, 1,563, and 3,201 mg/kg/d. This study confirmed that the target organs for cupric sulfate are the liver and kidney and additionally noted the occurrence of forestomach lesions. No adverse effects were observed at a dose level of 64 mg/kg/d for rats and 362 for mice. At 129 mg/kg/d for rats and 736 mg/kg/d for mice, adverse effects were observed including an increase in the occurrence of forestomach lesions. These numbers are in the same range as the NOAEL value of 257 mg/kg/d observed in the shrew (Dodds-Smith *et al.*, 1992). Of these rodent species, the rat appears to be the most sensitive species and will thus be used to derive the NOAEL and LOAEL values for copper toxicity to mammalian receptors. Since toxicological data for multiple rodent species are available and since the NOAEL value for the most sensitive of these species (*e.g.*, rats) is selected, no interspecies uncertainty factor will be applied in accordance with MCP Chapter 9 Guidance, section 9.5.3.5.). The greatest NOAEL value was not selected (as recommended by MCP Chapter 9 Guidance, section 9.5.3.5.). The greatest NOAEL (mouse) was greater than the LOAEL for the rat. However, since the study considered dietary exposure under subchronic conditions (13 weeks), the doses were divided by a subchronic to chronic conversion factor of 3 to derive chronic NOAELs and LOAELs of 21 and 43 mg/kg/d, respectively. Copper TRV derivations for mammalian receptors of concern are shown in Table 9-10.

Table 9-10. Copper TRV derivations for mammalian receptors of concern.

Receptor of concern	White-tailed deer		Meadow Vole		Muskrat	
Study Chemical Reference	Copper sulfate Hebert et al., 1993		Copper sulfate Hebert et al., 1993		Copper sulfate Hebert et al., 1993	
	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Reference TRV (mg/kg-d)	64	129	64	129	64	129
Test Species UCF	5 (mouse)	5 (mouse)	1 (mouse)	1 (mouse)	1 (mouse)	1 (mouse)
Duration UCF	10 (subchronic)	10 (subchronic)	10 (subchronic)	10 (subchronic)	10 (subchronic)	10 (subchronic)
Endpoint UCF	1 (reprod. / growth)	1 (reprod. / growth)	1 (reprod. / growth)	1 (reprod. / growth)	1 (reprod. / growth)	1 (reprod. / growth)
Total UCF	50	50	10	10	10	10
Final TRV (mg/kg-d)	**1.28**	**2.58**	**6.40**	**12.90**	**6.40**	**12.90**

Final TRV = Reference TRV / Total UCF

9.4.6.3 Birds

The literature reviewed included studies examining copper toxicity to both laboratory birds and more environmentally relevant species, such as the mallard duck.

A 28 d NOAEL of 83.3 mg/kg-d for mallard ducklings fed copper sulfate was reported by Van Vleet (1982). The NOAEL was based on mortality. Although this receptor is ecologically relevant, lethality is a relatively insensitive endpoint. Moreover, this dose exceeds the LOAEL observed in other studies.

A study in which 1-day old chicks were exposed by diet to eleven doses of copper oxide (36.8, 52.0, 73.5, 104, 147.1, 208, 294.1, 403, 570, and 749 mg/kg in the feed) for 70 d through a sensitive growth lifestage was conducted by Mehring *et al.*, (1960). Conversion of concentrations in diet to a daily dose follows the calculations of Sample *et al.*, (1996) which is based on a 5-week old chick body weight of 0.534 kg and a food consumption rate of 5-week old chicks of 0.11 kg/d (USEPA, 1988). No adverse effects were observed at a dose level of 570 mg/kg in feed or 47 mg/kg/d. At 749 mg/kg in feed or 61.7 mg/kg/d, adverse effects were observed including a 30% reduction in growth. Since the study considered dietary exposure during a sensitive lifestage, the 47 and 61.7 mg/kg/d doses were considered to be chronic NOAELs and LOAELs, respectively. Copper TRV derivation for avian receptors of concern are shown in Table 9-11.

Table 9-11. Copper TRV derivation for avian receptors of concern.

Receptor of concern	Mallard		Red-tailed Hawk	
Study Chemical	Copper oxide		Copper oxide	
Reference	Mehring et al., 1960		Mehring et al., 1960	
	NOAEL	LOAEL	NOAEL	LOAEL
Reference TRV (mg/kg-d)	47	61.7	47	61.7
Test Species UCF	5 (chicken)	5 (chicken)	5 (chicken)	5 (chicken)
Duration UCF	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)
Endpoint UCF	1 (growth / mortality)	1 (growth / mortality)	1 (growth / mortality)	1 (growth / mortality)
Total UCF	5	5	5	5
Final TRV (mg/kg-d)	**9.40**	**12.34**	**9.40**	**12.34**

Final TRV = Reference TRV / Total UCF

9.4.7 Toxicity Reference Values (TRVs) and Effect Benchmarks for Lead

9.4.7.1 Plants

Seven studies were identified that evaluated the soil-based phytotoxic effects of lead at multiple concentrations that reported effect levels and no-effect levels. The summaries of these studies (shown below) are from Efroymson *et al.*, (1997). The effects of lead, added to a 1:1:1 mixture of soil, sand and peat moss as lead chloride, on 1-year-old seedlings of autumn olive (*Elaeagnus umbellata*) grown for 49 d were measured by Rolfe and Bazzaz (1975). They found a reduction in transpiration of approximately25 % with the addition of 160 mg/kg lead, while 80 mg/kg had no effect.

The response of red oak seedlings grown for 16 weeks in a sandy loam soil (pH 6, % organic matter 1.5) with addition of lead chloride was measured by Dixon (1988). Lead at 50 mg/kg reduced tree weight by 26%, while 20 mg/kg had no effect.

The effects of addition of lead oxide to an alluvial soil (pH 6) on growth and yield of wheat grown from seed to maturity were measured by Muramoto *et al.* (1990). Root weight was reduced 22% by 1000 mg/kg lead, while 300 mg/kg had no effect.

In a study using Brown earth soil, Khan and Frankland (1984) investigated the effects of lead chloride, the less soluble lead oxide, or a combination, on root weight of wheat and oats. Wheat root weight was reduced 34% by the addition of 1000 mg/kg lead chloride, while 500 mg/kg had no effect. Oat growth was reduced 37% by the addition of 500 mg/kg lead chloride, while 100 mg/kg had no effect. Wheat and oats were grown from seedlings for 42 d.

A 48% reduction in corn root length after 7 d of growth from seed in a loamy sand soil (pH 6.5, % organic matter 2, CEC 2 meq/100g soil) to which 500 mg/kg lead chloride was added, was measured by Hassett *et al.* (1976). Lead at 250 mg/kg did not affect growth.

Corn (*Zea mays L.*) grown from seed for 31 d in a loamy sand used in the 1976 work (pH 6, CEC 2 meq/100g soil) experienced a 42% decrease in plant weight after addition of 250 mg/kg lead (Miller *et al.*, 1977). Lead at 125 mg/kg did not affect growth.

The low and high soil-based benchmark levels for phytotoxicity are based on the average of the no-effect and effect levels from these 7 studies, which are 196 and 494 mg/kg, respectively.

Normal concentrations of lead in terrestrial plants are between 6-200 and 14-18 mg/kg (dry weight) in roots and leaves, respectively (Adriano, 1986). The tissue based benchmark level for cattails is 300 mg/kg, based on the lowest number in a range of critical values for rice plant roots (Adriano, 1986). The tissue based benchmark for buttonbush is 50 mg/kg based on the lowest number in a range of critical values for rice plant tops (Adriano, 1986).

9.4.7.2 Mammals

A 3-generation reproduction study in which rats were exposed by diet to five dose concentrations of lead acetate (10, 50, 100, 1,000, 2,000 mg/kg in feed) through the sensitive reproductive and developmental stages of mating and gestation was conducted by Azar *et al.*, (1973). Conversion of concentrations in diet to a daily dose are based on a body weight of 0.35 kg and a food consumption rate of 0.028 kg/d (USEPA, 1988). No adverse effects were observed at a dose of 100 mg/kg in feed (or 8 mg/kg/d). While none of the exposure concentrations resulted in changes in the number of pregnancies, number of live births, or other reproductive indices, exposure to lead at 1000 mg/kg in feed (or 80 mg/kg/d), resulted in adverse effects including a decrease in offspring weights and an increase in kidney damage in the young. Since the study considered dietary exposure during reproduction, the 8 and 80 mg/kg/d doses were considered to be chronic NOAELs and LOAELs, respectively. Lead TRV derivations for mammalian receptors of concern are shown in Table 9-12.

Table 9-12. Lead TRV derivations for mammalian receptors of concern.

Receptor of concern	White-tailed deer		Meadow Vole		Muskrat	
Study Chemical	Lead acetate		Lead acetate		Lead acetate	
Reference	Azar et al., 1973		Azar et al., 1973		Azar et al., 1973	
	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Reference TRV (mg/kg-d)	8	80	8	80	8	80
Test Species UCF	5 (rat)	5 (rat)	1 (rat)	1 (rat)	1 (rat)	1 (rat)
Duration UCF	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)
Endpoint UCF	1 (reprod.)	1 (reprod.)	1 (reprod.)	1 (reprod.)	1 (reprod.)	1 (reprod.)
Total UCF	5	5	1	1	1	1
Final TRV (mg/kg-d)	**1.60**	**16.00**	**8.00**	**80.00**	**8.00**	**80.00**

Final TRV = Reference TRV / Total UCF

9.4.7.3 Birds

The literature reviewed included studies examining lead toxicity to both surrogate birds such as chickens or quail and more environmentally-relevant species, such as the mallard.

An 84 d NOAEL for mallard ducks exposed to lead nitrate was reported by Finley *et al.* (1976). The value reported by Finley *et al.* (1976) as 25 mg/kg in feed was converted to a dietary daily dose of 2.5 mg/kg-d. The NOAEL was based on survival, body weight, and food consumption. Although the mallard is a more ecologically-relevant species than the Japanese quail, the endpoints that were measured were not based on reproductive endpoints.

A reproduction study in which Japanese quail were exposed by diet to four doses of lead acetate (1, 10, 100, and 1000 mg/kg in feed) for 84 d through a critical reproductive lifestage was conducted by Edens et al (1976). Conversion of concentrations in diet to a daily dose are based on a body weight of 0.15 kg and a food consumption rate of 0.0169 kg/d (as calculated in Sample *et al.*, 1996). No adverse effects were observed at a dose level of 10 mg/kg in feed (or 1.13 mg/kg-d). At 100 mg/kg in feed (or 11.3 mg/kg/d), adverse effects were observed including a decrease in egg hatching success. Since the study considered dietary exposure during reproduction, the 1.13 and 11.3 mg/kg/d doses were considered to be chronic NOAELs and LOAELs, respectively. Lead TRV derivations for avian receptors of concern are shown in Table 9-13.

Table 9-13. Lead TRV derivations for avian receptors of concern.

Receptor of concern	Mallard		Red-tailed Hawk	
Study Chemical Reference	Lead acetate Edens et al., 1976		Lead acetate Edens et al., 1976	
	NOAEL	LOAEL	NOAEL	LOAEL
Reference TRV (mg/kg-d)	1.13	11.3	1.13	11.3
Test Species UCF	5 (J. quail)	5 (J. quail)	5 (J. quail)	5 (J. quail)
Duration UCF	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)
Endpoint UCF	1 (reprod.)	1 (reprod.)	1 (reprod.)	1 (reprod.)
Total UCF	5	5	5	5
Final TRV (mg/kg-d)	**0.23**	**2.26**	**0.23**	**2.26**

Final TRV = Reference TRV / Total UCF

9.4.8 Toxicity Reference Values (TRVs) and Effect Benchmarks for Manganese

9.4.8.1 Plants

There is one study in the primary literature that evaluates the effects of soil amended with $MnSO_4$ on plant growth. Bush bean stem weight, a chronic endpoint, was reduced by 29% by 500 mg/kg Mn (lowest concentration tested) in the soil (Wallace et al., 1977). From this study, the low soil based benchmark concentration is 500 mg/kg.

9.4.8.2 Mammals

Laskey et al. (1982) conducted a reproduction study in which rats were exposed by diet to three doses of Mn (32, 88 and 284 mg/kg/d in feed as calculated in Sample et al. 1996) through a critical reproductive lifestage. Adverse effects, including decreased fertility, were observed among rats consuming 284 mg/kg/d Mn. No effects were observed at the lower dose levels. Since the study considered dietary exposure during reporduction, the 284 mg/kg/d dose was considered a chronic LOAEL and 88 mg/kg/d a chronic NOAEL. Manganese TRV derivations for mammalian receptors of concern are shown in Table 9-14.

Table 9-14. Manganese TRV derivations for mammalian receptors of concen.

Receptor of concern	White-tailed deer		Meadow Vole		Muskrat	
Study Chemical	Manganese Oxide		Manganese Oxide		Manganese Oxide	
Reference	Laskey et al. 1982		Laskey et al. 1982		Laskey et al. 1982	
	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Reference TRV (mg/kg-d)	88	284	88	284	88	284
Test Species UCF	5 (rat)	5 (rat)	1 (rat)	1 (rat)	1 (rat)	1 (rat)
Duration UCF	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)
Endpoint UCF	1 (reprod.)	1 (reprod.)	1 (reprod.)	1 (reprod.)	1 (reprod.)	1 (reprod.)
Total UCF	5	5	1	1	1	1
Final TRV (mg/kg-d)	**18**	**57**	**88**	**284**	**88**	**284**

Final TRV = Reference TRV / Total UCF
NA = Not available

9.4.8.3 Birds

Laskey and Edens (1985) conducted a study in which Japanese Quail were exposed to one does level of manganese. No reduction in growth was observed at 977 mg/kg/d. This was considered to be a chronic NOAEL because the study was greater than 10 weeks in duration. A chronic LOAEL of 2931 mg/kg/d was estimated by multiplying the chronic NOAEL by a factor of 3. Manganese TRV derivation for avian receptors of concern are shown in Table 9-15.

Table 9-15. Manganese TRV derivation for avian receptors of concern.

Receptor of concern	Mallard		Red-tailed hawk	
Study Chemical	Manganese Oxide		Manganese Oxide	
Reference	Laskey & Edens 1985		Laskey & Edens 1985	
	NOAEL	LOAEL	NOAEL	LOAEL
Reference TRV (mg/kg-d)	977	NA	977	NA
Test Species UCF	5 (J. quail)	5 (J. quail)	5 (J. quail)	5 (J. quail)
Duration UCF	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)
Endpoint UCF	1 (growth)	0.33 (growth)	1 (growth)	0.33 (growth)
Total UCF	5	1.66	5	1.66
Final TRV (mg/kg-d)	**195**	**589**	**195**	**589**

Final TRV = Reference TRV / Total UCF
NA = Not available

9.4.9 Toxicity Reference Values (TRVs) and Effect Benchmarks for Mercury

9.4.9.1 Plants

Few studies were identified that evaluated the soil-based phytotoxic effects of mercury. Concentrations of mercury chloride in soil at 50 mg/kg have been shown to reduce growth of common bermudagrass (Weaver *et al.*, 1984 in Adriano, 1986) while other species such as velvet bentgrass showed no effect when grown in soils containing mercury up to 450 mg/kg (Estes *et al.*, 1973 in Adriano, 1986). Since a low threshold was not identified from the studies, the low benchmark was derived by dividing the high benchmark by 10. The soil-based low and high phytotoxicity benchmarks for mercury are 5 and 50 mg/kg.

Normal concentrations of mercury in terrestrial plants are between 0.01-0.33 mg/kg (dry weight) (Adriano, 1986). The tissue based benchmark level for cattails is 3 mg/kg, based on the critical value for barley tissue (Adriano, 1986). The tissue based benchmark for buttonbush is 0.5 mg/kg based on the critical value for rice plant tops (Adriano, 1986).

9.4.9.2 Mammals

Mice were exposed to mercuric sulfide in diet at 30 dose concentrations ranging up to 13.2 mg/kg/d for 20 months (Revis *et al.* 1989). The measured endpoints were reproduction, liver and kidney histology, and mortality. No adverse effects were reported at any dose level. Since the study considered chronic, dietary exposure during reproduction, the 13.2 mg/kg/d dose level was considered to be the chronic NOAEL. A chronic LOAEL 39.6 mg/kg/d was estimated by multiplying the NOAEL by a factor of three. Mercury TRV derivations for mammalian receptors of concern are shown in Table 9-16.

Table 9-16. Mercury TRV derivations for mammalian receptors of concern.

Receptor of concern	White-tailed deer		Meadow Vole		Muskrat	
Study Chemical	Mercuric sulfide		Mercuric sulfide		Mercuric sulfide	
Reference	Revis et al., 1989		Revis et al., 1989		Revis et al., 1989	
	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Reference TRV (mg/kg-d)	13.2	NA	13.2	NA	13.2	NA
Test Species UCF	5 (mouse)	5 (mouse)	1 (mouse)	1 (mouse)	1 (mouse)	1 (mouse)
Duration UCF	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)
Endpoint UCF	1 (reprod.)	0.33 (reprod.)	1 (reprod.)	0.33 (reprod.)	1 (reprod.)	0.33 (reprod.)
Total UCF	5	1.66	1	0.33	1	0.33
Final TRV (mg/kg-d)	**2.64**	**7.92**	**13.20**	**40.00**	**13.20**	**40.00**

Final TRV = Reference TRV / Total UCF
NA = not available

9.4.9.3 Birds

Few studies of sufficient quality were located that examined mercury toxicity to avian wildlife. A reproduction study in which Japanese quail were exposed by diet to five dose concentrations of mercuric chloride (2, 4, 6, 8, and 16 mg/kg in feed) for 365 d through a critical reproductive lifestage was conducted by Hill and Schaffner (1976). Conversion of concentrations in diet to a daily dose are based on a body weight of 0.15 kg and a food consumption rate of 0.0169 kg/d (as calculated in Sample *et al.*, 1996). No adverse effects were observed at a dose level of 4 mg/kg in feed (or 0.45 mg/kg-d). At 8 mg/kg in feed (or 0.9 mg/kg/d), adverse effects were observed including a decrease in fertility and hatchability. Since the study considered dietary exposure during reproduction, the 0.45 and 0.9 mg/kg/d doses were considered to be chronic NOAELs and LOAELs, respectively. Mercury TRV derivations for avian recptors of concern are shown in Table 9-17.

Table 9-17. Mercury TRV derivations for avian receptors of concern.

Receptor of concern	Mallard		Red-tailed Hawk	
Study Chemical	Mercuric chloride		Mercuric chloride	
Reference	Hill and Schaffner, 1976		Hill and Schaffner, 1976	
	NOAEL	LOAEL	NOAEL	LOAEL
Reference TRV (mg/kg-d)	0.45	0.9	0.45	0.9
Test Species UCF	5 (J. quail)	5 (J. quail)	5 (J. quail)	5 (J. quail)
Duration UCF	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)
Endpoint UCF	1 (reprod.)	1 (reprod.)	1 (reprod.)	1 (reprod.)
Total UCF	5	5	5	5
Final TRV (mg/kg-d)	**0.09**	**0.18**	**0.09**	**0.18**

Final TRV = Reference TRV / Total UCF
NA = not available

9.4.10 Toxicity Reference Values (TRVs) and Effect Benchmarks for Silver

9.4.10.1 Plants

Data on the phytotoxicity of silver are scarce. Soil-based silver criteria for the protection of plants of 100 mg/kg for most species and 10 mg/kg for sensitive species were proposed by Eisler (1996).

Normal concentrations in terrestrial plants are typically around 0.01 to 16 mg/kg (Horovitz, 1974 in Adriano, 1986). Bush beans were exposed to silver nitrate in solution culture and the plants exhbited no toxicity even though the leaves, stems, and roots contained 5.8, 5.1, and 1,760 mg/kg (dry weight), respectively (Wallace *et al.*, 1977). Another study noted that 4 mg/kg was a critical level in foliage tissues of barley exposed to silver nitrate in solution culture (Davis *et al.*, 1978). Thus, the tissue-based benchmark levels for cattails and buttonbush are 1,760 mg/kg and 4 mg/kg, respectively.

9.4.10.2 Mammals

The effects of silver in wildlife, particularly terrestrial mammals, have not been well documented. Reported effects include renal dysfunction, alterations in copper metabolism and distribution, decreased weight gain, and mortality.

A study in which rats were exposed to a 0.25% solution of silver nitrate (1,587 mg/L in water or a daily dose of 222.2 mg/kg/d) for approximately 250 d was conducted by Matuk *et al.*, (1981, as cited in ATSDR 1990). Adverse effects were observed at this dose level for weight gain and mortality. The LOAEL was adjusted for the endpoint chosen since reproductive effects could occur at lower concentrations than those that affect mortality and weight gain (divided by an uncertainty factor of three). The resulting chronic LOAEL is 74.1 mg/kg/d. A chronic NOAEL of 24.7 mg/kg/d was estimated from the chronic LOAEL by applying an uncertainty factor of three. Silver TRV derivations for mammalian receptors of concern are shown in Table 9-18.

Table 9-18. Silver TRV derivations for mammalian receptors of concern.

Receptor of concern	White-tailed deer		Meadow Vole		Muskrat	
Study Chemical	Silver nitrate		Silver nitrate		Silver nitrate	
Reference	Matuk et al., 1981		Matuk et al., 1981		Matuk et al., 1981	
	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Reference TRV (mg/kg-d)	NA	222	NA	222	NA	222
Test Species UCF	5 (rat)	5 (rat)	1 (rat)	1 (rat)	1 (rat)	1 (rat)
Duration UCF	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)
Endpoint UCF	9 (mortality / body wt.)	3 (mortality / body wt.)	9 (mortality / body wt.)	3 (mortality / body wt.)	9 (mortality / body wt.)	3 (mortality / body wt.)
Total UCF	45	15	9	3	9	3
Final TRV (mg/kg-d)	**4.93**	**14.80**	**24.67**	**74.00**	**24.67**	**74.00**

Final TRV = Reference TRV / Total UCF
NA = not available

9.4.10.3 Birds

A study in which 4-week old mallard ducklings were exposed by diet to eight dose concentrations of silver acetate (25, 50, 100, 250, 500, 1000, 1500, and 3000 mg/kg in feed) for 28 d was conducted by Van Vleet (1982). The endpoints that were assessed in this study were myopathy and mortality induced by a selenium-vitamin E deficiency that is caused by a variety of metals. Conversion of concentrations in diet to a daily dose are based on a body weight of 0.6 kg for 4-week old mallard ducklings and a food consumption rate of 0.1 kg/d for 4-week old mallard ducklings (Heinz *et al.*, 1988). No adverse effects were observed at a dose level of 25 mg/kg in feed (or 4.2 mg/kg-d). At 50 mg/kg in feed (or 8.3 mg/kg/d), adverse effects were observed including an increase in a selenium-vitamin E deficiency induced-myopathy. However, since the study considered dietary exposure under subchronic conditions, the doses were divided by a subchronic to chronic conversion factor of 3 to derive chronic NOAELs and LOAELs of 1.4 and 2.8 mg/kg/d, respectively. Silver TRV derivations for avian receptors of concern are shown in Table 9-19.

Table 9-19. Silver TRV derivations for avian receptors of concern.

Receptor of concern	Mallard		Red-tailed Hawk	
Study Chemical	Silver acetate		Silver acetate	
Reference	Van Vleet, 1982		Van Vleet, 1982	
	NOAEL	LOAEL	NOAEL	LOAEL
Reference TRV (mg/kg-d)	4.2	8.3	4.2	8.3
Test Species UCF	1 (mallard)	1 (mallard)	5 (mallard)	5 (mallard)
Duration UCF	10 (subchronic	10 (subchronic	10 (subchronic	10 (subchronic)
Endpoint UCF	1 (mortality)	1 (mortality)	1 (mortality)	1 (mortality)
Total UCF	10	10	50	50
Final TRV (mg/kg-d)	**0.42**	**0.83**	**0.08**	**0.17**

Final TRV = Reference TRV / Total UCF
NA = not available

9.4.11 Toxicity Reference Values (TRVs) and Effect Benchmarks for Tin

9.4.11.1 Plants

Only one study was identified that examined the phytotoxicity of tin in soil at more than one concentration. Romney *et al.* (1975) studied the effect of tin chloride on shoot weight of bush beans grown for 17 d in soil (pH 6). Shoot weight was reduced 22% by 500 mg/kg tin, while 50 mg/kg had no effect. Thus, the low and high soil-based benchmarks are 50 and 500 mg/kg, respectively.

Normal concentrations in terrestrial plants are typically around 0.1-2 mg/kg (Adriano, 1986). While there are no tissue-based toxicity values available for tin, a screening level benchmark is set to 2 mg/kg in cattail roots and buttonbush seeds.

9.4.11.2 Mammals

No studies were located that examined tin toxicity to mammalian wildlife.

9.4.11.3 Birds

No studies were located that examined tin toxicity to avian wildlife.

9.4.12 Toxicity Reference Values (TRVs) and Effect Benchmarks for Vanadium

9.4.12.1 Plants

No studies were identified that investigated the potential phytotoxic effects of vanadium.

Few studies were identified that evaluated the soil-based phytotoxic effects of vanadium. Concentrations of vanadium 150 mg/kg in soil has been shown to have no effects on rice seedlings while 500 mg/kg has been shown to produce toxicity (Pratt 1966). Based on this study, the soil-based low and high phytotoxicity benchmarks for vanadium are 150 and 500 mg/kg.

Normal concentrations of vanadium in terrestrial plants are between 0.27-4.2 mg/kg (dry weight) (Adriano, 1986). The tissue based benchmark level for cattails is 170 mg/kg, based on the critical value for soybean roots (Pratt, 1966). The tissue based benchmark for buttonbush is 2 mg/kg based on the critical value for soybean tops (Pratt, 1966).

9.4.12.2 Mammals

The effects of vanadium in wildlife, particularly terrestrial mammals, have not been well documented.

A reproduction study in which Sprague-Dawley rats were exposed by oral intubation to three dose concentrations of sodium metavanadate (5, 10, and 20 mg/kg/d) for more than 60 d through several critical reproductive lifestages including 14 d prior to mating for females (60 d for males), gestation, parturition, and lactation was conducted by Domingo *et al.*, (1986). Conversion of concentrations in diet to a daily dose were not necessary because this information was provided by the authors of the study. Adverse effects were observed at the lowest dose level of 5 mg/kg/d. Therefore, the lowest dose was considered as the chronic LOAEL. The chronic NOAEL of 1.67 mg/kg/d was estimated by dividing the chronic LOAEL by an uncertainty factor of 3. Vanadium TRV derivations for mammalian receptors of concern are shown in table 9-20.

Table 9-20. Vanadium TRV derivations for mammalian receptors of concern.

Receptor of concern	White-tailed deer		Meadow Vole		Muskrat	
Study Chemical Reference	Sodium metavanadate Domingo et al., 1986		Sodium metavanadate Domingo et al., 1986		Sodium metavanadate Domingo et al., 1986	
	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Reference TRV (mg/kg-d)	NA	5	NA	5	NA	5
Test Species UCF	5 (rat)	5 (rat)	1 (rat)	1 (rat)	1 (rat)	1 (rat)
Duration UCF	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)
Endpoint UCF	3 (reprod.)	1 (reprod.)	3 (reprod.)	1 (reprod.)	3 (reprod.)	1 (reprod.)
Total UCF	15	5	3	1	3	1
Final TRV (mg/kg-d)	**0.33**	**1.00**	**1.67**	**5.00**	**1.67**	**5.00**

Final TRV = Reference TRV / Total UCF
NA= not available

9.4.12.3 Birds

A study in which mallard ducks were exposed by diet to three dose concentrations of vanadyl sulfate (2.84, 10.36, and 110 mg/kg in feed) for 84 d was conducted by White and Dieter (1978). Conversion of concentrations in diet to a daily dose are based on a body weight of 1.17 kg and a food consumption rate of 0.121 kg/d (from study). No adverse effects were observed at any dose level. Therefore the maximum dose level of 110 mg/kg in feed (or 11.4 mg/kg/d) was considered to be the chronic NOAEL. The chronic LOAEL of 34.2 was estimated by multiplying the chronic NOAEL by an uncertainty factor of three. Vanadium TRV derivations for avian receptors of concern are shown in Table 9-21.

Table 9-21. Vanadium TRV derivations for avian receptors of concern.

Receptor of concern	Mallard		Red-tailed Hawk	
Study Chemical	Vanadyl sulfate		Vanadyl sulfate	
Reference	White and Dieter, 1978		White and Dieter, 1978	
	NOAEL	LOAEL	NOAEL	LOAEL
Reference TRV (mg/kg-d)	11.4	NA	11.4	NA
Test Species UCF	1 (mallard)	1 (mallard)	5 (mallard)	5 (mallard)
Duration UCF	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)
Endpoint UCF	1 (reprod.)	0.33 (reprod.)	1 (reprod.)	0.33
Total UCF	1	0.33	5	1.66
Final TRV (mg/kg-d)	**11.40**	**34.20**	**2.28**	**6.84**

Final TRV = Reference TRV / Total UCF
NA= not available

9.4.13 Toxicity Reference Values (TRVs) and Effect Benchmarks for Zinc

9.4.13.1 Plants

The recent USEPA Draft Ecological Soil Screening Level Guidance contains a soil-based phytotoxic benchmark level of 190 mg/kg dry weight) for zinc to plants. The benchmark is based on nineteen records that were obtained from two papers and four species. All of the toxicity data were based on growth effects, a chronic endpoint. The experiments were conducted with natural soils under conditions of high or relatively high bioavailability.

Normal concentrations of zinc in most terrestrial plants are between 13-119 and 10-334 mg/kg (dry weight) for roots and tops (Adriano, 1986). The tissue based benchmark levels for cattails and buttonbush are 100 mg/kg, based on the lowest number in a range of critical values (range = 100-300 mg/kg) for rice tops (Chino, 1980 in Adriano, 1986).

9.4.13.2 Mammals

A reproductive study in which rats were exposed by diet to two dose concentrations of zinc oxide (2000 and 4000 mg/kg in feed) for 16 d during the gestational period, which is a critical lifestage, was conducted by Schlicker and Cox (1968). Conversion of concentrations in diet to a daily dose are based on a body weight of 0.35 kg and a food consumption rate of 0.028 kg/d (USEPA, 1988). No adverse effects were observed at a dose level of 2000 mg/kg in feed (or 160 mg/kg/d). At an exposure level of 4000 mg/kg in feed (or 320 mg/kg/d), adverse effects were observed including increased rates of resorption and reduced fetal growth rates. Because no adverse effects were observed at 160 mg/kg/d and since the exposure occurred during gestation which is a critical lifestage, this dose was considered a chronic NOAEL. The 320 mg/kg/d exposure level was considered to be the chronic LOAEL. Zinc TRV derivations for mammalian receptors of concern are shown in Table 9-22.

Table 9-22. Zinc TRV derivations for mammalian receptors of concern.

Receptor of concern	White-tailed deer		Meadow Vole		Muskrat	
Study Chemical Reference	Zinc oxide Schlicker and Cox, 1968		Zinc oxide Schlicker and Cox, 1968		Zinc oxide Schlicker and Cox, 1968	
	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Reference TRV (mg/kg-d)	160	320	160	320	160	320
Test Species UCF	5 (rat)	5 (rat)	1 (rat)	1 (rat)	1 (rat)	1 (rat)
Duration UCF	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)
Endpoint UCF	1 (systemic effects)	1 (systemic effects)	1 (systemic effects)	1 (systemic effects)	1 (systemic effects)	1 (systemic effects)
Total UCF	5	5	1	1	1	1
Final TRV (mg/kg-d)	**32.00**	**64.00**	**160.00**	**320.00**	**160.00**	**320.00**

Final TRV = Reference TRV / Total UCF

9.4.13.3 Birds

The literature reviewed included studies examining zinc toxicity to both laboratory birds and more environmentally-relevant species, such as the mallard duck.

A reproductive study in which white leghorn hens were exposed by diet to three dose concentrations of zinc sulfate (20, 200, and 2000 mg/kg supplemented in feed plus 28 already in basal diet to equal 48, 228, and 2028 mg/kg diet) for 308 d during a critical lifestage was conducted by Stahl et al (1990). Conversion of concentrations in diet to a daily dose are based on a body weight of 1.935 kg and a food consumption rate of 0.123 kg/d were determined from the study for the 228 mg/kg dose level. Likewise, a body weight of 1.766 kg and a food consumption rate of 0.114 kg/d were determined from the study for the 2028 mg/kg dose level. No adverse effects were observed at a dose level of 228 mg/kg in feed (or 14.49 mg/kg/d). At an exposure level of 2028 mg/kg in feed (or 130.9 mg/kg/d), adverse effects were observed including reduced egg hatchability. Because no adverse effects were observed at 14.49 mg/kg/d and since the exposure occurred during reproduction which is a critical lifestage, this dose was considered a chronic NOAEL. The 130.9 mg/kg/d exposure level was considered to be the chronic LOAEL.

A 60 d LOAEL of 102.1 mg/kg-d for 7-week-old mallard ducklings exposed to zinc in feed was reported by Gasaway and Buss (1972). This value was calculated by multiplying 3,000 mg Zn/kg feed by the average amount of dry feed consumed daily by exposed mallards (0.034 kg), and dividing by the body weight of an adult mallard (1.0 kg; Terres, 1982) to yield a LOAEL of 102.1 mg/kg-d. The endpoints were based on mortality, food consumption, changes in body weight, and various organ weights. Even though this study was conducted with a more ecologically-relevant species than the chickens used in Stahl *et al.*, (1990), the endpoint from Stahl *et al.*, is more ecologically relevant. Therefore, the chronic NOAEL and LOAEL are from Stahl *et al.*, (1990). Zinc TRV derivations for avian receptors of concern are shown in Table 9-23.

Table 9-23. Zinc TRV derivations for avian receptors of concern.

Receptor of concern	Mallard		Red-tailed Hawk	
Study Chemical	Zinc sulfate		Zinc sulfate	
Reference	Stahl et al., 1990		Stahl et al., 1990	
	NOAEL	LOAEL	NOAEL	LOAEL
Reference TRV (mg/kg-d)	14.5	131	14.5	131
Test Species UCF	5 (chicken)	5 (chicken)	5 (chicken)	5 (chicken)
Duration UCF	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)
Endpoint UCF	1 (reprod.)	1 (reprod.)	1 (reprod.)	1 (reprod.)
Total UCF	5	5	5	5
Final TRV (mg/kg-d)	**2.90**	**26.20**	**2.90**	**26.20**

Final TRV = Reference TRV / Total UCF

9.4.14 Toxicity Reference Values (TRVs) and Effect Benchmarks for Polycyclic Aromatic Hydrocarbons (PAHs)

9.4.14.1 Plants

Acenapthene was used as a surrogate for total PAHs, since no data were identified for phytotoxicity of PAHs. Effects of 75 organic compounds on growth of lettuce from seed for 14 d in two loam soils, and of 1-wk old lettuce seedlings in nutrient solution for 16 to 21 d have been Hulzebos *et al.* (1993). The difference in the loams was the clay content (12 and 24%). The calculated EC50 value for acenaphthene was 25 ppm in the soil. This value of 25 mg/kg was used as the high benchmark and the low benchmark of 2.5 mg/kg was derived by dividing the high benchmark by a factor of 10.

9.4.14.2 Mammals

Data available on the reproductive toxicity of PAHs exposure through diet is limited. For mammalian species, benzo(a)pyrene was determined to be a surrogate for total PAHs since B(a)P is considered to be among the most toxic of the PAHs. A study in which white Swiss mice (CFW strain) were exposed by diet to benzo(a)pyrene (250 mg/kg in diet) for 80-140 d was conducted by Rigdon and Neal (1969). Conversion of concentrations in diet to a daily dose are based on a body weight of 0.03 kg and a food consumption rate of 0.0055 kg/d (USEPA, 1988). At an exposure level of 250 mg/kg in feed (or 45.8 mg/kg/d), adverse effects were observed including an increase in pulmonary adenomas. Because adverse effects were observed at 45.8 mg/kg/d, this dose was considered a chronic LOAEL. The chronic NOAEL of 15.3 was estimated from the chronic LOAEL by dividing by an uncertainty factor of three. PAHs TRV derivations for mammalian receptors of concern are shown in Table 9-24.

Table 9-24. PAHs TRV derivations for mammalian receptors of concern.

Receptor of concern	White-tailed deer		Meadow Vole		Muskrat	
Study Chemical	Benzo(a)-pyrene		Benzo(a)-pyrene		Benzo(a)-pyrene	
Reference	Rigdon and Neal, 1969		Rigdon and Neal, 1969		Rigdon and Neal, 1969	
	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Reference TRV (mg/kg-d)	NA	45.8	NA	45.8	NA	45.8
Test Species UCF	5 (mouse)	5 (mouse)	1 (mouse)	1 (mouse)	1 (mouse)	1 (mouse)
Duration UCF	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)
Endpoint UCF	3 (pulmonary adenoma)	1 (pulmonary adenoma)	3 (pulmonary adenoma)	1 (pulmonary adenoma)	3 (pulmonary adenoma)	1 (pulmonary adenoma)
Total UCF	15	5	3	1	3	1
Final TRV (mg/kg-d)	**3.05**	**9.16**	**15.27**	**45.80**	**15.27**	**45.80**

Final TRV = Reference TRV / Total UCF
NA = not available

9.4.14.3 Birds

For avian species, two relevant studies were found in the literature. In the first study, mallard ducks were exposed to 5 dose concentrations of No. 2 fuel oil (0%, 0.25%, 0.5%, 1.0%, and 1.5% in feed) for 100 d, with the additional stresses of saline drinking water and cold (3^0C) for 50 of the 100 d (Holmes *et al.*, 1979). The authors reported that aromatic hydrocarbons comprised 38% of the No. 2 fuel oil. Therefore, the TRV is calculated based on the amount of aromatic hydrocarbons in the diet. In the second study, mallard ducks were exposed to diets containing either 400 or 4000 mg/kg of aromatic hydrocarbons, including PAHs (Patton and Dieter, 1980) for 210 d. Conversion of concentrations in diet to a daily dose are based on a body weight of 1 kg and a food consumption rate of 0.1 kg/d (Sample *et al.*, 1996). The NOAELs for the two studies were 0.5% No. 2 fuel oil (or 380 mg PAHs/kg/d) for the No. 2 fuel oil study and 400 mg/kg in feed (or 40 mg/kg/d) for the 210 d study. At 1% No. 2 fuel oil (or 380 mg/kg/d), adverse effects were observed including an increase in mortality. At 4000 mg/kg in feed (or 400 mg/kg/d), mild adverse effects were observed including increased liver weight and an increase in hepatic blood flow. Since the No. 2 fuel oil study primarily evaluated mortality and was not carried out for as long as the study by Patton and Dieter (1980), the study by Patton and Dieter (1980) was used as the basis for the chronic NOAEL and LOAEL. It is recognized that the results from these studies are confounded by the unknown contribution of toxicity from chemicals other than PAHs. PAHs TRV derivations for avian receptors of concern are shown in Table 9-25.

Table 9-25. PAHs TRV derivations for avian receptors of concern.

Receptor of concern	Mallard		Red-tailed Hawk	
Study Chemical	Aromatic hydrocarbon mix		Aromatic hydrocarbon mix	
Reference	Patton and Dieter, 1980		Patton and Dieter, 1980	
	NOAEL	LOAEL	NOAEL	LOAEL
Reference TRV (mg/kg-d)	40	400	40	400
Test Species UCF	1 (mallard)	1 (mallard)	5 (mallard)	5 (mallard)
Duration UCF	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)
Endpoint UCF	1 (growth)	1 (growth)	1 (growth)	1 (growth)
Total UCF	1	1	5	5
Final TRV (mg/kg-d)	**40.00**	**400.00**	**8.00**	**80.00**

Final TRV = Reference TRV / Total UCF
NA = not available

9.4.15 Toxicity Reference Values (TRVs) and Effect Benchmarks for Polychlorinated Biphenyls (PCBs)

9.4.15.1 Plants

A few studies have investigated the phytotoxic effects of PCBs. The summaries of these studies (shown below) are from Efroymson *et al.*, (1997). Strek and Weber (1980) investigated the effects of the PCB Aroclor 1254 on fescue, sorghum (*Sorghum bicolor L.*), corn, soybean, and beets grown in a sandy soil (pH 4.7, % organic matter 1, CEC 1.5 meq/100g soil) from seed for 16 d. Height, water use, and top fresh weight of corn, sorghum, and fescue were unaffected by the 1000 ppm test concentration. Fresh top weight of three soybean varieties was reduced an average of 28% and water use 43%. Beet height and fresh top weight were reduced 100% and water use 94%. Fresh foliage weight of pigweed (*Amaranthus retroflexus L.*) was assessed in soil containing up to 100 ppm Aroclor 1254. The more sensitive variety had a 22% reduction in weight at 40 ppm, while 20 ppm had no effect.

The effects of Aroclor 1254 on pigweed grown in the sandy soil used by Strek and Weber in the 1980 work have also been (Strek and Weber 1982). They found a 23% reduction in the height of plants grown from seed for 28 d in soil containing 100 ppm. A treatment level of 50 ppm had no effect.

The effects of Aroclor 1254 on soybean grown in the sandy soil used by Strek and Weber in the 1980 work have been evaluated (Weber and Mrozek 1979). They found a 27% reduction in the fresh shoot weight of plants grown from seed for 26 d in soil containing 100 ppm. A treatment level of 10 ppm had no effect. There was also a 45% reduction in water use at the 100 ppm level.

Thus, based on these studies, the low and high soil-based benchmarks are 40 and 100 mg/kg, respectively.

9.4.15.2 Mammals

A two-generation reproduction study in which Sherman-strain rats were exposed by diet to five doses of Aroclor 1254 (0, 1, 5, 20, and 100 mg/kg in feed) for up to 274 d through a critical reproductive lifestage was conducted (Linder et al 1974). Conversion of concentrations in diet to a daily dose were not necessary because this information was provided in the study. No adverse effects were observed at a dose level of 5 mg/kg in feed (or 0.32 mg/kg-d). At 20 mg/kg in feed (or 1.5 mg/kg/d), adverse effects were observed including a reduction in litter size. Since the study considered dietary exposure during reproduction, the 0.32 and 1.5 mg/kg/d doses were considered to be chronic NOAELs and LOAELs, respectively. PCBs TRV derivations for mammalian receptors of concern are shown in Table 9-26.

Table 9-26. PCBs TRV derivations for mammalian receptors of concern.

Receptor of concern	White-tailed deer		Meadow Vole		Muskrat	
Study Chemical	Aroclor 1254		Aroclor 1254		Aroclor 1254	
Reference	Linder et al., 1974		Linder et al., 1974		Linder et al., 1974	
	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Reference TRV (mg/kg-d)	0.32	1.5	0.32	1.5	0.32	1.5
Test Species UCF	5 (rat)	5 (rat)	1 (rat)	1 (rat)	1 (rat)	1 (rat)
Duration UCF	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)
Endpoint UCF	1 (systemic effects)	1 (systemic effects)	1 (systemic effects)	1 (systemic effects)	1 (systemic effects)	1 (systemic effects)
Total UCF	5	5	1	1	1	1
Final TRV (mg/kg-d)	**0.06**	**0.30**	**0.32**	**1.50**	**0.32**	**1.50**

Final TRV = Reference TRV / Total UCF
NA = not available

9.4.15.3 Birds

A reproduction study in which penned pheasants were exposed by gelatin capsule to three doses of Aroclor 1254 (0, 12.5, and 50 mg/week) once per week for 112 d through a critical reproductive lifestage was conducted (Dahlgren et al 1972). Conversion of concentrations in diet to a daily dose are based on a body weight of 1 kg (as calculated in USEPA's Great Lakes Water Quality Initiative Criteria Documents for the Protection of Wildlife, 1995). No adverse effects were observed on egg fertility or chick growth at either dose level. At 12.5 mg/week (or 1.8 mg/kg/d), adverse effects were observed including a decrease in egg hatchability. Since the study considered dietary exposure during reproduction, the 1.8 mg/kg/d dose was considered to be a chronic LOAEL. A chronic NOAEL of 0.6 mg/kg/d was estimated by dividing the chronic LOAEL by an uncertainty factor of three. PCBs TRV derivations for avian receptors of concern are shown in Table 9-27.

Table 9-27. PCBs TRV derivations for avian receptors of concern.

Receptor of concern	Mallard		Red-tailed Hawk	
Study Chemical	Aroclor 1254		Aroclor 1254	
Reference	Dahlgren et al., 1972		Dahlgren et al., 1972	
	NOAEL	LOAEL	NOAEL	LOAEL
Reference TRV (mg/kg-d)	NA	1.8	NA	1.8
Test Species UCF	5 (pheasant)	5 (pheasant)	5 (pheasant)	5 (pheasant)
Duration UCF	1 (chronic)	1 (chronic)	1 (chronic)	1 (chronic)
Endpoint UCF	3 (reprod.)	1 (reprod.)	3 (reprod.)	1 (reprod.)
Total UCF	15	5	15	5
Final TRV (mg/kg-d)	**0.12**	**0.36**	**0.12**	**0.36**

Final TRV = Reference TRV / Total UCF
NA = not available