8 The Treated Wood Environment

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The purpose of this chapter is to provide default values for environmental inputs when using the model presented in Chapter 9. These defaults are ball-park values useful only for making screening assessments when they are required. Large projects and those proposed for waters where threatened or endangered species live should be based on site-specific measurements. The chapter begins with a review of background concentrations of the following elements and compounds: polycyclic aromatic hydrocarbons (PAH), pentachlorophenol, copper, chromium, arsenic, and zinc; followed by discussions of pH; hardness; inputs for the Biotic Ligand Model; sediment total organic carbon (TOC); sediment redox potential and depth of the reduction oxidation potential discontinuity; and a summary of recommended default values.

8.1 POLYCYCLIC AROMATIC HYDROCARBONS

Polycyclic aromatic hydrocarbons are formed by a variety of processes, including indirect and direct biosynthesis, fossil fuel production and distribution, and incomplete combustion of organic matter (Neff 1979, Eisler 1987). Once formed, PAH can be transported into an aquatic environment by a number of pathways, including atmospheric deposition, fossil fuel distribution, stormwater runoff, and sewage effluent.

8.1.1 Biosynthesis of PAH

Indirect biosynthesis of PAH occurs when extended quinones and related polycyclic materials (mostly plant and animal pigments) are exposed to the reducing conditions found in anoxic sediments. The resulting PAH tend to accumulate in the sediments where they were formed. De novo biosynthesis of PAHs by aerobic and anaerobic bacteria, fungi, and plants is controversial. However, Mallet

et al. (1972) and Brisou (1972) concluded that both aerobic and anaerobic bacteria can biosynthesize benzo[α]pyrene (BαP) and certain other PAHs using fatty acids, sterols, plant pigments, and aliphatic terpenes as substrates. In most cases where PAH biosynthesis has been reported, accumulation of PAH in the organisms purported to have synthesized them could also have been attributed to uptake of PAH from exogenous sources. In light of all of the literature reviewed, it appears that PAH biosynthesis may occur to a limited extent under special environmental conditions when necessary bacterial growth substrates are present. Eisler (1987) suggested that, on a global scale, biosynthesis annually contributes 5,952,420 pounds of PAH to aquatic environments.

8.1.2 Industrial sources of PAH

Paine et al. (1996) examined sediment PAH concentrations and biological responses associated with an aluminum smelter. Sediment levels of 10,000 mg ΣPAH/kg dw were observed in some samples within one kilometer of the smelter. However, most samples revealed sediment PAH concentrations near 150 mg/kg. Hydrocarbon—including PAH—contamination by industry resulted in significant accumulations of PAH in specific areas such as Elliott Bay, Washington, and the Elizabeth River in Virginia during the 19th century and early part of the 20th century (Huggett et al. 1992). Prior to the 1970s, some creosote-treating plants, such as those in Eagle Harbor, Washington (Malins et al. 1985), discharged creosote-treated wastewater directly into marine waters. These point sources of pollution are now well controlled and most of the ongoing PAH contamination is associated with non-point sources.

8.1.3 Municipal wastewaters

Secondary sewage treatment removes some PAH, but most are released to aquatic environments through STP outfalls. Eisler (1987) noted that untreated, raw sewage

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contains 100 to 500 µg ΣPAH/L and sewage sludge contains 200 to 1,750 µg ΣPAH/L.

8.1.4 Stormwater runoff

Hoffman et al. (1985) noted that stormwater runoff from urban areas and highways accounted for 71% of the high molecular weight PAH (HPAH) and 36% of the total PAH (ΣPAH) loading to Narrangansett Bay. More than 30% of all pyrolytic PAH in the coastal sediments of Washington State are supplied by riverine transport of suspended particulate materials, while direct atmospheric input accounts for a maximum of 10% (Prahl et al. 1984).

8.1.5 Transportation systems

Polycyclic aromatic hydrocarbons are released from many transportation system sources, including asphalt, lubricating oils, and gasoline or diesel engine exhaust. Asphalt contains significant concentrations of PAH, including the known carcinogen, benzo[α]pyrene at 6 mg/kg (Machado et al. 1993). Johnston and Harrison (1984) report that BaP deposition along a United Kingdom motorway was 2.8 µg/m^2-wk. Note that BaP is approximately 0.5% to 2.5% of some PAH mixtures and a direct extrapolation suggests that the total PAH loading along a well-used highway may be 560 µg/m^2-wk. Muench (1992) observed roadside PAH concentrations of 6.6 to 9.8 mg/kg dw.

8.1.5.1 Crankcase oil

Neff (1979) reports that little-used (224 km) motor oil contained 6.4 µg/L BaP, equivalent to nearly 1,280 µg/L ΣPAH. Dunn and Stich (1976) found up to 22,000 µg/L BaP in well-used crankcase oil, which is equivalent to 4,400,000 µg ΣPAH/L.

8.1.5.2 Vehicle exhaust

O'Malley et al. (1996) used molecular abundance and carbon isotope measurements to partition four- and five-ring PAH by source in Saint John's Harbor, Newfoundland. Their mixing model suggested that approximately 50% to 80% of the PAH input to the harbor was of combustion origin, likely dominated by vehicular emissions carried by surface runoff from the city of Saint John's. In addition, direct petroleum-related contributions, dominated by crankcase oil, accounted for the remaining 20% to 50% of the ΣPAH input.

8.1.6 Atmospheric deposition

Combustion of organic matter at temperatures between 400° and 2000° C results in the generation of a wide variety of PAH. Reducing conditions (insufficient oxygen) in the pyrolytic environment favor PAH production. Forest and grass fires, vehicle emissions, industrial processes, home heating, power generation, and petroleum refining release significant amounts of PAH into the atmosphere. These combustion products are subject to chemical- and photo-oxidation. However, their residence time in the atmosphere is long enough to allow wide dispersal, and they are a major source of PAH to terrestrial and aquatic environments. According to Eisler (1987), forest and prairie fires, together with agricultural burning, release nearly 72 million pounds of PAH/y into the atmosphere, which is three times the amount from all other pyrolytic sources combined (24,000,000 lb). Wild and Jones (1995) estimated that the upper 50 cm of freshwater sediments in the United Kingdom contained 17.2 mg ΣPAH/kg dw. The suite of detected PAH was dominated by fluoranthene, pyrene, and higher-molecular-weight compounds (benz[α]anthracene through benzo[g,h,i]perylene). Total annual PAH emissions to the UK atmosphere were estimated 712 mt, of which 80.2 mt were associated with vehicle emissions, 600 mt were from domestic coal combustion, and 18.3 mt were from industrial processing. The remaining 13.5 mt were associated with a variety of lesser contributors. Winter contributions of PAH in coastal areas were higher than summer levels. This was attributed to increased pyrolytic input from the burning of fossil fuels for power generation and heating (Bouloubassi and Saliot 1991). Broman et al. (1990) suggested that primary PAH inputs in the Baltic region were from exhaust emissions associated with automobiles, domestic heating, refuse incineration plants, ships, and aircraft.

8.1.7 Hydrocarbon spills

Peat, coal, and petroleum are relatively rich in complex assemblages of PAH. These compounds reach aquatic environments through surface runoff, in wastewater, and as a result of petroleum spillage. Eisler (1987) estimated that spilled petroleum contributed 374,782,200 pounds of PAH to aquatic environments each year. This source overwhelmed all others in terms of global inputs.

8.1.8 Trends in PAH contamination of surface waters

Van Metre et al. (2000) and Van Metre and Mahler (2005) emphasized the contribution of PAH to lake sediments from vehicle emissions and urban environments in general. The authors studied sediment cores from ten reservoirs

and lakes in six urban centers around the United States. They found that PAH concentrations increased during the first half of the 20th century, peaking in the 1950s. Total PAH concentrations then remained steady or declined until the 1970s, after which sediment ΣPAH concentrations again increased. Surficial (depth not specified) sediment concentrations of ΣPAH ranged from 2.79 mg/kg at White Rock Lake in Texas to 224 mg/kg in Newbridge Pond, New York. The authors emphasized that ΣPAH concentrations in these urban lakes rose quickly in the last two decades of the 20th century and that the trends were better correlated with vehicle miles traveled as opposed to percent urbanization. In that context, they noted that ΣPAH may increase in watersheds outside areas of urban sprawl associated with increased vehicular traffic. Their results are difficult to compare with other studies because they included 19 parental and alkylated PAH rather than the 16 parental PAH generally considered priority pollutants.

8.1.9 Summary of sources

Eisler (1987) summarized PAH loading to aquatic environments from a number of sources. Values are in pounds of PAH per year:

Petroleum spillage	374,782,200
Atmospheric deposition	
(from combustion)	110,230,000
Wastewater	9,700,240
Surface land runoff	6,481,524
Biosynthesis	5,952,420

8.1.10 Observed concentrations of PAH in aquatic environments

As a result of the many natural sources, polycyclic aromatic hydrocarbons have been ubiquitous in aquatic environments for thousands of years. However, significantly increased levels have been recorded in sediments since the turn of the 20th century. Neff (1979) reviewed the distribution of PAH in aquatic environments and found very low (<1–2 µg/L) concentrations in pristine surface waters. HPAH tend to accumulate in sediments, and Neff (1979) recorded low sediment contaminations in pristine areas (< 0.050 mg/kg) and higher concentrations (to 15 mg/kg) in industrialized areas and/or human population centers. Eisler (1987) found a similar distribution, with concentrations of 0.005 to 0.544 mg ΣPAH/kg in pristine areas of Alaska, Africa, and the Amazon Basin. Concentrations in urbanized and industrialized areas increased to 791 mg/kg in the United Kingdom. Similarly, Cerniglia and Heitkamp

Table 8.1 Lake sediment ΣPAH concentrations as a function of landscape class. Data from Van Metre and Mahler (2005).

Site class	ΣPAH in surficial sediments (mg/kg dw)
All lakes	3.4
Dense urban	8.9
Light urban	1.3
Rural reference areas	0.3

(1989) measured sediment concentrations ranging from 0.005 mg ΣPAH/kg for an undeveloped area in Alaska to 1,790 mg/kg at an oil refinery outfall in Southampton, England. Sediment PAH concentrations in other industrialized areas ranged from 0.198 to 232 mg/kg. Wild and Jones (1995) inventoried sources and concentrations of PAH in the United Kingdom and concluded that ΣPAH (for 10 PAH), averaged over the upper 50 cm of freshwater sediment cores, was 17.2 mg ΣPAH/kg dw.

8.1.11 Default background concentrations for PAH

The report of Van Metre and Mahler (2005) is most useful for estimated background sediment concentrations of PAH. Consistent with other reports, these authors observed increasing concentrations of PAH in sediments in recent years. They used sediment cores to reconstruct water quality histories for 38 urban and reference lakes across the United States. The cores were sectioned at 0.5 to 5 cm intervals. They observed downward trends in ΣDDT and ΣPCBs at all locations and upward trends in ΣPAH at 16 of the 38 areas sampled. The authors analyzed sediments for 19 parental and a suite of alkylated PAH. Their results are summarized in Table 8.1 by type of landscape. In the absence of site-specific sediment ΣPAH concentrations, it is recommended that model users input the values given in Table 8.1 based on a general characterization of the area where the project is proposed. Note that for the suite of PAH examined by Van Metre and Mahler (2005), the Probable Effects Concentration was 22.8 mg/kg. Sediment TOC was not reported in their paper.

8.2 PENTACHLOROPHENOL

Historically, pentachlorophenol has been used extensively in agriculture and industry as an insecticide, fungicide, herbicide, algicide, and disinfectant. However, the major commercial application of technical grade pentachlorophenol is in the wood preservation industry. Eisler (1989) reported that 80% of the 23 million kg of technical penta produced annually was used for wood preservation. He

noted that pentachlorophenol is found at concentrations from < 1 to 7.3 µg/L in some British Columbia waters. In general, Eisler's (1989) data infer a correlation between industrial-urban centers and increasing pentachlorophenol concentrations in the water column. The U.S. Department of Agriculture (USDA 1980) noted that pentachlorophenol is, "ubiquitous in aquatic environments and its sources are unclear." Observable concentrations may result from direct contamination, from degradation of other organic compounds, or from chlorination of water. The report noted that circumstantial evidence, including the detection of penta in rainwater, indicated that pentachlorophenol may occasionally be present in ambient air. Matsumoto (1982) reported that many phenolic acids are naturally produced by vascular plants and their detritus. However, he stipulated that no evidence of naturally occurring pentachlorophenol was found in his study of polluted Tokyo River water and pristine river, reservoir, and pond waters on the Bonin Islands. No information was obtained suggesting that penta is biosynthesized and there are no known significant natural sources of pentachlorophenol. However, Lampi et al. (1992) observed up to 0.070 mg pentachlorophenol/kg dry sediment in cores dated back to the 17th century. They concluded that this may have been due to wood burning and aerial transport, because the lake is 5 km from the nearest settlement and tens of kilometers from the nearest industries.

The U.S. Environmental Protection Agency (EPA) Storet database contains 5,413 water and sediment quality cases analyzed for pentachlorophenol. Of these, pentachlorophenol was not detected at or above the quantitation limit in 4,971 cases. Of the 442 cases in which pentachlorophenol concentrations were quantified, 343 cases, all from the Oklahoma Water Resources Board or the Oklahoma Department of Environmental Quality, had identical values of 0.12 µg penta/L. It might be assumed that this is half the detection limit of the method used or some other default value. Due to the uncertainty in interpreting these values, they were not included in the following analysis.

8.2.1 Pentachlorophenol in water

One case with an exceptionally high penta concentration was reported from New Mexico by the National Park Service Water Resources Division. The non-routine water sample contained 1191.5 µg penta/L. Ninety-two samples with quantifiable data are included in the remainder of the database. These samples contained a mean of 0.75 ± 0.12 µg penta/L with a maximum reported value of 2.2 µg/L.

8.2.2 Pentachlorophenol in sediments

Quantifiable concentrations of pentachlorophenol were observed in six cases in the database with a mean of 0.41 ± 0.38 mg penta/kg sediment and a maximum concentration of 0.88 mg/kg.

8.2.3 Recommended default background penta concentrations for water and sediment

- Rural and urban environments: 0.00 µg penta/L and 0.00 mg penta/kg dw sediment

- Industrial environments: 0.75 µg penta/L and 0.41 mg penta/kg dw sediment

8.3 COPPER

8.3.1 Freshwater sediment copper

Shacklette and Boerngen (1984) reported a median soil (20 cm depth) copper concentration of 25 mg/kg with a range of <1 to 700 mg/kg in the United States. Table 8.2 provides freshwater sediment concentrations of copper reported in the U.S. Geological Survey (USGS) National Water Quality Assessment (NAWQA) Program (USGS 2010a). The database contains 1,302 sediment copper results. The mean for all states was 24.9 mg Cu/kg dw. Analysis of variance found that not all of the means were equal. Post-hoc testing using Tukey's Honest Significant Difference Test indicated that concentrations in California (49.3 mg/kg) and Oregon (52.7 mg/kg) were significantly higher than in 16 and 20 states, respectively. Therefore, one could reasonably use default values of 49.3 and 52.7 for California and Oregon baseline values and 24.9 mg/kg dry sediment for all other areas of the United States. However, this approach does not take into account the variability associated with land uses discussed by Horowitz and Stephens (2008). The authors examined trace metal content in baseline sediments (i.e., those without anthropogenic effects) of watersheds dominated by six geologic rock types (continental, sedimentary, intrusive, marine, metamorphic, and volcanic). The analysis suggested that within the context of the study's protocols, "geologic factors had relatively little impact on sediment chemistry."

Horowitz and Stephens (2008) collected a total of 1,013 sediment samples from streams and rivers flowing through a variety of landscapes, including undeveloped, forested, agricultural, and urbanized. Table 8.3 summarizes the results for arsenic, copper, chromium, zinc, and total or-

Table 8.2 Concentrations of copper in fluvial sediments, by state, in the United States (data from the USGS NAWQA Program). N = 1323 (no missing data in dep. var. list).

State	Copper means (mg/kg dw)	Confidence (%) -95.000	+95.000	N	Copper concentration (mg/kg dw) Minimum	Maximum
AL	9.6	6.8	12.4	20	2.0	20.0
AR	17.1	12.4	21.7	21	5.0	50.0
AZ	28.8	20.0	37.6	47	5.0	200.0
CA	49.3	37.1	61.4	75	5.0	300.0
CO	24.1	20.6	27.6	72	2.0	70.0
CT	15.0	-48.5	78.5	2	10.0	20.0
DE	5.0	-20.4	30.4	2	3.0	7.0
FL	27.4	14.5	40.4	41	1.0	102.0
GA	18.4	10.5	26.2	33	2.0	100.0
IA	30.9	21.5	40.3	18	3.0	50.0
ID	27.8	22.2	33.4	30	7.0	70.0
IL	24.4	19.1	29.7	22	7.0	50.0
IN	26.8	14.8	28.9	11	15.0	70.0
KS	25.0	20.0	30.0	33	2.0	70.0
KY	19.5	11.7	27.2	26	5.0	100.0
LA	33.3	12.2	54.3	16	5.0	150.0
MA	15.5	2.1	28.9	6	1.0	30.0
MD	19.9	-1.4	41.1	7	5.0	70.0
ME	27.5	10.7	44.2	13	3.0	100.0
MI	14.8	7.4	22.2	32	1.5	103.0
MN	36.9	-1.1	74.8	37	2.0	700.0
MO	18.7	11.8	25.7	30	5.0	104.0
MS	19.6	13.0	26.2	19	3.0	50.0
MT	29.2	24.0	34.5	64	7.0	150.0
MY	50.0			1	50.0	50.0
NB	14.7	9.3	20.2	19	2.0	30.0
NC	37.2	23.0	51.4	20	3.0	104.0
ND	23.2	17.7	28.8	28	3.0	50.0
NH	12.3	0.9	23.8	3	7.0	150.
NJ	19.1	-2.8	41.1	7	2.0	70.0
NM	21.0	16.7	25.3	59	5.0	100.0
NV	25.2	21.0	29.5	52	5.0	70.0
NY	22.1	14.8	29.4	25	3.0	70.0
OH	28.2	18.3	38.2	13	7.0	70.0
OK	15.9	11.1	20.6	25	3.0	50.0
OR	52.7	36.2	69.1	34	7.0	200.0
PA	37.0	25.5	48.5	16	7.0	70.0
RI	15.0			1	15.0	15.0
SC	42.5	10.4	74.6	10	3.0	104.0
SD	28.5	22.3	34.7	30	7.0	50.0
TN	22.4	16.8	27.9	19	10.0	50.0
TX	14.6	13.1	16.2	119	3.0	50.0
UT	25.8	20.7	31.0	47	7.0	100.0
VA	32.6	14.9	50.2	16	5.0	100.0
VT	18.3	11.2	25.5	3	15.0	20.0
WA	35.9	26.1	45.7	22	15.0	100.0
WI	32.2	16.3	48.1	27	3.0	104.0
WV	22.0	17.8	26.2	10	15.0	30.0
WY	21.3	18.1	24.5	40	7.0	50.0
All groups	26.1	24.4	27.8	1323	1.0	700.0

Table 8.3 Summary of mean trace metal concentrations (mg/kg dw) in fluvial sediments associated with six landscape types. Data from Horowitz and Stephens (2008).

Landscape type	Arsenic (mg/kg)	Copper (mg/kg)	Chromium (mg/kg)	Zinc (mg/kg)	TOC (%)
Rural					
≥ 50% agriculture	8.8	24	65	99	2.7
Cropland	8.8	24	65	97	2.8
Pasture	8.8	26	62	100	2.6
≥ 50% forest	7.9	29	71	130	4.8
≥ 50% rangeland	6.9	24	59	85	1.6
Urban					
≥ 50% urban	13.0	76	97	330	4.4
≥ 90% urban	9.4	68	84	330	4.2
≥ 27 people/km^2	10.0	51	79	200	3.9
≥ 557 people/km^2	12.3	69	88	310	4.5
≥ 221 people/km^2	12.0	68	91	270	4.3
≥ 1145 people/km^2	11.0	70	86	340	4.4

ganic carbon (TOC). The authors concluded that the rural landscapes had little effect on the chemical composition of fine-grained sediments. However, urban landscapes and population density had significant influences on trace metal concentrations in sediments. They also concluded that population density appeared to have a more consistent influence on sediment chemistry than did urban percentage. In other words, densely packed cities with large impermeable surfaces had a greater effect than lower density suburbs with a smaller proportion of impermeable surfaces.

8.3.2 Sediment concentrations of copper in freshwater lakes and reservoirs

Sediments are constantly moving in lotic waters, resulting in the redistribution of sediment-adsorbed contaminants like trace metals. In contrast, sediments tend to be more stable in lakes. Mahler et al. (2006) examined sediment cores from 35 reservoirs and lakes in urban and reference landscapes located throughout the United States. Protocols generally followed those of the NAWQA program, except that sediments were not sieved. The authors reported decreasing trends in sediment metal concentrations at 45% and no change at 43% of the sites between 1970 and 2001. Increasing trends in sediment metal concentrations were observed at only 12% (4 of 35) sites. Table 8.4 summarizes their results for copper, chromium, and zinc. Data for arsenic was not reported in their paper. Bear in mind that concentrations in Tables 8.2 and 8.3 are based

on only the ≤63 μm fraction of sediments, whereas Table 8.4 includes the whole sediment. Assuming that a single sample was analyzed, the sieving procedure used in protocols for Tables 8.2 and 8.3 would result in higher metal concentrations than would be reported in Table 8.4. Given that caveat, a comparison of Tables 8.3 and 8.4 suggests the following:

- Sediment copper concentrations are somewhat higher in lakes compared with streams.

- Chromium is slightly lower in urban lakes than in urban streams and significantly lower in reference lakes compared with rural streams.

- Zinc is about equal in stream and lake sediments.

The concentrations provided in Tables 8.3 (for rivers) and 8.4 (for lakes and ponds) are recommended as default background concentrations, along with a qualitative assessment of a watershed's dominant upstream land-use. It should be noted that NAWQA data is based on trace metal concentrations in the fine-grained portion (i.e., ≤63

Table 8.4 Summary of median trace metal concentrations (mg/kg) in lake sediments associated with three landscape types in the 1990s (data from Mahler et al. 2006).

| Landscape type | Metal (mg/kg) | | |
	Copper	Chromium	Zinc
Dense urban	83	75	343
Light urban	68	65	203
Reference	34	47	134
All landscapes	58	65	216

µm) of stream sediment matrices. Sediment quality criteria are typically based on the total sediment matrix. This is not a significant problem in depositional environments dominated by silts and clays, where trace metals migrating from pressure-treated wood are most likely to accumulate; therefore, this factor should not significantly compromise the use of the recommended default values. In addition, NAWQA protocols call for an aggressive four acid digestion that reportedly releases 95% of the metals present. This total or near total digestion overestimates the bioavailable portion of sedimented metals (see Chapter 5 for a discussion of this issue). When conducting site-specific risk assessments in environments that are semi-depositional (i.e., sediments containing significant quantities of sand), or when a more precise understanding of the risks associated with the use of pressure-treated wood is required, it is recommended that surficial sediment samples (i.e., the top 2.0 cm of the sediment column) be collected and analyzed for the metals of concern, using a strong acid digestion rather than a total acid digestion.

8.3.3 Marine sediment copper concentration

Meador et al. (1994) reported the results of the 1984–1988 Benthic Surveillance Project on the Pacific Coast of the United States. The report concluded that the copper concentration (upper comparison interval) of 26 mg/kg dw in the Nisqually Reach reference location suggested anthropogenic contamination because many of the apparently unaffected sites had lower concentrations. However, a qualitative examination of the data suggests that depending on location, unaffected sediments likely contain

2 to 42 µg Cu/kg dw. Brooks (2007b) evaluated sediment data collected by the British Columbia Ministry of Environment. The database contained 44 reference location surficial sediment samples (upper 2.0 cm) examined for a range of physicochemical endpoints, including copper and zinc (Table 8.5). Consistent with the NOAA data for more southern areas of the United States, the mean (± 95% CI) reference sediment concentration was 14.4 ± 2.8 mg Cu/kg dw, with minimum and maximum values of 1.2 and 45.8 mg Cu/kg dw. Included in Table 8.5 is a matrix of Pearson correlation coefficients demonstrating that zinc and copper were strongly and positively correlated with the proportion of silt and clay (i.e., grain size ≤63 µm). As expected, the percent of total volatile solids (TVS) was significantly and positively correlated with the proportion of fines. Also note that redox potential decreased with increasing fines (and %TVS). Thus, when estimating sediment copper and zinc, users should consider the size distribution of the sediment grain and assume higher background metal concentrations in finer-grained sediments. Figure 8.1 is provided to assist users in developing these estimates. The basis of the regression analysis is the percentage of silt and clay in the reference sediment and the concentrations of zinc and copper in the upper 2.0 cm of the sediment column, after using a strong acid digestion. The coefficients of determination are reasonably high (80% for zinc and 81% for copper). Figure 8.1 includes values calculated for erosional environments dominated by gravel (<20% fines); sandy sediments with 20 to 60% fines; and muddy sediments with >60% fines.

Table 8.6 provides sediment copper and zinc concentrations observed by Brooks (2007a) at 13 sites in California.

Table 8.5 (a) Sediment total volatile solids (TVS), zinc (mg/kg dw), copper (mg/kg dw), and percent silt and clay (mud) in British Columbia reference sediments.

Variable	Valid N	Mean	Confidence (%)		Minimum	Maximum
			-95.000	+95.000		
TVS (%)	44	6.8	4.4	9.2	1.2	45.1
Zn (mmg/g)	44	37.9	32.0	43.8	3.5	90.0
Cu (mmg/g)	44	14.4	11.5	17.2	1.2	45.8
Mud (%)	20	33.0	22.8	43.3	15.6	79.0

Table 8.5 (b) Matrix of Pearson correlation coefficients describing the relationship between the independent and dependent variables. Statistically significant (p < 0.050) correlations are in bold. (Data from Brooks, 2007b.) N = 20.

Variable	Redox (mV)	Zn (mmg/g)	Cu (mmg/g)	Mud (%)
TVS (%)	-0.35	0.39	0.32	0.46
Redox (mV)	1.00	-0.43	-0.28	-0.48
Mud (%)	-0.48	0.89	**0.90**	1.00
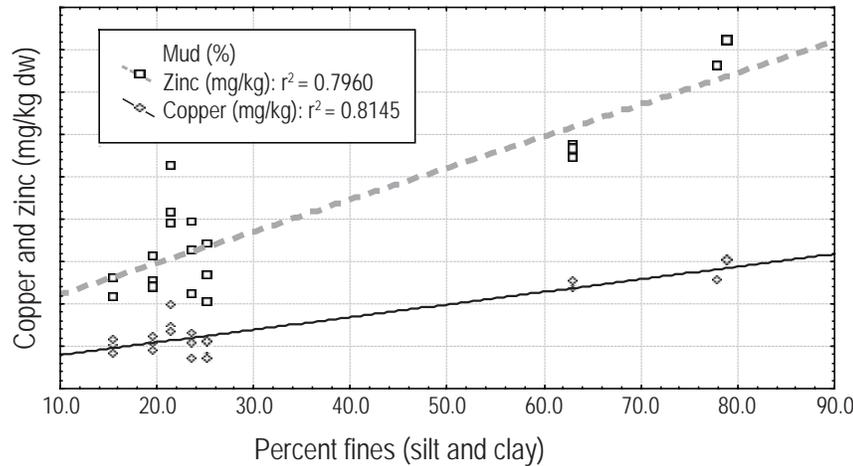
Sediment type	Default copper (mg/kg)	Default zinc (mg/kg)
Erosional (<20% fines)	8	22.2
Sandy (20% to 60% fines)	17	44.7
Muddy (>60% fines)	29	74.7

Figure 8.1 Concentrations of copper and zinc (mg/kg dw) in British Columbia marine reference sediments as a function of the percent silt and clay in the sediment matrix. Data from Brooks (2007b). The included table provides recommended default marine sediment copper and zinc concentrations for erosional, sandy, and muddy sediments. Zinc (mg/kg dw) = 14.7 + 0.75*percent fines. Copper (mg/kg dw) = 5.0 + 0.30*percent fines.

Table 8.6 (a) Sediment redox potential (mV), copper (mg Cu/kg dw), zinc (mg Zn/kg dw), proportion total volatile solids (TVS), and percent silt and clay in sediments collected at 13 California locations. Smallest N for any variable: 61.

Site	N	Means				
		Redox	Sediment copper	Sediment zinc	Proportion TVS	% silt and clay
San Diego Bay	3	20.2	143.7	225.0	0.074	76.01
Hedionda Lagoon	3	-32.4	10.7	41.0	0.014	16.93
Outer Santa Catalina Island	3	45.6	34.3	88.7	0.035	45.37
Dana Point	5	45.7	357.7	267.3	0.051	65.96
San Diego Bay SW Yacht Club	6	21.8	203.4	167.0	0.034	51.23
Mission Bay	7	12.1	210.1	201.7	0.060	57.92
Dana Landing in Mission Bay	6	-64.6	79.4	108.3	0.056	39.99
Channel Islands Harbor	6	-69.3	100.9	137.3	0.050	84.93
Santa Barbara Harbor	7	124.9	10.3	29.9	0.017	47.70
Marina Del Rey	6	-41.8	313.3	384.3	0.075	73.20
Huntington Harbor	33	-3.8	136.3	340.7	0.056	69.99
Inner Santa Catalina Island	33	38.5	25.2	77.1	0.028	43.99
Newport Bay	33	36.9	57.2	138.7	0.046	78.23
All groups	61	13.5	143.2	171.7	0.044	55.75

Table 8.6 (b) Matrix of Pearson correlation coefficients describing the relationship between the independent and dependent variables. Statistically significant (p < 0.05000) correlations are in bold. (Data from Brooks, 2007a). N = 49 (casewise deletion of missing data).

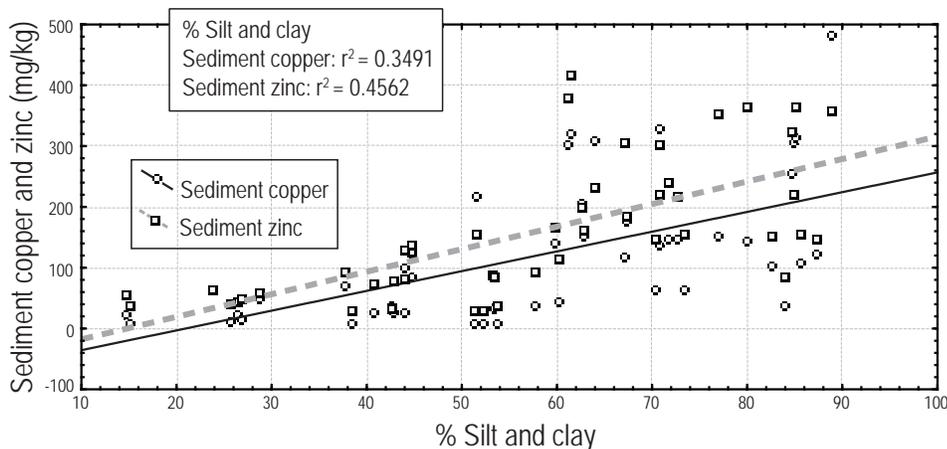| Variable	Sediment copper	Sediment zinc	% TVS	% Silt and clay
Sulfide	0.36	0.34	0.51	0.17
Corr Redox	-0.20	-0.25	-0.32	-0.04
% TVS	**0.68**	0.75	1.00	0.57
% Silt and clay	0.59	0.67	0.57	1.00

*Figure 8.2 Concentrations of copper and zinc (mg/kg dw) at California sites evaluated by Brooks (2007a). Scatterplot (November 2006 database 24v*317c). Sediment copper = -68.0011+3.2468*x. Sediment zinc = -54.7959+3.7062*x.*

San Diego Bay, Dana Point, Mission Bay, Channel Islands Harbor, Marina Del Rey, and Huntington Harbor are marinas where boat hulls are protected with copper antifouling paints and sacrificial zinc anodes, resulting in high concentrations of both metals. Table 8.6b and Figure 8.2 demonstrate the same relationships described above.

8.3.4 Dissolved copper in fresh water

Copper concentrations of 1–10 µg/L were reported by Boyle (1979) from unpolluted U.S. waters; however, concentrations downstream from municipal and industrial outfalls may be higher (Hutchinson 1979). Hem's (1992) review of copper in surface waters suggested that concentrations as high as 10 µg/L can be expected in uncontaminated rivers. Higher concentrations can be expected in areas where surface waters are influenced by leachate from mining activities. For instance, copper concentrations in the Allegheny and Monongahela Rivers in western Pennsylvania may exceed 100 µg Cu/L when flows are low. Brooks (2004) summarized dissolved copper concentrations in reference sections of seven major salmon-producing rivers in the Pacific Northwest (Table 8.7). Figure 8.3 emphasizes the variability in Copper River data. Over 9,000 dissolved copper entries were returned from the USGS NAWQC database for these states (USGS 2010a). Of these, 146 (1.46%) were returned in a query restricted to dissolved copper concentrations ≥2.0 µg/L (Figure 8.4) and, to be conservative, a background default value of 2.0 µg Cu/L is recommended for use when evaluating freshwater projects in rural environments using the model presented in Chapter 9.

8.3.5 Dissolved copper in marine water

Reported concentrations of dissolved copper in marine environments vary between 0.25 µg/L (Chester 1990) and 1.15 µg/L, with a rather broad range of 0.06 to 6.7 µg/L (Schmidt 1978). Copper concentrations in coastal and nearshore waters are typically higher, with a mean of 2.0 µg/L. In nearshore water, more copper was found bound to particulate material (50.7%) than was found in a dissolved form (49.3%) by Schmidt (1978). In open-sea samples, copper was partitioned between particulate (34.8%)

Table 8.7 Dissolved copper concentrations (µg/L) in seven Pacific Northwest salmon-producing rivers.

| River	Dissolved copper (µg/L)	Source
Sacramento River, California	≤11.5	Alpers et al. (2000)
Columbia River, Washington	<3.0; generally 1.7 to 2.2	USGS; Johnson and Hopkins (1994)
Humtullips River, Washington	<30	USGS
Spokane River, Washington	0.7 to 0.8	Pelletier (1994)
Puyallup River, Washington	17	Johnson (1994)
Copper River, Alaska	<23.0; average = 6.44	USGS
Salmon River, Alaska	<60.0; generally 4 to 15	USGS
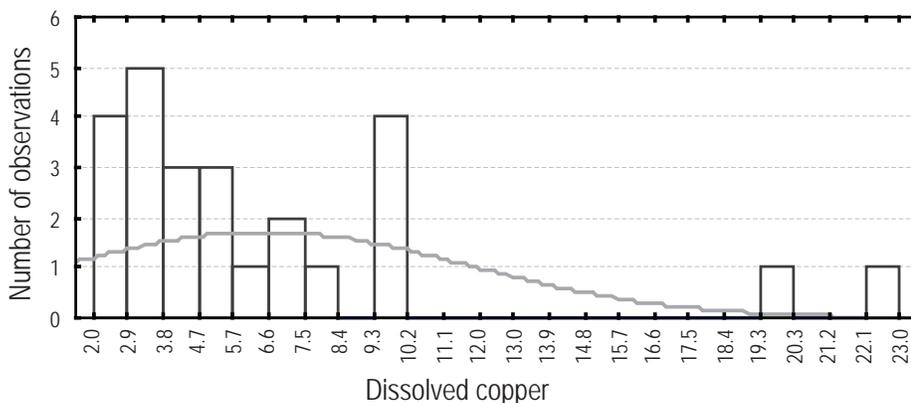
Figure 8.3 USGS NAWQA data describing concentrations of dissolved copper (mg/L) in the Copper River, Alaska, for all years and stations.
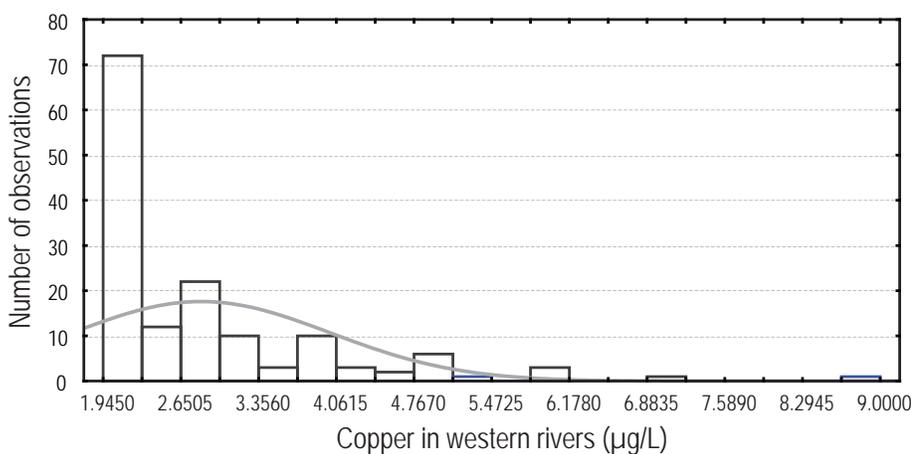

Figure 8.4 Frequency of dissolved copper concentrations > 2.0 µg/L in the USGS NAWQA database for California, Oregon, British Columbia, and Alaska, from 1996 to 2003.

and dissolved (65.2%) compartments. The average concentration of copper in suspended particulate material in the ocean was 109 µg/g, with a range of 52 to 202. Schmidt (1978) noted that these concentrations were higher than those found in most nearshore sediments, and suggested that fine suspended particulates, rich in copper, are probably an important media for transporting continentally derived copper from the near shore to pelagic areas, where the final repository for copper is likely in deep ocean sediments. A default background concentration of 1.0 µg Cu/L is recommended for rural marine environments. A higher value of 2.0 µg/L is recommended for marine waters adjacent to highly urbanized areas; site-specific data should be used when constructing treated wood projects in industrialized areas and marinas or ports where copper-based antifouling paints are used on hulls.

8.3.6 Summary of default copper concentrations for use in the model provided in Chapter 9

Copper is a natural element found in all soils and water on earth. It is a micronutrient supporting most forms of life at low concentrations, but becomes toxic at elevated concentrations. Copper has been used by people for thousands of years and it has a myriad of uses in modern societies. Some of that copper makes its way into surface waters, where the resulting concentrations can have negative effects on biota. Major contributions of copper to surface waters come from mining, urban environments, and industrial activity or from non-point sources. These anthropogenic inputs create significant variability in background copper concentrations, which range from a few

µg/L in pristine environments to several hundred µg/L in association with anthropogenic activity. Because of this, default copper concentrations for use in the model presented in Chapter 9 vary significantly—mostly in association with the degree of human activity in upstream watersheds. For these reasons, there is no single default concentration applicable to surface waters and sediments. Instead, this section of Chapter 8 presents a variety of default values that should reasonably characterize surface waters and sediments in all but heavily industrialized areas, where site-specific data is recommended.

8.4 Chromium

Chromated copper arsenate (CCA-C) is the only preservative currently using chromium, and losses of this metal from wood treated by using WWPI/CITW BMPs are very low. Therefore, additional chromium contamination is not anticipated from the use of pressure-treated wood. Eisler (1986) reported that the earth's crust contains a mean of 125 mg Cr/kg. Hem (1992) gave a mean value of 198 mg/kg in igneous rock. For the United States, Shacklette and Boerngen (1984) reported a median chromium concentration in surficial soils (20 cm below the surface) of 54 mg/kg, with a range of 1 to 2,000 mg/kg. Natural weathering processes release an estimated 32,000 tons of chromium per year. The annual world production is estimated at 7 million metric tons (mmt) and industrial inputs can be significant. Stormwater, sewage treatment plants, metal plating, leather tanning, and mining contribute chromium to aquatic environments. Untreated industrial effluents contain up to 5,000,000 µg Cr/L, and electroplating waste streams contain up to 1,290,000 µg Cr/L.

8.4.1 Chromium concentrations in freshwater sediments

Turekian and Scott (1967) reported suspended loads of chromium in North American rivers ranging from 37,000 to 460,000 µg Cr/L. They estimate that 870 tons of chromium are transported downstream by the Susquehanna River each year. Rivers located east of the Mississippi have higher concentrations than western rivers; however, all of these lotic systems transport significant amounts of chromium to the estuaries at their mouths. Eisler (1986) reported that marsh sediments receiving fertilizers containing sewage sludge for 7 y contained between 2,150 and 4,750 mg Cr/kg dw. Sediments associated with industrial outfalls contained as much as 54,300 mg Cr/kg, while sediments from reference areas contained 50 to 54 mg Cr/kg dw.

Freshwater sediment chromium statistics, by state, are provided in Table 8.8 (USGS 2010a).

As an example of easily available local information, Table 8.9 describes chromium concentrations in stream sediments in the Sacramento Quadrangle, California, available from the USGS National Geochemical Database (USGS 2010b). The freshwater sediment summary presented by Eisler (1986), the USGS NAWQA data presented in Table 8.8 (USGA 2010a), and the USGS NURE data in Table 8.9 (USGS 2010b) suggest that mean freshwater-sediment reference concentrations of chromium range between 50 and 70 mg Cr/kg dw sediment. A default concentration of 60 mg Cr/kg is recommended for use in the model presented in Chapter 9, when site-specific information is not available.

8.4.2 Marine sediment concentrations of chromium

On the Pacific Coast of the United States, Meador et al. (1994) reported that sediments at uncontaminated southern California stations contained between 53 mg/kg outside Mission Bay and 210 mg Cr/kg at the Farallon Islands off San Francisco. Their report summarized other data showing chromium concentrations in deep water to vary between 23.1 and 140 mg/kg. In Washington State, Meador et al. (1994) reported a chromium concentration of 140 mg Cr/kg dw at their Nisqually Reach reference station and lower concentrations varying from 40 mg/kg in the Columbia River estuary to 80 mg/kg in Coos Bay, Oregon. Chromium concentrations in Alaska were generally <90 mg Cr/kg, with higher concentrations reported near Kotzebue, Alaska (200 to 333 mg/kg). A default uncontaminated marine-sediment concentration of 90 mg Cr/kg is recommended.

8.4.3 Dissolved chromium in fresh water

Mean concentrations of chromium in freshwater were reported to be 1.0 to 10 µg Cr/L by Eisler (1986). More detailed data provided by the EPA (1983) reported values ranging from undetectable in some California well water to 2,790 µg Cr/L near a cooling tower. Average values ranged between 4 and 20 µg Cr/L, with a range of <1 to 112 for U.S. surface waters. Hem (1992) reported that chromium concentrations in uncontaminated surface fresh waters of the United States generally fall in the range of 1 to 10 µg/L. Kharkar et al. (1968) reported a mean river-water concentration in the United States of 1.4 µg/L, and

Table 8.8 Concentrations of chromium (mg Cr/kg dw) in fluvial sediments, by state, in the United States (data from the USGS NAWQA Program). N = 1323.

State	Chromium means (mg Cr/kg dw)	Confidence (%)		N	Chromium (mg/kg dw)	
		-95.000%	+95.000%		Minimum	Maximum
AL	30.60	22.6	38.63	20	7.0	70.0
AR	53.10	38.0	68.20	21	10.0	150.0
AZ	57.00	40.6	73.40	47	5.0	300.0
CA	117.73	69.1	166.34	75	10.0	1500.0
CO	42.92	37.6	48.28	72	3.0	100.0
CT	40.00	-87.1	167.06	2	30.0	50.0
DE	30.00	-224.1	284.12	2	10.0	50.0
FL	14.88	6.4	23.36	41	1.0	150.0
GA	36.88	26.5	47.31	33	5.0	100.0
IA	64.72	52.6	76.86	18	15.0	100.0
ID	52.10	41.3	62.90	30	3.0	100.0
IL	48.41	40.9	55.88	22	15.0	70.0
IN	46.82	34.2	59.41	11	15.0	70.0
KS	49.00	40.8	57.15	33	7.0	100.0
KY	116.50	41.8	191.23	26	30.0	1000.0
LA	60.75	43.3	78.22	16	7.0	100.0
MA	39.50	9.9	69.14	6	7.0	70.0
MD	47.86	10.2	85.51	7	15.0	100.0
ME	71.15	53.8	88.52	13	15.0	100.0
MI	25.63	18.7	32.52	32	3.0	70.0
MN	46.68	38.5	54.83	37	2.0	100.0
MO	45.67	38.4	52.90	30	15.0	100.0
MS	53.16	37.5	68.80	19	15.0	150.0
MT	63.28	55.3	71.22	64	15.0	150.0
MY	100.00			1	100.0	100.0
NB	32.47	19.7	45.21	19	3.0	70.0
NC	66.70	33.3	100.11	20	7.0	300.0
ND	53.21	42.5	63.93	28	20.0	100.0
NH	25.00	3.5	64.51	3	15.0	30.0
NJ	42.86	12.2	73.51	7	5.0	100.0
NM	55.49	37.6	73.34	59	5.0	500.0
NV	36.77	29.6	43.94	52	7.0	150.0
NY	33.88	25.4	42.33	25	7.0	100.0
OH	55.00	39.9	70.11	13	15.0	100.0
OK	46.00	35.3	56.71	25	10.0	100.0
OR	121.56	5.0	238.13	34	3.0	2000.0
PA	52.81	36.4	69.24	16	15.0	100.0
RI	50.00			1	50.0	50.0
SC	45.60	15.4	75.78	10	5.0	102.0
SD	58.67	49.7	67.63	30	15.0	100.0
TN	63.68	45.4	81.95	19	30.0	200.0
TX	39.60	34.6	44.63	119	3.0	150.0
UT	45.64	36.9	54.42	47	15.0	150.0
VA	52.25	16.4	92.11	16	7.0	300.0
VT	66.67	-20.6	153.91	3	30.0	100.0
WA	66.59	47.2	58.98	22	15.0	150.0
WI	40.26	29.8	50.68	28	7.0	100.0
WV	46.00	34.7	57.29	10	30.0	70.0
WY	47.87	40.6	55.17	40	15.0	100.0
All groups	54.28	49.6	58.96	1323	1.0	2000.0

Table 8.9 USGS NURE sediment chromium data (mg Cr/kg) for the Sacramento Quandrangle downloaded from the USGS National Geochemical Database (available at http://tin.er.usgs.gov/nure/sediment).

Area	N	Minimum	25th percentile	Median	75th percentile	Maximum
Sacramento, California	234	1	42	67	106.5	2,759
United States	7,451	1	25	39	70	33,800

Durum et al. (1971) found only 11 of 720 U.S. surface water samples exceeding 5 µg/L. A default freshwater chromium concentration of 1.4 µg/L is recommended for uncontaminated sites in the United States.

8.4.4 Dissolved chromium in seawater

Background concentrations of 0.0 to 5 µg Cr/L were reported by Eisler (1986) for uncontaminated seawater. A default value of 3 µg Cr/L is recommended for use in the model presented in Chapter 9.

8.4.5 Chromium summary

Chromated copper arsenate is the only preservative currently using chromium, and losses from wood produced using WWPI/CITW (2002) BMPs are very low (~0.1 µg/cm²-day). This review finds that anthropogenic inputs of chromium can be significant and that determining site-specific sediment concentrations of chromium is recommended when evaluating CCA-C pressure-treated wood projects in industrial areas. In uncontaminated freshwater environments, default values of 1.4 µg dissolved Cr/L and 60 mg Cr/kg dry sediment are recommended. In marine environments, default values of 3 µg Cr/L and 90 mg/kg dry sediment are recommended for use in the model presented in Chapter 9.

8.5 ARSENIC

Clayton and Tanner (1994) reported a median, worldwide soil concentration of 6 mg As/kg, with a typical range of 0.1 to 40 mg/kg. In the conterminous United States, Shacklette and Boerngen (1984) reported a mean soil arsenic concentration of 7.2 mg As/kg, with a range of <0.1 to 97 mg As/kg dw. Natural levels as high as 4,600 mg As/kg have been reported in soils near mineralized veins in British Columbia (Warren et al. 1964). Bhumbla and Keefer (1994) reported average crustal concentrations ranging between 4.1 and 14.4 mg/kg in sandstone and shale. Woolson (1983) discussed the arsenic cycle and suggested that the environmental fate of all arsenic is incorporation into oceanic sediments.

8.5.1 Arsenic concentrations in freshwater sediments

Mean concentrations of arsenic in freshwater sediment, by state, are provided in Table 8.10 using data in the USGS National Water Quality Assessment (USGS 2010a). The mean for the United States is 12.24 mg/kg, with a range of 3.5 to 52.8 mg/kg; however, samples in excess of 100 mg As/kg were reported in 17 states. Robinson and Ayotte (2007) determined arsenic concentration in 1,597 New England stream sediment samples, finding a mean of 5.0, a median of 2.8, and an upper 90th percentile concentration of 11.7 mg As/kg dw. The authors concluded that stream sediment concentrations were well correlated with concentrations in parent rock. They also noted that increased sediment concentrations were found in areas of high historic arsenic use in agriculture. However, sediment concentrations were more dependent on concentrations in parental rock than on agricultural uses. Arsenic levels in freshwater sediment tend to be lower than in terrestrial soils; Table 8.11 summarizes USGS sediment data for rural Okanogan County in Washington State, where the mean of 3.4 mg As/kg is slightly less than Washington State's mean soil concentration of 4.5 mg/kg.

8.5.2 Dissolved arsenic in fresh surface waters

Hem (1992) reported that 79% of the water samples examined by Durum et al. (1971) contained arsenic concentrations below the detection limit of 10 µg/L; however, concentrations >50 µg As/L were reported in 2% of the samples. Hem (1992) also cited Waslenchuk's (1979) report of arsenic concentrations ranging from 0.15 to 0.45 µg/L in streams in the Southeastern United States. In contrast, Saracino-Kirby (2000) reported an average of 42 µg As/L for 188 California surface water samples. However, only 2.0% of the samples were >50 µg/L and only 22% of the samples were >5 µg/L, while about half (46%) of the samples were <3.0 µg/L. Table 8.12 describes metal concentrations in Ferry County, Washington, streams, wells, springs, and one lake. The mean arsenic concentration in streams

Table 8.10 Concentrations of arsenic (mg As/kg dw) in fluvial sediments, by state, in the United States (data from the USGS NAWQA Program). N = 1323.

State	Arsenic means (mg/kg dw)	Confidence -95.000%	+95.000%	N	Arsenic (mg/kg dw) Minimum	Maximum
AL	4.66	3	6.1	20	0.8	10.6
AR	9.67	7	12.1	21	2.6	27.0
AZ	25.48	14	36.8	47	1.4	102.0
CA	10.45	6	15.2	75	0.4	102.0
CO	24.70	16	33.7	72	1.3	102.0
CT	4.10	-1	9.2	2	3.7	4.5
DE	52.84	-584	690.2	2	2.7	103.0
FL	24.25	12	36.9	41	0.2	103.0
GA	13.85	4	24.1	33	0.3	102.0
IA	7.29	6	9.0	18	0.9	13.0
ID	9.58	3	16.3	30	2.7	102.0
IL	7.06	6	8.2	22	2.7	11.0
IN	7.53	5	10.0	11	3.6	15.0
KS	6.79	6	7.8	33	2.4	14.0
KY	25.93	11	41.3	26	2.0	102.0
LA	7.57	4	11.5	16	1.350	32.0
MA	8.55	1	15.8	6	1.980	22.0
MD	17.81	-17	52.2	7	1.1	102.0
ME	9.41	6	12.7	13	1.4	19.0
MI	4.03	3	4.8	32	1.0	10.0
MN	5.50	4	7.0	37	0.5	23.0
MO	7.13	6	8.5	30	1.9	15.0
MS	8.83	6	11.4	19	2.4	21.0
MT	8.77	7	10.4	64	1.3	46.5
MY	5.79			1	5.8	5.8
NB	5.51	4	7.1	19	1.6	12.9
NC	4.76	3	6.5	20	1.0	18.0
ND	6.98	6	8.3	28	2.2	18.0
NH	3.93	-2	9.4	3	1.6	6.0
NJ	19.19	-15	53.5	7	0.6	103.0
NM	12.49	6	18.9	59	1.2	103.0
NV	25.06	15	35.0	52	2.9	102.0
NY	6.99	5	8.8	25	1.5	16.0
OH	12.05	8	15.9	13	5.2	27.0
OK	11.12	3	19.6	25	1.9	103.0
OR	5.07	4	6.1	34	1.1	12.0
PA	12.88	9	17.0	16	3.8	31.0
RI	3.50			1	3.5	3.5
SC	13.70	-9	35.9	10	1.0	102.0
SD	8.51	7	10.2	30	1.7	17.3
TN	27.30	10	44.9	19	3.1	102.0
TX	6.38	6	7.0	119	1.1	18.0
UT	27.99	16	39.6	47	1.5	102.0
VA	5.05	3	7.2	16	0.7	18.4
VT	3.60	-1	8.5	3	2.4	5.9
WA	4.50	3	5.7	22	0.4	8.6
WI	8.02	1	15.5	27	1.4	102.0
WV	8.64	7	10.5	10	5.9	13.0
WY	6.48	5	7.7	40	1.6	19.0
All groups	12.24	11	13.5	1323	0.2	103.0

Table 8.11 Sieved (i.e., <64 µm grain size) sediment arsenic concentrations (mg/kg dw) in rural Okanogan County, Washington streams (http://tin.er.usgs.gov/nure/sediment).

Variable	Valid N	Mean (mg/kg dw)	Confidence		Arsenic load (mg/kg dw)		
			-95.000%	+95.000%	Median	Minimum	Maximum
pH	178	8.38	7.34	9.41	7.9	5.5	8.6
As ppm	482	3.38	2.90	3.86	2.0	1.0	17.0
Chromium ppm	639	33.21	30.32	36.09	25.0	-5.0	550.0
Copper ppm	638	14.70	13.68	15.72	11.0	-2.0	139.0

Table 8.12 Dissolved metal concentrations in surface and ground waters of Ferry County, Washington (http://tin.er.usgs.gov/nure/water). Smallest N for any variable: 85.

Water Type	Means				
	pH	Arsenic (mmg/L)	Chromium (mmg/L)	Copper (mmg/L)	Zinc (mmg/L)
STREAM	7.60	1.41	6.32	5.05	23.22
WELL	7.44	2.50	32.74	33.43	414.42
SPRING	7.36	3.40	12.88	12.23	138.53
LAKE	6.80			7.00	29.00
All groups	7.51	2.26	14.15	12.77	128.70

was 1.41 µg/L. It was higher in wells (2.5 µg/L) and higher yet in springs (3.40 µg/L). These values are consistent with the report of Hem (1992) in suggesting that outside areas of intense agriculture, mining, and industrial landscapes, background concentrations of arsenic in surface water will likely be <3.0 µg/L. In the absence of site-specific data, 3.0 µg/L is recommended as a default freshwater arsenic concentration for the model presented in Chapter 9. However, some caution is advised when assessing projects located in areas having historic volcanic activity, such as the Pacific Northwest. For example, USDA (1980) and Whanger et al. (1977) reported that Oregon spring water typically contains several hundred (133-900) µg/L arsenic, while the amount in lake water is significantly lower (<1 to 9 µg/L). Site-specific water quality data is recommended for areas of intense row-crop agriculture or industry and in association with mines. Site-specific data may be available from USGS, USDA, or the EPA.

8.5.3 Dissolved arsenic in marine water

Andreae (1978) found background concentrations of arsenite (arsenic III) to be less than 0.9 µg/L in seawater and total inorganic arsenic (primarily arsenate or arsenic V) to be less than 1.5 µg/L. Penrose (1974) reported arsenic concentrations of 2.3 to 8.3 µg/L in the Caribbean and 0.8 to 4.5µg/L in the Gulf of Mexico. The EPA (1985) reported arsenic concentrations of 1.5 µg/L in Puget Sound water.

In 1991, Brooks (unpublished data) found arsenic concentrations of 19 µg/L in marine water samples from Little Skookum Inlet, South Puget Sound, adjacent to an upland forest that had been sprayed with an insecticide containing arsenic. Hem noted a consensus among other authors (Waslenchuk 1977, Waslenchuk and Windom 1978, Sanders and Windom 1980) that total arsenic concentrations in marine waters generally range between 1 and 1.5 µg/L, and in marine areas unimpacted by heavy industry, mining, and agriculture, a default value of 1.25 µg As/L is recommended. Site-specific information is recommended in areas suspecting of being contaminated by arsenic.

8.5.4 Arsenic in marine sediments

Woolson (1983) concluded that marine sediments are a likely sink for arsenic derived from uplands. However, the following data suggest that these fluvial inputs may be widely dispersed in the marine environment, resulting in reduced concentrations in relatively pristine environments. The data also indicate that marine sediments influenced by mining and industrial activity may contain elevated arsenic concentrations. For instance, Glass (2002) reported that of 58 surface-water sites in the Cook Inlet basin of Alaska, 50 had concentrations <1.0 µg As/L. The remaining 8 samples had concentrations ranging from 3 to 9 µg dissolved As/L. Sieved (63 µm) freshwater sediment concentrations of arsenic from the same region contained 1.7 to

88 mg As/kg dw (Glass and Frenzel 2001). While there was little anthropogenic influence on these sediments, the authors noted that 47% of the sediment samples exceeded the probable effects level (PEL) of 17 mg As/kg. Valette-Silver and Hameedi (2001) reported mean As concentrations in Beaufort Sea sediments of 34 mg/kg and of 15 mg/kg in the eastern Bering Sea. In Cook Inlet sediment, the authors reported a mean As concentration of 9 mg/kg, which was consistent with the mean value of 8 mg/kg found in Susitna and Copper River sediments. The lower marine sediment concentrations compared with freshwater sediment concentrations in Cook Inlet suggests significant dispersion and dilution in the marine environment. Carpenter (cited in Penrose and Woolson 1974) reported arsenic concentrations of 3 to 15 mg/kg in sediments of Puget Sound and deep sea locations. He also reported concentrations of 290 to 980 mg As/kg dw in sediments near a Puget Sound smelter.

More recently, data in Meador et al. (1994) suggested that uncontaminated marine sediments on the West Coast of the United States contained between 7 and 9 mg As/kg dw. Crecelius et al. (1985) reported concentrations of 15 to 20 mg/kg in industrialized Commencement Bay, Washington, and reported a preindustrial concentration of 7 mg/kg for that area. Likewise, Chapman et al. (1986) reported sediment concentrations in San Francisco Bay of 44 to 70 mg As/kg. San Diego and Santa Monica bays did not receive the fluvial sediment loads associated with placer mining during the 19th century California gold rush—which significantly changed the geology of San Francisco Bay—and had lower mean sediment arsenic concentrations (5.0 to 9.4 mg/kg dw) than San Francisco Bay. The foregoing reports suggest the default concentrations for arsenic in marine sediments on the West Coast of the United States given in Table 8.13. However, it is recommended that site-specific sediment trace metal concentrations be determined for risk assessments in industrialized areas.

Table 8.13 Recommended default marine sediment concentrations of arsenic for use in the risk assessment process described in Chapter 9.

	As (mg/kg dw)
Uncontaminated sediments in rural areas	8
San Francisco Bay sediments	57
Industrialized harbors and marinas	20

8.5.5 Sources of arsenic in aquatic environments

Bhumbla and Keefer (1994) reported the natural transport of 5×10^{11} g of arsenic from land and the atmosphere to earth's oceans. Anthropogenic activities (mining and smelting) add an additional 10% (5×10^{10} g). Bhumbla and Keefer (1994) reported reduced use of arsenic in pesticides. However, they noted that erosional sediments associated with arsenic contamination can contain substantial (100 to 300 mg/kg) total arsenic and acid mine drainage > 2000 mg/kg arsenic. Woolson (1983) suggested that the natural arsenic cycle is not greatly disturbed, on a global basis, by man's activities. He reported that natural sources contribute 7.8×10^{10} g/year. Anthropogenic sources add between 10% and 25% to the total global loading. Iron, steel, lead, zinc, and copper production contribute 82% of the anthropogenic emissions of 2.36×10^{10} g/y. Pesticides account for 2.0×10^{8} g/y, or less than one percent of total anthropogenic input.

8.6 ZINC

Ammoniacal copper zinc arsenate (ACZA) is the only preservative currently using zinc. This preservative was developed to treat western species such as Douglas-fir (*Pseudotsuga menziesii*) for use in freshwater and marine environments. Shacklette and Boerngen (1984) reported a mean zinc concentration of 60 mg Zn/kg dw soil in the conterminous United States, with a range of <5 to 2,900 mg/kg. Like copper, zinc is an essential trace element for most forms of life. It also has many uses in modern society, including sacrificial nodes on marine craft and steel piling; elevated concentrations of this metal are commonly found in marinas and other areas where watercraft are berthed (see Table 8.6a).

8.6.1 Concentrations of zinc in freshwater sediments

Consistent with the many human uses of zinc, Mahler et al. (2006) reported increasing concentrations of zinc in lake sediments across the United States between 1970 and 2001. Table 8.14 provides mean concentrations of zinc in freshwater sediment, by state, in the United States. The mean for the entire database was 65.8 mg Zn/kg dw sediment, with a range of 23 mg/kg in Delaware and New Hampshire to 224.9 in Virginia. Maximum concentrations in the database exceeded 2,000 mg Zn/kg in Colorado. The Utah concentration of 89.7 mg Zn/kg in sediments

Table 8.14 Concentrations of zinc (mg Zn/kg dw) in fluvial sediments, by state, in the United States (data from the USGS NAWQA Program). N = 1323.

State	Zinc means (mg/kg dw)	Confidence -95.000%	Confidence +95.000%	N	Zinc (mg/kg dw) Minimum	Zinc (mg/kg dw) Maximum
AL	40.95	25.1	56.79	20	10.0	102.0
AR	45.00	32.9	57.13	21	10.0	102.0
AZ	64.64	55.1	74.13	47	12.0	150.0
CA	77.63	70.2	85.09	75	20.0	212.0
CO	107.67	50.7	164.68	72	15.0	2080.0
CT	40.00	-23.5	103.53	2	35.0	45.0
DE	23.00	-53.2	99.24	2	17.0	29.0
FL	49.61	36.4	62.79	41	5.0	103.0
GA	63.73	39.0	88.44	33	5.0	400.0
IA	56.67	44.9	68.41	18	19.0	118.0
ID	83.03	65.5	100.57	30	25.0	250.0
IL	68.68	57.0	80.37	22	34.0	140.0
IN	55.55	41.2	69.85	11	32.0	113.0
KS	67.30	52.0	82.60	33	14.0	246.0
KY	48.00	35.2	60.85	26	20.0	150.0
LA	66.75	49.6	83.89	16	20.0	102.0
MA	54.17	9.5	98.83	6	10.0	120.0
MD	39.00	4.3	73.74	7	8.0	113.0
ME	80.38	57.1	103.68	13	10.0	150.0
MI	58.22	48.0	68.46	32	9.0	102.0
MN	58.97	48.1	69.87	37	10.0	177.0
MO	47.13	39.9	54.39	30	25.0	108.0
MS	56.74	42.7	70.78	19	15.0	102.0
MT	68.48	58.9	78.05	64	15.0	306.0
MY	118.00			1	118.0	118.0
NB	54.37	28.8	79.91	19	10.0	220.0
NC	74.20	57.7	90.67	20	25.0	124.0
ND	63.50	51.9	75.11	28	20.0	170.0
NH	23.33	9.0	37.68	3	20.0	30.00
NJ	42.43	6.9	77.97	7	6.0	105.0
NM	47.25	41.7	52.79	59	10.0	110.0
NV	69.40	62.6	76.24	52	25.0	128.0
NY	63.72	52.2	75.24	25	20.0	120.0
OH	68.77	52.8	84.76	13	25.0	110.0
OK	60.72	46.9	74.58	25	15.0	102.0
OR	70.32	53.5	87.19	34	16.0	310.0
PA	80.75	61.3	100.21	16	24.0	155.0
RI	30.00			1	30.0	30.0
SC	94.30	76.9	111.72	10	25.0	102.0
SD	75.43	65.9	84.99	30	31.0	150.0
TN	55.42	42.1	68.73	19	25.0	102.0
TX	40.32	36.0	44.68	119	5.0	108.0
UT	95.77	12.2	179.33	47	20.0	2000.0
VA	224.69	-154.3	603.72	16	13.0	2890.0
VT	43.33	14.6	72.02	3	30.0	50.0
WA	69.00	57.7	80.31	22	30.0	131.0
WI	44.26	32.2	56.34	27	20.0	180.0
WV	60.10	47.2	72.99	10	40.0	98.0
WY	56.47	49.4	63.55	40	20.0	110.0
All groups	65.85	59.6	72.07	1323	5.0	2890.0

Table 8.15 Summary of mean zinc concentrations (mg/Zn kg dw) in fluvial sediments associated with six landscape types. (Data from Horowitz and Stephens 2008.)

Landscape type	Zinc (mg/kg)
Rural	
≥ 50% agriculture	99
Cropland	97
Pasture	100
> 50% forest	130
≥ 50% rangeland	85
Urban	
≥ 50% urban	330
≥ 90% urban	330
≥27 people/km^2	200
≥ 557 people/km^2	310
≥ 221 people/km^2	270
≥ 1145 people/km^2	340

Table 8.16 Recommended default concentrations of zinc for assessing treated wood structures in uncontaminated marine environments. The sediment type can be evaluated qualitatively for screening assessments.

Sediment type	Default zinc (mg/kg)
Erosional (<20% fines)	22.2
Sandy (20% to 60% fines)	44.7
Muddy (>60% fines)	74.7

containing 100% fines compared favorably with the values given for rural landscapes in Table 8.15, which are recommended default concentrations of zinc, by landscape type, in freshwater environments.

8.6.2 Sedimented zinc in marine environments

Figure 8.1 describes zinc concentrations in British Columbia marine reference sediments as a function the proportion of fines (silt and clay) with zinc (mg/kg dw) = 14.7 + 0.75* percent fines. The coefficient of determination was 0.80. Table 8.6 and Figure 8.2 describe similar zinc data for 13 sites in southern California evaluated by Brooks (2007a). Based on these results, recommended default concentrations of marine sedimented zinc in uncontaminated areas are provided in Table 8.16.

Additional information for the West Coast of the United States was provided by Meador et al. (1994), who reported marine sediment zinc concentrations of 22 mg/kg in the Columbia River estuary and 64 mg/kg in Coos Bay, Oregon.

In California, they reported concentrations of 48 mg/kg at the Farallon Islands, 47 mg/kg outside Mission Bay, 44 mg/kg in San Luis Obispo, and 38 mg/kg at Estero Bay. Zinc concentrations in Alaskan sediments were generally higher, ranging from 72 mg/kg at the Endicott Field sites to 160 mg/kg near Oliktok Point. The authors acknowledged the strong correlation between sedimented zinc and sediment grain size. Concentrations near major urban and industrial landscapes were higher, with 310 mg Zn/kg dw in the Oakland, California estuary; 230 to 282 mg/kg in San Diego Bay; 189 mg/kg at Long Beach; and 187 mg/kg in Elliott Bay off Seattle, Washington. These concentrations are similar to the concentrations of zinc reported in freshwater sediments near highly urbanized landscapes (see Table 8.15). Site-specific sediment zinc determinations are recommended in highly urbanized and industrial environments. However, in the absence of site-specific information, a default concentration of 300 mg/kg dw is recommended in urban and industrial marine sediments.

8.6.3 Freshwater concentrations of dissolved zinc

Hem (1992) provided several citations describing zinc concentrations in river water ranging from 5 to 45 µg Zn/L, with a median value in U.S. streams of 20 µg/L (N = 726). He also noted that streams affected by effluent from mining operations may contain concentrations of 100 µg/L. For instance, the Allegheny River in Pennsylvania exceeded 100 µg/L for 5 of 12 mo in 1970, and Pelletier (1994) found zinc concentrations of 80.9 to 164 µg/L in the Spokane River in eastern Washington. On a more site-specific basis, dissolved zinc concentrations for Ferry County, Washington, from the USGS NURE database (summarized in Table 8.17) show a mean surface water concentration (streams and lakes) of ~25 µg/L (USGS 2010b). Some fresh waters have lower dissolved zinc. Median concentrations of 1.0 to 1.8

Table 8.17 Dissolved metal concentrations in surface and ground waters of Ferry County, Washington. (http://tin.er.usgs.gov/nure/water). Smallest N for any variable: 85

Water type	pH means	Zinc (µg/L) means
Stream	7.60	23.22
Well	7.44	414.42
Spring	7.36	138.53
Lake	6.80	29.00
All groups	7.51	128.70

µg/L were reported by Johnson (1994) in four Washington State rivers. The highest metal concentrations occurred during winter and were well correlated with total suspended solids (TSS) and flow rate. Johnson and Hopkins (1991) reported zinc concentrations in unfiltered Lower Columbia River water of <0.33 to 1.7 µg/L. The report of Hem (1992) and the site-specific data provided in Table 8.17 suggest a default dissolved zinc concentration of 25 µg Zn/L for surface waters when using the model presented in Chapter 9. However, these reports indicated a wide range of values (<1.0 to 100 µg/L) and site-specific determinations may be required in areas that are suspected of being contaminated.

8.6.4 Dissolved zinc in seawater

The EPA (1985) reported typical zinc concentrations of 0.50 µg Zn/L in the marine waters of Puget Sound. Eisler (1993) reported zinc concentrations ranging between 0.5 and 2.6 µg Zn/L in uncontaminated seawater. Higher dissolved zinc concentrations of 7 to 26 µg Zn/L were reported for heavily polluted marine waters. A default dissolved zinc concentration of 1.5 µg Zn/L is recommended for evaluating projects located in uncontaminated rural marine environments. Site-specific data is recommended for marine environments adjacent to highly urbanized or industrialized areas. In the absence of site-specific data, a default concentration of 17 µg Zn/L is recommended for these areas.

8.7 pH

Within a range of pH = 5.5 to 8.5, the receiving water pH has a minor to moderate effect on preservative loss from pressure-treated wood. More importantly, it affects the toxicity of contaminants, particularly metals, with increasing toxicity at lower pH. Hem (1992) reported that uncontaminated streams generally have pH in the range of 6.5 to 8.5. However, as plants use carbonic acid during photosynthesis, the pH may increase during the day and cycle downward during periods of reduced sunlight (i.e., at night). Pope et al. (2004) reported mean stream pH values, by state, given in Table 8.18. Overall, median pH in U.S. streams is 7.7 ± 0.15 standard units. However, the data ranged from pH = 2.6 in at least one stream in Montana to 9.1 in Colorado. States with reported low minimum surface water pH (i.e., pH < 5.0) include Colorado (4.6), Florida (3.8), Georgia (3.9), Montana (2.8), New Jersey (4.3), New Mexico (4.6), Oklahoma (2.6), Pennsylvania (4.6), South Dakota (4.8), and Virginia (3.7). However, none of these

Table 8.18 Minimum, median, and maximum pH in streams, by state, reported in Pope et al. (2004).

State	Contributing sites	pH Minimum	pH Median	pH Maximum
Alabama	45	5.1	7.4	8.0
Alaska[a]				
Arizona[a]				
Arkansas	131	6.2	7.3	8.0
California[a]				
Colorado	205	4.6	8.1	9.0
Connecticut	32	7.1	7.4	8.0
Delaware	103	6.5	7.2	8.0
Florida	690	3.8	7.0	8.0
Georgia	284	3.9	7.0	8.0
Hawaii[a]				
Idaho	51	7.1	8.2	9.0
Illinois	11	7.2	7.6	8.0
Indiana	101	7.4	7.8	8.0
Iowa	36	7.9	8.2	9.0
Kansas	247	6.9	8.0	9.0
Kentucky	60	6.6	7.5	8.0
Louisiana	172	5.9	7.1	8.0
Maine[a]				
Maryland	40	7.0	7.8	8.0
Massachusetts[a]				
Michigan	79	7.3	8.0	8.0
Minnesota	66	7.5	8.2	8.0
Mississippi	66	6.0	7.1	8.0
Missouri	38	7.2	7.9	8.0
Montana	276	2.8	7.6	9.0
Nebraska	22	7.9	8.1	8.0
Nevada	35	7.8	8.2	9.0
New Hampshire	55	6.1	7.0	8.0
New Jersey	89	4.3	7.6	8.0
New Mexico	211	4.6	8.2	9.0
New York	19	7.3	7.7	8.0
North Carolina[a]				
North Dakota	72	7.2	8.2	9.0
Ohio	39	7.0	8.0	8.0
Oklahoma	133	2.6	7.5	9.0
Oregon	140	6.8	7.6	9.0
Pennsylvania	174	4.6	7.4	8.0
Rhode Island	5	6.6	6.7	7.0
South Carolina	339	5.4	6.7	8.0
South Dakota	96	4.8	8.2	9.0
Tennessee	43	6.2	7.3	8.0
Texas	546	6.0	7.8	9.0
Utah	64	7.8	8.2	9.0
Vermont[a]				
Virginia	898	3.7	7.2	8.0
Washington	207	6.8	7.6	9.0
West Virginia	24	6.8	7.6	8.0
Wisconsin	38	7.1	7.9	9.0
Wyoming	36	7.6	8.2	9.0

a. Data not presented for states with fewer than nine sites.

Table 8.19 Water pH and temperature in streams, wells, springs, and a lake in Ferry County, Washington. Data from the USGS NURE database. Smallest N for any variable: 1348.

Water type	Temperature means (°C)	Confidence (%) -95.000	+95.000	N	Temperature (°C) Minimum	Maximum	pH means	Confidence (%) -95.000	+95.000	pH Minimum	Maximum
Stream	14.5	14.3	14.7	761	6.0	27.0	7.6	7.6	7.7	5.2	8.7
Well	16.3	15.9	16.7	259	8.0	30.0	7.4	7.4	7.5	6.0	8.7
Spring	15.4	15.0	15.9	327	5.0	30.0	7.4	7.3	7.4	5.3	8.8
Lake	19.0			2	19.0	19.0	6.8			6.8	6.8
All groups	15.1	14.9	15.3	1349	5.0	30.0	7.5	7.5	7.5	5.2	8.8

states had median pH concentrations less than 7.0, suggesting that the low values were restricted to specific areas. Site-specific pH determinations should be made in areas suspected of having pH <5.5—such as acid mine drainages in Colorado, Montana, Pennsylvania, and Virginia or brown waters containing high concentrations of humic acid found in swamps in the Southeast or in the Pine Barrens of New Jersey. On a more site-specific stream or county basis, the USGS NURE database (USGS 2010b) provides high-quality data that is easy to access. For instance, there are 1,364 cases for Ferry County, Washington, describing pH in well water, springs, and streams (Table 8.19). Measurement of pH is relatively easy with properly calibrated field equipment, and site-specific determinations are recommended. However, screening evaluations can rely on local, state, and federal databases, which typically include pH. If site-specific data are not available, the median values provided in Table 8.18 are recommended for purposes of screening projects, unless a project is known to be located in an area of low pH.

8.8 HARDNESS

The amount of dissolved calcium and magnesium in water determines its hardness, which varies throughout the United States. Water hardness has not been shown to significantly affect the loss of active ingredients from pressure-treated wood. However, hardness does mediate the toxicity of contaminants, particularly metals, by competing for binding sites on biotic ligands. Until the Biotic Ligand Model (BLM) is adopted by states, hardness forms the basis of determining freshwater quality criteria for the metals used in wood preservatives, including copper and zinc. Average hardness values, by region in the United States, are provided in Figure 8.5, from the USGS (2010c). The median value in each range is recommended as a default. For instance, along most of the Atlantic Coast and

Table 8.20 Lower and upper bounds for inputs to the Biotic Ligand Model. Worldwide means described by Hem (1992) are provided for perspective.

Parameter	Lower bound	Upper bound	Worldwide means value
Temperature (°C)[a]	10	25	
pH[b]	4.9	9.2	
DOC (mg/L)	0.05	29.65	
Humic acid content (%)	10	60	
Calcium (mg/L)	0.204	120.24	15.0
Magnesium (mg/L)	0.024	51.9	4.1
Sodium (mg/L)	0.16	236.9	6.3
Potassium (mg/L)	0.039	156	
Sulfate (mg/L)	0.096	278.4	11.0
Chloride (mg/L)	0.32	279.72	7.8
Alkalinity (mg/L)	1.99	360	
DIC (mmol/L)	0.056	44.92	
Sulfide (mg/L)	0	0	

a. Temperature: measure in-situ during the season when construction will occur.
b. pH: measure in-situ or use the median values provided by state in Table 8.18.

in the Pacific Northwest, surface water hardness generally falls in a range of 0 to 60 mg/L. A default hardness of 30 mg/L is recommended for these areas.

8.9 INPUTS FOR THE BIOTIC LIGAND MODEL

The range of allowable inputs for the BLM (Di Toro et al. 2001) are provided in Table 8.20, together with the worldwide mean value from Hem (1992). The remainder of this section describes default values when site-specific data is not available.

8.9.1 Dissolved organic carbon

Dissolved organic carbon (DOC) can be measured in-situ or the mean value for streams and lakes given in Table
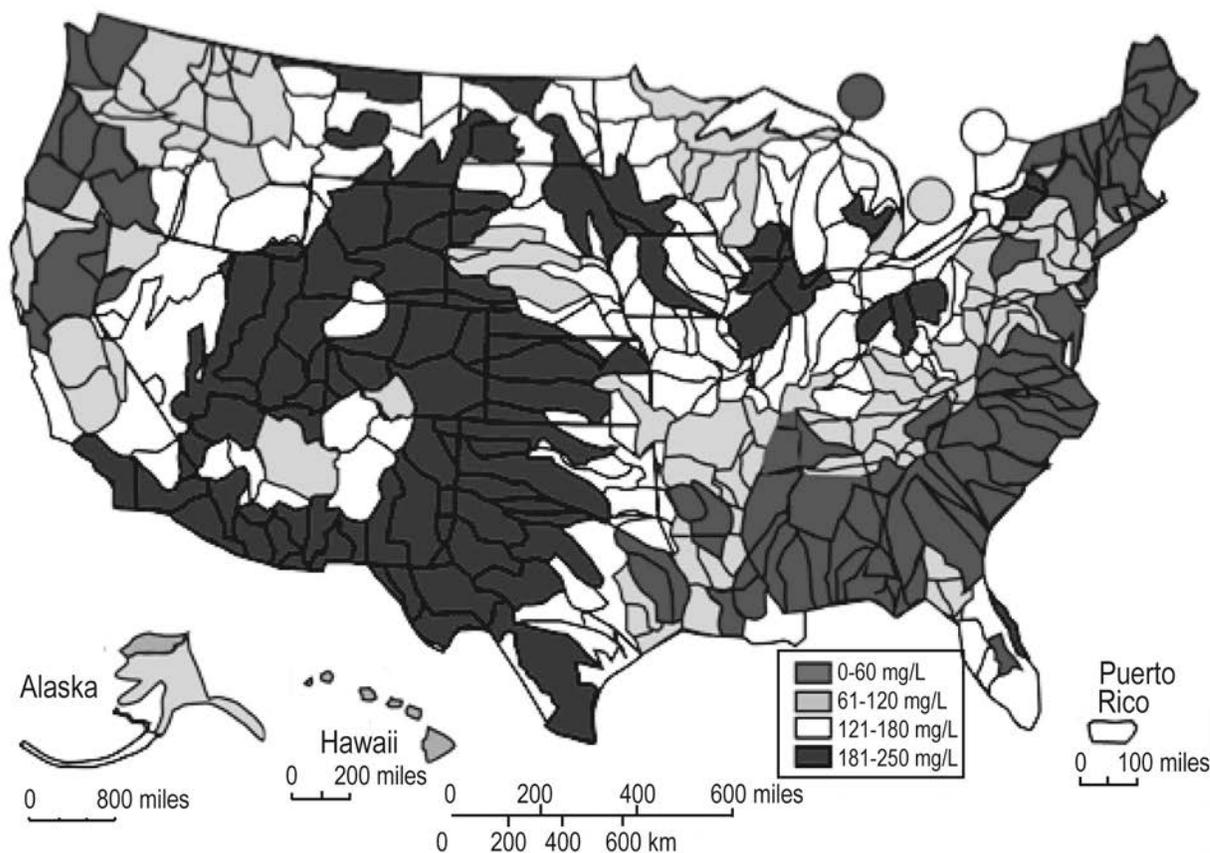
Figure 8.5 Water hardness in the United States measured as mg CaCO$_3$/L. Available at http://water.usgs.gov/owq/hardness-alkalinity.html.

8.21 from DeGraeve (2003) can be employed. This is also available in Appendix C of EPA (2007). For all streams, the mean (± 95% CI) was 7.22 ± 1.37; for lakes it was 5.7 ± 1.6 mg DOC/L. The minimum value for any case in streams was 2.1 mg/L; in lakes, the minimum was 2.3 mg/L. Maximum values were 16.1 and 15.0 mg/L in streams and lakes, respectively. Dissolved organic carbon is a primary determinant for metal toxicity; using the BLM and site-specific determinations are recommended where sensitive species are present. Otherwise, the values provided in Table 8.21 are recommended as defaults.

8.9.2 Humic acid content (%)

Information describing this input was not obtained in this review. For screening purposes, the lower quartile bound of the range of values provided in Table 8.20 is recommended as a default (22.5%). Appendix C to EPA (2007) provides the following linear regression fits to data describing the relationship between required elements in WFTS well water and water hardness (mg CaCO$_3$/L). In the ab-

$$\text{Calcium (mg/L)} = 0.274 + 0.3085*\text{hardness (mg/L); } R^2 = 0.71 \qquad [8.1]$$

$$\text{Magnesium (mg/L)} = 0.5429 + 0.0573*\text{hardness (mg/L); } R^2 = 0.80 \qquad [8.2]$$

$$\text{Sodium (mg/L)} = 3.032 + 0.0713*\text{hardness (mg/L); } R^2 = 0.60 \qquad [8.3]$$

$$\text{Chloride (mg/L)} = 2.7842 + 0.1278*\text{hardness (mg/L); } R^2 = 0.56 \qquad [8.4]$$

$$\text{Potassium (mg/L)} = 0.04332 + 0.0058*\text{hardness (mg/L); } R^2 = 0.12 \qquad [8.5]$$

$$\text{Sulfate (mg/L)} = -3.0430 + 0.2816*\text{hardness (mg/L); } R^2 = 0.80 \qquad [8.6]$$

Table 8.21. Dissolved organic carbon (DOC) described in Appendix C -2 of US EPA (2007).

State	DOC in streams (mg/L)	DOC in lakes (mg/L)
Alabama	4.6	6.4
Alaska	4.6	6.4
Arizona	5.2	4.2
Arkansas	7.2	2.7
California	8.2	2.3
Colorado	8.6	-
Connecticut	4.8	-
Delaware	7.1	-
Florida	16.1	12.1
Georgia	4.3	-
Hawaii	4.0	-
Idaho	3.2	-
Illinois	6.8	4.7
Indiana	9.2	-
Iowa	11.6	-
Kansas	5.2	4.5
Kentucky	3.1	-
Louisiana	6.9	5.6
Maine	15.3	-
Maryland	3.7	-
Massachusetts	5.9	-
Michigan	6.3	2.7
Minnesota	12.2	4.8
Mississippi	-	-
Missouri	4.2	-
Montana	9.4	8.2
Nebraska	6.8	-
Nevada	4.2	-
New Hampshire	4.2	-
New Jersey	5.5	5.0
New Mexico	6.3	5.2
New York	4.0	2.4
North Carolina	11.5	-
North Dakota	14.5	14.9
Ohio	5.0	2.6
Oklahoma	7.7	15.0
Oregon	2.1	4.4
Pennsylvania	5.4	3.2
Rhode Island	8.3	-
South Carolina	5.7	-
South Dakota	7.6	-
Tennessee	2.3	-
Texas	6.5	10.3
Utah	8.9	2.4
Vermont	4.5	-
Virginia	4.7	-
Washington	5.4	2.8
West Virginia	2.8	-
Wisconsin	9.2	4.1
Wyoming	8.2	-

sence of site-specific data, the following relationships are recommended for use in defining default values for each element when using the BLM. Coefficients of determination are provided for each regression.

8.9.3 Dissolved inorganic carbon (DIC in mmol/L)

No information describing dissolved inorganic carbon (DIC) was found in this review; the lower quartile of the range described in Table 8.20 is recommended as a default value for screening assessments (11.3 mmol/L).

8.9.4 Sulfide (mg/L)

Use 0.0 unless site-specific information is available.

8.9.5 Alkalinity (mg/L)

Figure 8.6 provides alkalinity data for the conterminous United States from the USGS National Water Quality Assessment (2010a). In general, alkalinity on the East and West Coasts is relatively low (<81.6 mg $CaCO_3$)/L. Areas in the South, Midwest, and Rocky Mountain regions have higher alkalinity (>164.5 mg/L). Between 10 and 100 mg/L alkalinity does not appear to be as important for assessing metal toxicity as DOC and hardness are. Therefore, for initial screening purposes, a conservative default value of 30 mg/L is recommended. However, the distribution of alkalinity across the United States is quite variable and users are encouraged to use the EPA STORET or USGS NAWQA databases (USGS 2010a) for more specific information when rigorous assessments are required.

8.10 Sediment Total Organic Carbon (TOC)

The proportion or percent of organic matter, total organic carbon (TOC) or total volatile solids (TVS), in sediments is dependent on a number of factors. The supply of organic matter from adjacent upland and upstream aquatic environments is difficult to include in default benchmarks because it varies on small geographic scales (i.e., <100 m). This is particularly true in lotic waters and less true in large marine environments. A second important factor is the depositional nature of the benthos. Lotic freshwaters are often characterized by riffle and pool complexes, with riffles being erosional and pools being depositional. The scale of these features can be several meters to several hundred meters—making development of default values difficult for upland rivers and streams. When assessing
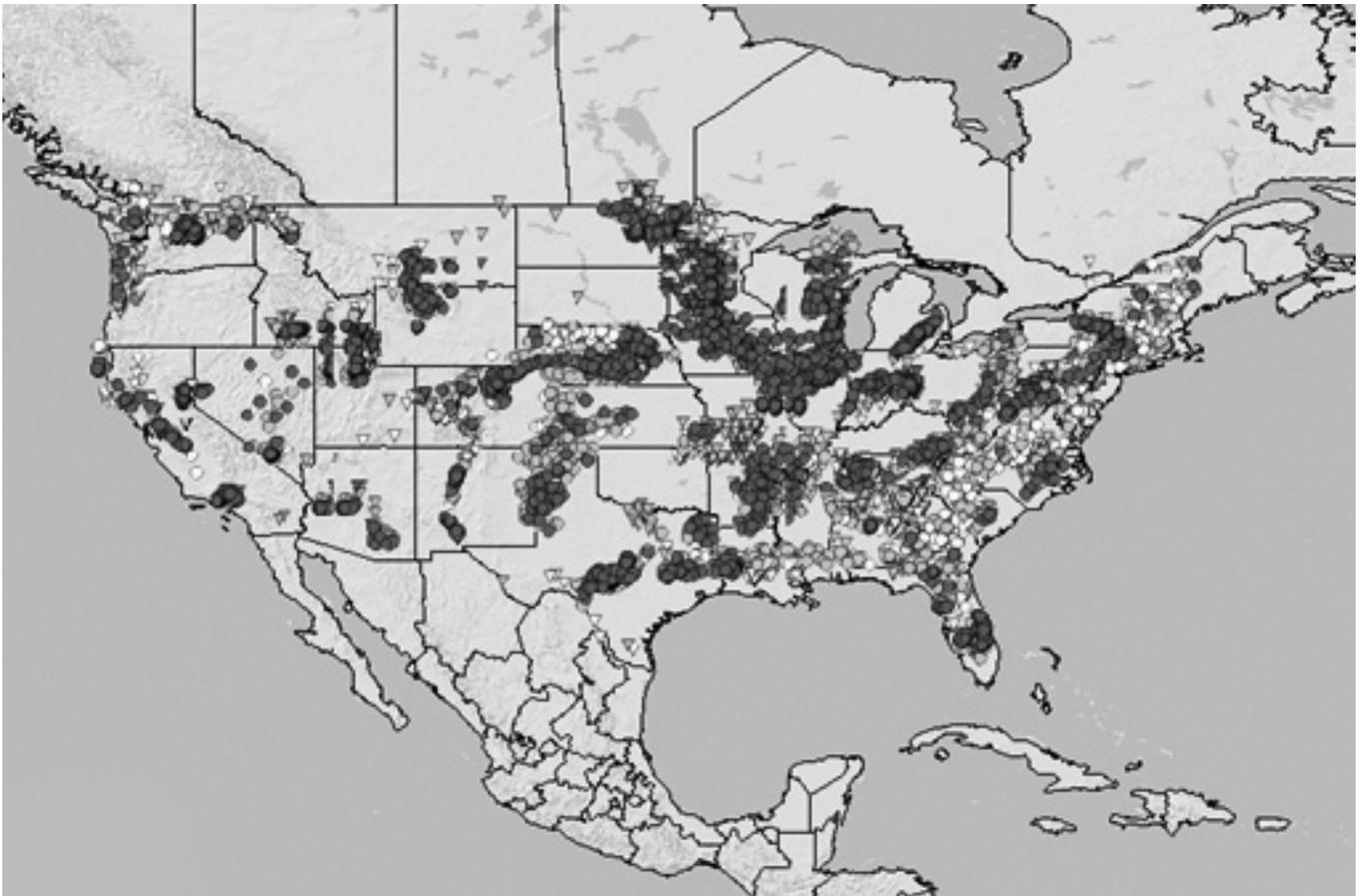
Figure 8.6 Alkalinity data from the USGS NAWQA database available at http://infotrek.er.usgs.gov/nawqa_map.

creosote and pentachlorophenol pressure-treated wood projects in these environments, site-specific determinations of sediment TOC or TVS are recommended. Loss on ignition and TVS are synonymous terms applied to the loss of weight of thoroughly dried sediments when combusted at 550°C. Total organic carbon and TVS are not identical, but are marginally ($R^2 = 0.61$) well correlated, with a reasonable 95% confidence interval (Figure 8.7). However, sediment quality criteria for organic contaminants are generally based on TOC, rather than TVS, and therefore this is the preferred endpoint.

The organic content in marine sediments, particularly those located in protected areas where treated wood structures are often sited, tends to be more predictable. Brooks (2005, 2007b) assessed sediment physicochemical data over broad ranges of sediment types (<5 to >90% silt and clay) in support of assessing the environmental response to finfish and shellfish aquaculture. A subset of the several thousand cases in the databases included same sample sediment grain-size distribution (SGS), TVS and

TOC. As seen in Figure 8.8, the percent TOC in sediments is linearly related to the percent of fines (i.e., silt and clay having a grain size ≤63 μm) with TOC = 0.65 + 0.037*fines; R^2 0.73. The default TOC values provided in Table 8.22 are based on this regression and a quantitative or qualitative assessment of the grain size distribution at the site of the proposed structure. Where sediments are complex, the finest-grained sediments (most depositional area) should be assessed.

8.11 Sediment Redox Potential (mV) or Depth of the Reduction Oxidation Potential Discontinuity (RPD in cm)

Organic matter in surface waters settles to the sediments where it is catabolized by bacteria and detritivores or is buried. The top millimeter of the sediment column is typically aerobic (i.e., positive redox potentials). The distribution of oxygen for aerobic catabolism depends, in large
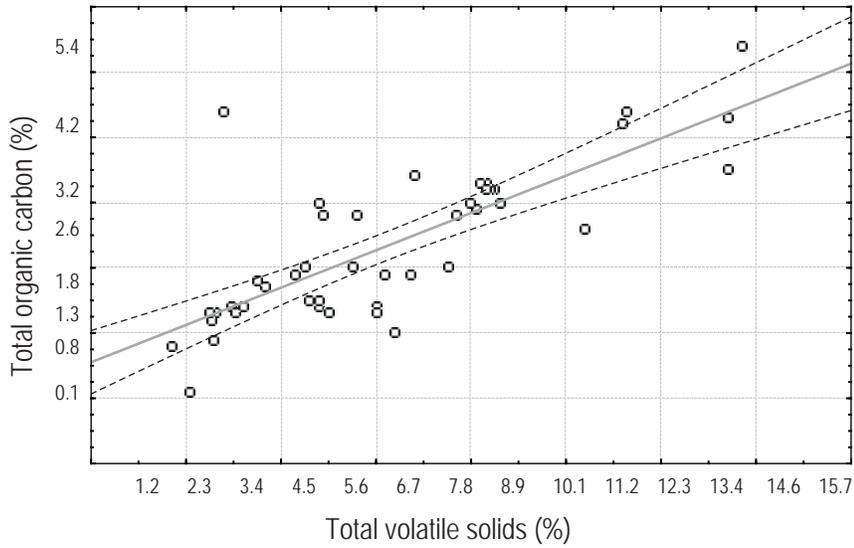
Figure 8.7 Predicted percent total organic carbon (TOC) from sediment percent total volatile solids (TVS) data. Percent TOC = 0.552 + 0.286*Percent TVS. 95% confidence interval provided; $R^2_a = 0.61$.

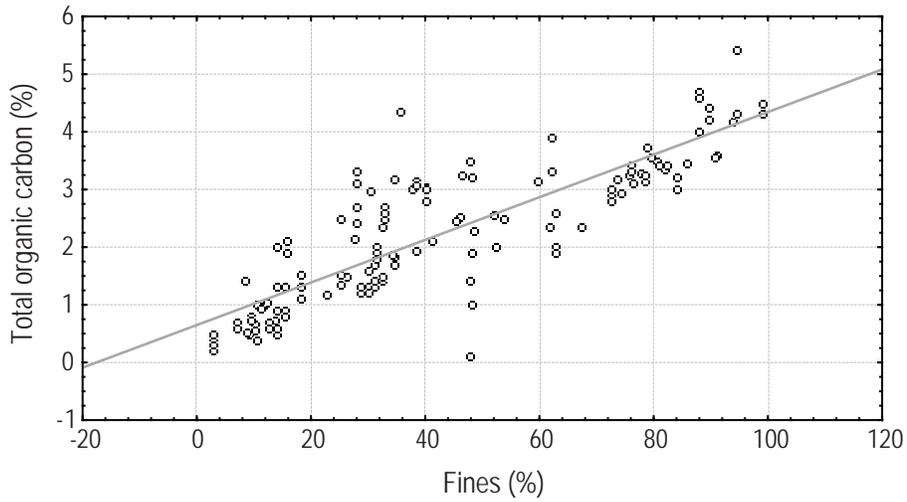*Figure 8.8 Percent TOC in marine sediments as a function of the percent silt and clay (i.e., grain size <63 μm). Percent TOC = 0.65 + 0.037*Percent Fines; $R^2_a = 0.73$.*

Table 8.22 Recommended default percent TOC values for use in screening assessments of treated wood projects proposed in marine environments.

Qualitative	Percent silt and clay	Default TOC (%)
Mud	>60	3.6
Sandy mud	40 to 60	2.5
Sand	20 to 40	1.8
Gravel	<20	1.0

part, on the circulation of surface water containing oxygen within the interstitial spaces between sediment particles. Pore water circulates more freely in coarse sediments than in fine-grained sediments characterized by muds. These fine-grained sediments are characteristic of depositional areas where organic detritus also accumulates. The combination of reduced pore-water circulation and increased organic deposition can lead to oxygen depletion as organic matter is decomposed by bacteria and benthic organisms. At that point, organic carbon is further consumed by facultative anaerobes like *Desulfovibrio* bacteria and marine invertebrates like *Capitella capitata* that tolerate low oxygen and high sulfide concentrations (see Brooks 2001). When pore-water oxygen is depleted, facultative anaerobes strip oxygen from sulfate in sediments, leaving sulfide as a metabolic byproduct. The free sulfide combines with hydrogen to form hydrogen sulfide (H_2S), giving a typical rotten egg smell to these sediments. The free sulfide also binds with iron to form iron sulfides (FeS), which reduce sediment chroma, creating a dull appearance. In some sediment, the color change is marked, forming a color discontinuity between aerobic and anaerobic conditions referred to as the RPD. In dark, basaltic sediments, this discontinuity can be hard to discern, as it is in marginally aerobic conditions. Sediment redox potential measures the oxygen content of sediment in mV, using field meters and probes (see Brooks 2001), and is a more quantitative and precise measure. However, it changes rapidly with depth and can be difficult to measure accurately. In the absence of site-specific information, examine the top 4 cm of the sediment and use the default RPD and redox potentials provided in Table 8.23, based on the sediment's appearance. The values given in Table 8.23 are based on the author's personal experience in evaluating thousands of freshwater and marine sediment samples, and the redox potential and RPD depths are recommended as default values for use in screening assessments. The upper 4 cm of sediments are typically examined because that is where most of the macrofauna reside and because this more recently deposited material is where most contaminants are found.

8.12 Summary of Recommended Default Values

The default values presented in this chapter and summarized in Table 8.24 are intended for use when conducting screening assessments. They are based on average values and should not be used for large projects or for moderately sized projects when particularly sensitive species are involved. In those cases, risk assessors are encouraged to seek site-specific data available in local, state, and federal water quality databases—or to evaluate the endpoints necessary to run the model at the project site during an appropriate time of the year.

Table 8.23 Qualitative assessment of sediment redox potentials and depth of the RPD based on a visual inspection of the top 4 cm of sediment columns. Matrix colors refer to the Munsel (1994) color charts.

Visual indicators of sediment redox	RPD (cm)	Redox potential (mV)
Brightly colored (high chroma and value)	≥4 cm	>100
Matrix color easily identified (chroma >3 or 4 and value >2 or 3)	3 cm	0 to +50
Gray or gleyed sediments with poor matrix color (chroma 2 to 3)	2 cm	-50 to 0
Dark but not black sediments (chroma and value <2)	1 cm	-100 to -50
Dark with little indication of color other than dark gray or black	0 cm	<-100

Table 8.24 Recommended default user inputs for the model presented in Chapter 9 when conducting screening assessments of pressure treated wood projects used in sensitive environments.

User input for the model	Default value(s)
17. Average water temperature (°C)	Site specific measurement
18. Freshwater pH (standard units)	Site specific measurement or use the median values given in Table 8.18 for each state.
19. Water hardness (mg CaCO$_3$/L)	USGS data (see Figure 8.5)
20. Salinity (PSU)	
Freshwater	0.0 PSU
Estuarine	10.0 PSU
Marine	30.0 PSU
21. Background dissolved copper (µg/L). Required only when using waterborne preservatives.	
Freshwater	
Rural and light urban	2.0 µg Cu/L
Dense urban and industrial	Site specific measurement recommended.
Marine	
Rural and light urban	1.0 µg Cu/L
Dense urban	2.0 µg Cu/L
Heavy industry	Site specific measurement recommended.
22. Background dissolved arsenic (µg/L). Required only for CCA-C and ACZA assessments.	
Freshwater	
Rural (without intensive agriculture)	3.0 µg As
Mining and intensive row crop agriculture	Site specific information is recommended.
Marine	
Unless arsenic contamination is suspected	1.25 µg As/L
When arsenic contamination is suspected	Site specific information is recommended.
23. Background dissolved chromium (µg/L). Required only when evaluating CCA-C projects.	
Freshwater	
Rural (uncontaminated) sites	1.4 µg Cr/L
Heavily urbanized or industrial areas.	Site specific information is recommended
Marine (all areas)	3.0 µg Cr/L
24. Background dissolved zinc (µg Zn/L). Required only when evaluating ACZA treated wood projects.	
Freshwater	
Rural (uncontaminated sites)	25.0 µg Zn/L
Highly urbanized or industrial areas	Site specific information is recommended.
Marine	
Rural (uncontaminated sites)	17 µg Zn/L
Highly urbanized or industrial areas	Site specific information is recommended.
25. Background dissolved penta (µg penta/L). Required only for pentachlorophenol-treated projects.	
Rural and urban	0.00 µg penta/L
Industrial areas	0.75 µg penta/L

30. Sediment percent total organic carbon. Required for pentachlorophenol and creosote projects.

Qualitative	Percent silt and clay	Default TOC (%)
Mud	>60	3.6
Sandy mud	40 to 60	2.5
Sand	20 to 40	1.8
Gravel	<20	1.0

31. Sediment density (g/cm^3)   2.6 g/cm^3. Required for all project assessments.

32. Sediment redox (mV) or RPD (cm) for pentachlorophenol (redox) and creosote (RPD) projects. In the absence of site specific measurements, these default values are based on a qualitative (visual and olfactory) evaluation of the sediments.

Visual indicators of sediment redox	RPD (cm)	Redox potential (mV)

Table 8.24, *continued*

User input for the model	Default value(s)	
Brightly colored (high chroma and value)	≥4 cm	>100 mV
Matrix color easily identified (chroma >3 or 4 and value >2 or 3)	3 cm	0 to +50 mV
Gray or gleyed sediments with poor matrix color (chroma 2 to 3)	2 cm	-50 to 0 mV
Dark but not black sediments (chroma and value <2)	1 cm	-100 to -50 mV
Dark with little indication of color other than dark gray or black	0 cm	<-100 mV

33. Background sediment copper (mg Cu/kg dw). See Tables 8.3 and 8.4.

Rivers and streams		
Dense urban	70 mg Cu/kg dw	
Light urban	51 mg Cu/kg dw	
Rural	25 mg Cu/kg dw	
Lakes		
Dense urban	83 mg Cu/kg dw	
Light urban	68 mg Cu/kg dw	
Rural	58 mg Cu/kg dw	
Marine		
Rural >60% mud	29 mg Cu/kg dw	
Rural 20 to 60% mud	17 mg Cu/kg dw	
Rural <20% mud	8 mg Cu/kg dw	
Dense urban or industrial	Site specific measurement recommended.	

34. Background sediment arsenic (mg As/kg dw)

Freshwater	Use the mean values given in Table 8.10.	
Marine		
Rural	8 mg As/kg	
San Francisco Bay	57 mg As/kg	
Other industrialized harbors and marinas	20 mg As/kg	

35. Background sediment chromium (mg Cr/kg dw). Site specific information recommended in areas of suspected industrial contamination.

Freshwater	In uncontaminated areas use 60 mg Cr/kg dw or the state means provided in Table 8.8.	
Marine	90 mg Cr/kg in uncontaminated areas.	

36. Background sediment zinc (mg Zn/kg dw)

Freshwater	Use the mean values provided for each state in Table 8.14 or the values associated with the various landscape types provided in Table 15.	
Marine	Use the following default values based on a qualitative assessment of the proportion fines (silt and clay) in the sediment matrix.	
Environment		
Default background zinc concentration (mgZn/kg)		
Erosional (<20% fines)	22.2	
Sandy (20 to 60% fines)	44.7	
Muddy (>60% fines)	74.7	

37. Background sediment penta (mg penta/kg dw for both freshwater and marine environments)

Rural and urban	0.00 mg penta/kg dw	
Industrial areas	0.41 mg penta/kg dw	

38. Background sediment PAH (mg ΣPAH/kg for both freshwater and marine environments)

Lakes	3.4 mg ΣPAH/kg dw	
Dense urban	8.9 mg ΣPAH/kg dw	
Light urban	1.3 mg ΣPAH/kg dw	
Rural areas	0.3 mg ΣPAH/kg dw	

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