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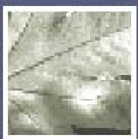

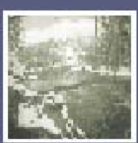

Treated Wood in Aquatic Environments: Technical Review and Use Recommendations

Prepared for:

National Marine Fisheries Service
Southwest Region
Habitat Conservation Division
501 West Ocean Boulevard, Suite 4200
Long Beach, CA 90802

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Prepared by:

Stratus Consulting Inc.
PO Box 4059
Boulder, CO 80306-4059
(303) 381-8000

-with-

Paladin Water Quality Consulting
Corvallis, OR

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In the fall of 2004, the National Marine Fisheries Service (NMFS) contracted with Stratus Consulting to conduct an independent, third party review of treated wood utilization in aquatic environments. This review is meant to support developing or updating NMFS guidelines for the use of treated wood along the Pacific Coast of the United States. The contract was awarded for copper-treated wood products and later amended to include a review of creosote-treated products as well. Substantive work on the project was completed in the fall of 2005. These reports are the findings of Stratus Consulting regarding the use of treated wood. They have been subject to peer review and public comment. NMFS may utilize these reports and other available information, as appropriate, to develop or update guidelines on the use of treated wood in aquatic environments. Accordingly, these documents are not NMFS guidelines themselves.

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Acronyms and Abbreviations

ACA	ammoniacal copper arsenate
ACQ	alkaline copper quaternary
ACQ-B	ammoniacal copper quat
ACQ-D	amine copper quat
ACZA	ammoniacal copper zinc arsenate
ADBAC	alkyldimethylbenzylammonium chloride
AETA	apparent effects threshold approach
ALC	Aquatic Life Criteria
As(III)	trivalent arsenic
As(V)	pentavalent arsenic
AWPA	American Wood Preservers Association
BA	biological assessment
BLM	Biotic Ligand Model
BMPs	best management practices
CA-B	copper azole
CBA	copper boron azole
CC	copper citrate
CCA	chromated copper arsenate
CCC	criterion continuous concentration
CDDC	copper dimethyldithiocarbamate
CITW	Canadian Institute of Treated Wood
CMC	criterion maximum concentration
CRC	California Recycling Company
Cr(III)	trivalent chromium
Cr(VI)	hexavalent chromium
CuN	copper naphthenate
DDAC	didecyldimethylammonium chloride
EFH	Essential Fish Habitat
EqPA	equilibrium partitioning approach
ESA	Endangered Species Act
ESUs	Evolutionarily Significant Units
FRP	fiber reinforced polymer

HAPC	habitat areas of particular concern
HDPE	high density polyethylene
MSA	Magnuson-Stevens Act
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
PAH	polycyclic aromatic hydrocarbon
pcf	pounds per cubic foot
PEC	probable effect concentration
ppb	parts per billion
ppm	parts per million
ppt	parts per thousand
PVC	polyvinyl chloride
RPL	recycled plastic lumber
SBA	sediment background approach
SLCA	screening level concentration approach
SQGs	sediment quality guidelines
SQTA	sediment quality triad approach
SSBA	spiked sediment bioassay approach
T&E	threatened and endangered
TEC	threshold effect concentration
TRA	tissue residue approach
USACE	U.S. Army Corps of Engineers
U.S. EPA	U.S. Environmental Protection Agency
USFWS	U.S. Fish and Wildlife Service
UV	ultraviolet
WEA	weight of evidence approach
WPC	wood plastic composite
WWPI	Western Wood Preservers Institute

1. Introduction

1.1 Background and Report Organization

Wood is a common construction material used for bridges, docks, piers, and other submerged and overwater structures. Wood is subject to fungal decay and to attack by wood boring organisms, especially in saltwater and estuarine environments. To reduce the incidence of decay and attack, chemicals are impregnated into wood used for submerged and near-water construction. Wood-treating chemicals, which include a wide array of organic and inorganic chemicals, can leach from the wood into the immediate aquatic environment, potentially harming aquatic biota.

The National Marine Fisheries Service (NMFS) of the National Oceanic and Atmospheric Administration (NOAA) is developing guidance on the use of treated wood in aquatic environments utilized by federal trust fishery resources. NMFS trust resources include commercially important marine species and their habitats, as well as threatened and endangered (T&E) marine species and their habitats. NMFS provides review and consultation on marine, estuarine, and freshwater construction projects that potentially could impact trust resources. Federal and state agencies and industry have requested guidelines from NMFS on the use of construction materials, including treated lumber, in aquatic environments in the Pacific coastal region.

The purpose of this report is to assist NMFS with the development of these guidelines. Data and information are reviewed to evaluate potential hazards to aquatic organisms from treated wood in aquatic environments. The data and information review focused specifically on the Pacific Coast states of California, Oregon, Washington, and Alaska. This report is a companion to "Creosote-Treated Wood in Aquatic Environments: Technical Review and Use Recommendations" (Stratus Consulting and DiGiulio, 2005). That report describes creosote wood treatments; this report describes water-soluble treatments. The two reports share a similar introduction and overall structure; however, the other report does not include separate chapters about alternative materials and current regulations and best management practices (BMPs) that are covered in this report.

In the following sections, we describe NMFS trust resources, types of wood treatment, the chemicals used in waterborne wood treatment processes, and the treatments that have been evaluated and approved by the American Wood Preservers Association (AWPA) as effective preservatives for use in aquatic environments.

The remainder of this report is organized as follows. Chapter 2 discusses data and information regarding leaching of chemicals from treated wood into aquatic environments, and the potential for exposure of aquatic organisms to leached chemicals. In Chapter 3, we discuss the toxicity of the leached chemicals to aquatic biota, focusing on copper. Chapter 4 discusses potential risks to NMFS trust resources. In Chapter 5, we identify materials that can be used as alternatives to wood in marine and freshwater constructions, and discuss briefly the economic considerations related to the alternative materials. In Chapter 6, we describe existing regulations and BMPs governing the use of treated wood in aquatic environments. Finally, in Chapter 7, we provide conclusions regarding conditions of use and recommendations to minimize the environmental risks of toxic chemicals in aquatic environments. Literature cited follows Chapter 7.

1.2 Trust Resources

Under the Magnuson-Stevens Act (MSA), sections 303(a)(7) and 305(b)(2), NMFS is responsible for managing commercially harvested aquatic species (including several salmonid species) by, among other things, implementing fishery management plans and by designating protective Essential Fish Habitat (EFH) areas. The fishery management plans for commercially important species are managed by regional fisheries management councils. The Pacific Fisheries Management Council manages commercially important species for the States of California, Oregon, and Washington. The Northern Pacific Fisheries Management Council manages commercially important species for the State of Alaska.

The fishery management plans must designate both the habitat essential to the commercial species of concern and the threats to their habitat from fishing and non-fishing activities. EFH areas include, as defined by Congress, ". . . those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity." EFH guidelines at 50 CFR 600.10 also specifically define substrate as including, ". . . associated biological communities." Salmonid EFH areas designated in accordance with the MSA include all streams, lakes, and other water bodies currently or historically accessible to salmon in Alaska, Washington, Oregon, and California, and includes most Pacific Coast rivers, streams, and estuaries. In addition, NMFS may identify priority habitats within EFH areas as habitat areas of particular concern (HAPC) for conservation and management of the species.

Chinook salmon (*Oncorhynchus tshawytscha*), coho salmon (*Oncorhynchus kisutch*), and pink salmon (*Oncorhynchus gorbuscha*) are the three main commercially significant salmon species managed under the MSA by the North Pacific and Pacific Fishery Management Councils. EFH for these species in marine and estuarine areas of the Pacific Coast region extends seaward from the shoreline out to the 200-mile limit of the U.S. Exclusive Economic Zone. Shoreward, salmonid EFH comprises all bodies of water extending inland that were historically accessible to salmon, with the exception of certain barriers and dams that fish cannot pass (PFMC, 2004).

Chinook salmon habitat spans from the U.S.-Mexico border to Kotzebue Sound in northwestern Alaska. Coho salmon spawn in tributaries from the San Lorenzo River in Monterey Bay, California, to Point Hope, Alaska, and throughout the Aleutian Islands (PFMC, 2003).

Under the Endangered Species Act (ESA), NMFS' trust resources include T&E aquatic species. In addition to the MSA mandated habitat protections, sections 3(5)(A) and 7 of the ESA require NMFS to conserve the ecosystems upon which T&E species depend, to provide a program for the conservation of T&E species, and to ensure that they (and all federal agencies) do not fund, authorize, or carry out any actions that will jeopardize the continued existence of listed species or result in the destruction or adverse modification of designated critical habitat. To this end, NMFS is authorized to designate "critical habitat" for those species. Under ESA section 7(a)(2), NMFS is responsible for developing guidelines and policies to protect federally listed T&E aquatic organisms and their habitats from pollutants.

There are 1,290 species, subspecies, Distinct Population Segments, and Evolutionarily Significant Units (ESUs) listed under the ESA. Of the aquatic species, the NMFS Office of Protected Resources manages mostly marine and anadromous species. The U.S. Fish and Wildlife Service (USFWS) manages the remainder of the listed species, which are primarily terrestrial and freshwater species. The NMFS Office of Protected Resources manages 61 ESA-listed aquatic species, 43 aquatic species of concern, and approximately 175 marine mammal stocks listed under the Marine Mammal Protection Act. Of the 51 salmonid ESUs, 30 are either listed as T&E, or are candidates for listing (Table 1.1).

Table 1.1. Status of West Coast salmonid species and ESUs

Species	ESU	Listing status[a]	T/E status[b]
Pink salmon	Even year ESU[c]	NW	
	Odd year ESU[c]	NW	
Coho salmon	Central CA ESU	L	E
	Southern OR/northern CA coasts ESU	L	T
	OR coast ESU	L	T
	Puget Sound/Strait of Georgia ESU	C	
	Lower Columbia River ESU	C	
	Olympic Peninsula ESU	NW	
	Southwest Washington	NW	
Chinook salmon	Sacramento River winter-run ESU	L	E
	Snake River fall-run ESU	L	T
	Snake River spring/summer-run ESU	L	T
	Puget Sound ESU	L	T
	Lower Columbia River ESU	L	T

Table 1.1. Status of West Coast salmonid species and ESUs (cont.)

Species	ESU	Listing status[a]	T/E status[b]
Chinook salmon (cont.)	Upper Willamette River ESU	L	T
	Upper Columbia River spring-run ESU	L	E
	Central Valley spring-run ESU	L	T
	CA coastal ESU	L	T
	Central Valley fall and late fall-run ESU	C	
	Upper Klamath-Trinity rivers ESU	NW	
	OR coast ESU	NW	
	WA coast ESU	NW	
	Mid-Columbia River spring-run ESU	NW	
	Upper Columbia River summer/fall-run ESU	NW	
	Southern OR/northern CA coasts ESU	NW	
	Deschutes River summer/fall-run ESU	NW	
Chum salmon	Hood Canal summer-run ESU	L	T
	Columbia River ESU	L	T
	Puget Sound/Strait of Georgia ESU	NW	
	Pacific Coast ESU	NW	
Sockeye salmon	Snake River ESU	L	E
	Ozette Lake ESU	L	T
	Baker River ESU	NW	
	Okanogan River ESU	NW	
	Lake Wenatchee ESU	NW	
	Quinault Lake ESU	NW	
	Lake Pleasant ESU	NW	
Steelhead	Southern CA ESU	L	E
	South-Central CA coast ESU	L	T
	Central CA coast ESU	L	T
	Upper Columbia River ESU	L	E
	Snake River Basin ESU	L	T
	Lower Columbia River ESU	L	T
	CA Central Valley ESU	L	T
	Upper Willamette ESU	L	T

Table 1.1. Status of West Coast salmonid species and ESUs (cont.)

Species	ESU	Listing status[a]	T/E status[b]
Steelhead (cont.)	Middle Columbia River ESU	L	T
	Northern CA ESU	L	T
	OR coast ESU	C	
	Southwest WA ESU	NW	
	Olympic Peninsula ESU	NW	
	Puget Sound ESU	NW	
	Klamath Mountains Province ESU	NW	

a. L = listed, C = candidate, NW = not warranted.
b. E = endangered, T = threatened.
c. Managed by NMFS every other year (jointly with Canada).
Source: NOAA, 2005.

1.3 Types of Wood Treatment

Treated wood pilings, timbers, and other wooden lumber have been used in marine construction in the United States for more than a hundred years (Lebow and Tippie, 2001). Although some woods are more naturally resistant to deterioration, wood construction materials exposed to water must be preserved with chemicals to prevent deterioration and eventual destruction by marine borers such as crustaceans (gribbles, *Limnaria* spp.), mollusks (boring clams, *Teredo* or *Bankia* spp.), and other wood degrading organisms, including fungi. To protect wood from these organisms, preservative formulations must be toxic to the wood degrading organisms.

Currently, wood preservatives include both water-soluble formulations and oil-based formulations. This report focuses on water-soluble treatment types (Table 1.2). Oil-based wood treatment products, including creosote, will be described and discussed in a companion report.

Water-soluble wood treatments recommended by the AWPA for use in or near water include chromated copper arsenate (CCA), ammoniacal copper zinc arsenate (ACZA), alkaline copper quaternary (ACQ), and copper azole (CA-B) (Table 1.2; AWPA, 2003). ACZA and CCA are the principle wood treatments used in saltwater. For freshwater and over-water structures, ACZA, CCA, ACQ, and CA-B are the principle treatments used. CCA is the most widely used treatment in the United States, but on the Pacific Coast, Douglas fir treated with ACZA is the most common because CCA treatment of Douglas fir is relatively ineffective (Lebow and Morell, 1995).

Table 1.2. Common water-soluble treatments and uses for wood construction materials used in aquatic environments

Type of preservative	Acronym	Components	Recommended use
Chromated copper arsenate, types A, B, C	CCA-A CCA-B CCA-C	Cr (VI)/(III), CuO, As_2O_5	Salt/freshwater Salt/freshwater Salt/freshwater
Ammoniacal copper zinc arsenate	ACZA	Ammonium bicarbonate, CuO, ZnO, As_2O_5	Salt/freshwater
Alkaline copper quat, types A, B, C	ACQ-B ACQ-D ACQ-C	CuO, quaternary ammonium compounds, ammonium bicarbonate, ethanolamine bicarbonate	Freshwater Freshwater Freshwater
Copper azole, type B	CA-B	CuO, tebuconazole	Freshwater
Copper dimethyldithiocarbamate (CDDC)	CDDC	Bivalent copper-ethanolamine (2-aminoethanol), sodium dimethyldithiocarbamate	No longer recommended for use
Source: AWPA, 2003.			

CCA contains chromium, copper, and arsenic acid (arsenate). Proportions of the components vary by formulation. Currently, three CCA formulations are registered for use in the United States: CCA types A, B, and C. Use of types A and B has diminished since the introduction of type C in 1968 (Brooks, 1997a). CCA type C contains, by weight, 47.5% hexavalent chromium as CrO_3, 18.5% copper as CuO, and 34% arsenic as As_2O_5 (AWPA, 2003). The wood treatment process involves the reduction of the more toxic hexavalent chromium to a less toxic trivalent form and, in the process, oxidization and fixation of the other metals within the wood.

ACZA is a reformulation of an older preservative no longer in use, ammoniacal copper arsenate (ACA), in which a portion of the arsenic is replaced with zinc (AWPA, 2003; Lebow et al., 2004). The current formulation of ACZA contains, by weight: 25% zinc as ZnO, 50% copper as CuO, and 25% arsenic as As_2O_5 (AWPA, 2003). The ammonia in the formulation catalyzes the fixation of the copper, arsenic, and zinc to the wood fibers. The metals are thought to be fixed by a combination of ionic bonds, Van der Waals forces, and bonds to wood sugars within insoluble metal complexes (Brooks, 1997a).

ACQ is a mixture containing copper as CuO and a quaternary ammonium (didecyldimethylammonium chloride or an alkylbenzyldimethylammonium chloride) in an ammonia and carbonate or ethanolamine and carbonate carrier (AWPA, 2003; CDPR, 2005). ACQ contains two active ingredients: copper oxide and a quaternary ammonium compound called quat. The ACQ-B formulation, which contains 66.7% copper oxide and 33.3% quat as didecyldimethylammonium chloride (DDAC), was standardized by the AWPA in 1992. ACQ-B is used on difficult-to-treat wood species such as Douglas fir because its ammonia carrier solution allows the ACQ to penetrate the wood better than other formulations. The ACQ-C formulation, which contains 66.7% copper oxide and 33.3% quat as alkyldimethylbenzyl-ammonium chloride (ADBAC), was standardized by the AWPA in 2002. Ammonia or ethanolamine or both can be used as the carrying solution for ACQ-C. ACQ-D was standardized by the AWPA in 1995 and contains 66.7% copper oxide and 33.3% quat as DDAC. Type D differs from type B in that it uses an ethanolamine carrier solution rather than ammonia.

Copper azole is a recently developed formulation. The copper azole CA-B contains copper and the fungicide tebuconazole. Both are carried in an ethanolamine carrier solution.

CDDC is one of a class of carbamate fungicides that have metal ions (Cu, Zn, Fe, Mn, or Na) complexed to the free thiol groups. CDDC is the only carbamate fungicide currently listed for wood treatment. CDDC is licensed for freshwater use, but it is potentially highly toxic to freshwater invertebrates at concentrations as low as 10 ng/L. The wood treatment industry is moving away from its use for economic reasons (personal communication, Randy Baileys, J.H. Baxter, Dennis Hayward, WWPI, January 7, 2005), and it is no longer commercially available (Lebow et al., 2004).

Two other waterborne preservatives currently being removed from the standards and no longer commercially available are ACA and copper citrate (CC) (personal communication, D. Hayward, WWPI, November 22, 2004). ACA contained copper as CuO and arsenic as As_2O_5. CC contained copper as CuO and citric acid. Commercial use of copper boron azole (CBA) is rapidly diminishing in North America, following the introduction of CA-B.

2. Models of Metal Leaching from Treated Wood and Environmental Exposure

This chapter reviews and evaluates models that have been developed to predict the leaching of metals from treated wood and the resultant metal concentrations in the environment. The rate and amount of metals that leach from treated wood is a key component in the evaluation of the potential effects of treated wood on aquatic biota, and much study has been conducted in this area. Nearly all studies of metal leaching from treated wood have been conducted in the laboratory under controlled conditions, and models have been developed to predict metal leaching under these conditions. Another key component in evaluating the potential effects of metals leached from treated wood is estimating the environmental concentrations of metals that result from the leaching. Several investigators have developed models that predict concentrations of metals in surface water or sediment based on a given metal leaching rate.

Because CCA-C and ACZA are the dominant water-soluble treatment solutions in use, and because most leaching and environmental studies have been conducted on these formulations (Lebow, 1996), this chapter focuses on these two preservatives. Additional information on other water-soluble preservatives is provided where available.

Section 2.1 discusses factors that affect metal leaching rates from treated wood, and Section 2.2 describes and discusses quantitative leaching models that have been developed and applied. Section 2.3 discusses the application of laboratory-derived leach estimates to field conditions. Section 2.4 discusses the estimation of environmental metal concentrations in the environment based on the metal leaching models, and Section 2.5 presents conclusions.

2.1 Factors that Influence Metal Leaching Rates

The chemical processes that occur when the metal-based treatment solutions are fixed are complex and poorly understood (Lebow, 1996; Hingston et al., 2001). It is generally thought that during the fixing process the mixtures of metals undergo a series of reactions that may include the formation of insoluble complexes, oxidation-reduction reactions with lignins and wood sugars, and precipitation reactions as volatile carrier chemicals such as ammonia or ethanolamine are removed from the wood by vacuum or air drying (Forsyth and Morrell, 1990; Lebow and Morrell, 1995). For CCA fixed wood, the conversion of Cr(VI) to Cr(III) is also a key process in the fixation (Forsyth and Morrell, 1990; Hingston et al., 2001).

Laboratory studies of metal leaching from treated wood have identified the following as the most important factors that affect leaching rates of metals from treated wood:

- The metal being considered (Cu, Cr, As, or Zn)
- Post-treatment procedures used to fix the treatment chemicals and remove excess treatment solution
- Time (i.e., duration of exposure to water post-treatment)
- The loading or retention of the treatment solution in the wood
- Ambient water quality conditions, including salinity, pH, and temperature
- Current speed
- Physical features of the wood surface, including surface area-to-volume ratio.

A brief summary of these factors is provided below. The literature reviews by Lebow (1996), Brooks (1997a, 1997b, 2003), and Hingston et al. (2001) provide more detailed evaluations of the available literature on the specific factors that affect metal leaching rates from treated wood.

Cu leaching rates from CCA- and ACZA-treated wood are typically higher than rates for the other metals in these preservative solutions (Cr, As, and Zn) (Hingston et al., 2001; Brooks, 2003). For example, Lebow et al. (1999) observed Cu leaching rates from CCA-treated wood that were generally several times to several orders of magnitude greater than the leaching rates for As or Cr. However, some studies have observed leaching rates of As that are similar to or higher than Cu under some conditions (Lebow et al., 2000; Hingston et al., 2001). Nevertheless, the generally higher leaching rate of Cu has typically resulted in more focus on this metal than the others (Brooks, 2003).

Metal leaching rates have been studied more extensively for CCA and ACZA treatment solutions than for the other water-based treatment solutions (Hingston et al., 2001). Under loading and fixing conditions as specified by relevant guidance and BMPs (see Chapter 6), the Cu leaching rates from wood treated with ACZA are much higher (at least initially) than for wood treated with CCA, all other factors being equal (Brooks, 1997a, 1997b, 2003). Results from a limited number of studies indicate that Cu leaching rates are also higher for ACQ and CA-B than CCA as well (Townsend et al., 2003; Brooks, 2005). Quantitative data are insufficient to draw definitive conclusions regarding relative As leaching rates between CCA and ACZA-treated wood (Brooks, 1997a, 1997b, 2003).

Higher amounts of residual, unfixed chemicals on the surface of the wood, or incomplete fixing within the wood, lead to higher rates of metal leaching (Lebow, 1996; Hingston et al., 2001). Post-treatment procedures to fix the chemicals and remove excess treatment solution are specified in the treatment BMPs developed by the Western Wood Preservers Institute (WWPI) and the Canadian Institute of Treated Wood (CITW). These BMPs specify procedures to retain treatment efficacy while minimizing the residual unfixed products in and on the wood surface.

Metal leaching rates from treated wood decline dramatically with time of exposure. The sharp decrease in metal leaching rates with time has been demonstrated consistently in laboratory and field studies (Hingston et al., 2001). After the initial steep decline, leaching rates reach a lower rate that is relatively constant over time (Lebow, 1996; Lebow et al., 1999; Hingston et al., 2001). Brooks (2003), in a review of the literature on metal leaching from CCA-treated wood, concludes that the decline in Cu leaching rates follows an exponential decay curve. It has been hypothesized that the leaching rates reflect two phases of leaching from the wood: an initial phase during which excess and unfixed preservative is quickly leached into the surrounding water, followed by a longer-term leaching that is driven by diffusion through the wood across concentration gradients (Lebow, 1996). The initial phase of rapidly declining leaching rate most likely varies depending on wood characteristics, treatment and fixation techniques, and environmental conditions, but it has been reported to take from several days to several weeks or more (Hingston et al., 2001; Brooks, 2003). Alternatively, Hingston et al. (2001) proposed three phases of leaching, with an initial phase (that lasts hours) consisting of loss of surface deposits, a mid-term phase (days to weeks) consisting of loss of unfixed and labile preservative, and a long-term phase that consists of reversible dissociation of bound metals and migration to the wood surface.

The relationship between preservative loading or retention and leaching rate is not clear, with some studies observing a positive effect, no effect, or a negative effect on the leaching rates of metals (Hingston et al., 2001). Increased CCA loading or retention is reported to cause a slight increase in Cu leaching rates, no effect on As leaching rates, and a decrease in Cr leaching rates (Brooks, 2003). Morrell et al. (1998) evaluated post-treatment processes on subsequent loss of metals from ACZA treated Douglas fir, and found markedly elevated As leaching (approximately 50-fold higher) from wood with lower retention levels versus higher levels (6.4 kg/m^3 vs. 40 kg/m^3). The Cu leaching rate was reduced and the zinc rate was elevated approximately two-fold from the wood at the lower retention level treatment.

The effect of salinity on metal leaching rates is also complex. For CCA-treated wood, increasing salinity increases the leaching rate of Cu (Hingston et al., 2001). Brooks (2003) estimates that copper leach rates in saltwater (at 30 parts per thousand, ppt, salinity) are about 1.5-2 times greater than in freshwater. The cause for the higher leaching rate in higher salinity water is thought to be related to the higher rates of ion exchange that can occur with saltwater (Lebow et al., 1999). In contrast to Cu, however, increasing salinity has been reported to decrease the leaching rate of Cr in CCA-treated wood (Brooks, 2003). Salinity does not appear to affect the leaching rate of As from CCA-treated wood (Lebow et al., 1999).

The effects of salinity on leaching rates for wood treated with ACZA are different than for CCA-treated wood. Brooks (1997a) used a regression model to compare Cu leaching rates in freshwater and saltwater from wood treated with ACZA, and concluded that Cu leaching rates in freshwater are approximately three times greater than in saltwater. Available data were insufficient to conduct a meaningful comparison of As and Zn leaching rates between freshwater and saltwater environments for ACZA (Brooks, 1997a). However, Brooks reports that the time to reach steady state leaching rates is longer for ACZA-treated wood in freshwater (approximately 10 days) than in seawater (approximately 4 days) (Brooks, 1997a).

Reported Cu leaching rates from CCA- and ACZA-treated wood tend to vary inversely with the pH of the ambient water, with higher leaching rates occurring at lower pH values (Brooks, 1997a, 1997b, 2003; Hingston et al., 2001). The leaching rates of As, Cr, and Zn are less dependent on ambient water pH, and results reported in the literature show no clear relationships (Hingston et al., 2001). Data from a test on Cu leaching from CA-B-treated wood following one year of immersion indicate that leaching is highest at circum-neutral pH, and lower at both higher (approximately 8.5) and lower (approximately 5.5) pH values (Brooks, 2005).

The leaching rates of Cu and Cr tend to increase with increasing ambient water temperature (Hingston et al., 2001; Brooks, 2003). Results for As are mixed, with some studies showing higher leaching rates at higher water temperatures, and some showing lower leaching rates at higher temperatures (Hingston et al., 2001; Brooks, 2003).

Finally, metal leaching rates can also vary based on the physical characteristics of the wood being tested. Different species have different affinities for preservative, affecting both retention and leaching rates (Lebow, 1996). In general, Cu, As, and Cr leach more readily from hardwoods treated with CCA than from softwoods (Lebow, 1996). Wood size, shape, and texture can also affect metal leaching rates, with increases in surface area or surface area to volume ratios typically causing an increase in leaching rates (Lebow, 1996; Hingston et al., 2001). Leaching rates from poles or piles are typically higher than from boards because of the greater variability in the surface of poles or piles that increases the effective surface area (Lebow, 1996; Lebow et al., 1999). Importantly, leaching from sawdust or wood shavings, such as is produced during construction of structures from the treated wood, is many times greater than leaching from whole boards or piles (Weis et al., 1991; Lebow, 1996; Brooks, 2003), which has implications for BMPs related to preventing the sawdust or shavings from entering the environment (Brooks, 2000).

2.2 Models of Metal Leaching Rates

Several investigators have developed quantitative models to predict the leaching rates of metals from treated wood. In this section, we describe the available models and evaluate their effectiveness at accurately predicting leaching rates observed in field and laboratory studies.

2.2.1 Description of the available models

Empirical models developed by Dr. Kenneth Brooks

On behalf of the WWPI or Arch Treatment Technologies, Inc., Brooks has developed quantitative models of metal leaching rates for CCA (Brooks, 1997b, 2003), ACZA (Brooks, 1997a), and the copper azole CA-B (Brooks, 2005). The models are empirical models that were fit to data from published and unpublished laboratory leaching studies. The models reviewed here have not been published in peer-reviewed journals. Where possible, only results from studies that met the following criteria were used:

- The wood being tested is treated to AWPA specifications, has small end grain to surface grain ratios or sealed ends, and does not consist of sawdust
- For CCA, preservative fixation is confirmed prior to leaching with the chromotropic acid test
- Leaching is conducted using water of pH between 5.0 and 9.0, salinity between 0 and 40 ppt, and temperature between 5 and 35°C
- The tests are static renewal or flow-through.

Based on the data included in the screened database, Brooks used regression models to predict "worst-case" average metal leaching rates (the dependent variable) as a function of exposure time and, for some metals and treatments, environmental variables. Specifically, the following regression models were developed by Brooks:

- For CCA, Cu leaching rate as a function of time, water salinity, water temperature, water pH, and retention; Cr leaching rate as a function of time, water salinity, water temperature, and retention; and As leaching rate as a function of time and salinity. An initial model was presented in Brooks (1997b), and was subsequently updated in Brooks (2003).

- For ACZA, Cu leaching rate as a function of time in saltwater and time and water pH in freshwater; As leaching rate as a constant; and Zn leaching rate as a function of time in saltwater and time and water pH in freshwater (Brooks, 1997a).

- For CA-B, Cu leaching rate as a function of time, water pH, and water temperature (Brooks, 2005).

Brooks' CCA leaching models

The CCA leaching models were developed from a database containing 322 observations (Brooks, 2003). For Cu leaching, Brooks (2003) reports that nonlinear regression was used to arrive at the following equation:

$$\text{Cu loss rate } (\mu g/cm^2 \text{ day}) = 0.036*T + 0.021*(S + 0.01) - 0.002*R - 0.031*pH + 6.95*e^{(0.007*R + 0.121*T + 0.015*S - 0.284*pH - 1.379*Time)}$$

where:

T	= receiving water temperature (in degrees centigrade)
S	= receiving water salinity
R	= CCA retention (in kg/m^3)
pH	= receiving water pH
Time	= duration of exposure (in days).

The nonlinear regression technique used by Brooks (2003) is not fully documented. It appears that the underlying form of the equation was first defined, followed by parameterization of equation constants using nonlinear regression. The equation includes both a time-dependent exponential decay function and time-independent terms, and thus assumes that the long-term steady state leaching rate will ultimately approach a constant value. The formulation of the equation means that the leaching rate initially follows an exponential decay with time, and asymptotically approaches a constant rate that is a function of water temperature, salinity, pH, and CCA retention. This underlying conceptual formulation of the model appears to be consistent with Cu leaching data from laboratory studies, but alternative model formulations could also be consistent with the available laboratory data.

The CCA leaching model for Cr developed by Brooks includes only an exponential decay function with time that includes the variables T, R, and S (Brooks, 2003):

$$\text{Cr loss rate } (\mu g/cm^2 \text{ day}) = 0.47*e^{(-0.013*R + 0.103*T - 0.031*S - 1.07*Time)}$$

Therefore, the Cr loss rate is modeled to asymptotically approach zero.

The CCA leaching model for As developed by Brooks includes both an exponential decay function and a time-independent constant that includes only salinity as a variable (Brooks, 2003):

$$\text{As loss rate } (\mu g/cm^2 \text{ day}) = 0.010*S + 0.754*e^{(-0.130*Time)}$$

Therefore, in saltwater As, leaching will asymptotically approach the value 0.010*S, whereas in freshwater As, leaching will asymptotically approach 0.

Brooks (2003) reports that the coefficients of determination for the regression equations are 80.4% for Cu, 36.6% for Cr, and 64.2% for As. However, to fully evaluate the effectiveness of the models at capturing the laboratory data, a more detailed examination of model predictions versus observations as a function of changes in environmental variables, time, and test conditions is required. Dr. Brooks provided the 322 raw data observations upon which the model is based in a personal communication (personal communication, Kenneth Brooks, July 8, 2005). Figure 2.1 is a plot of the Cu leaching rates predicted by the Brooks (2003) model against the observed leaching rates used to parameterize the model.

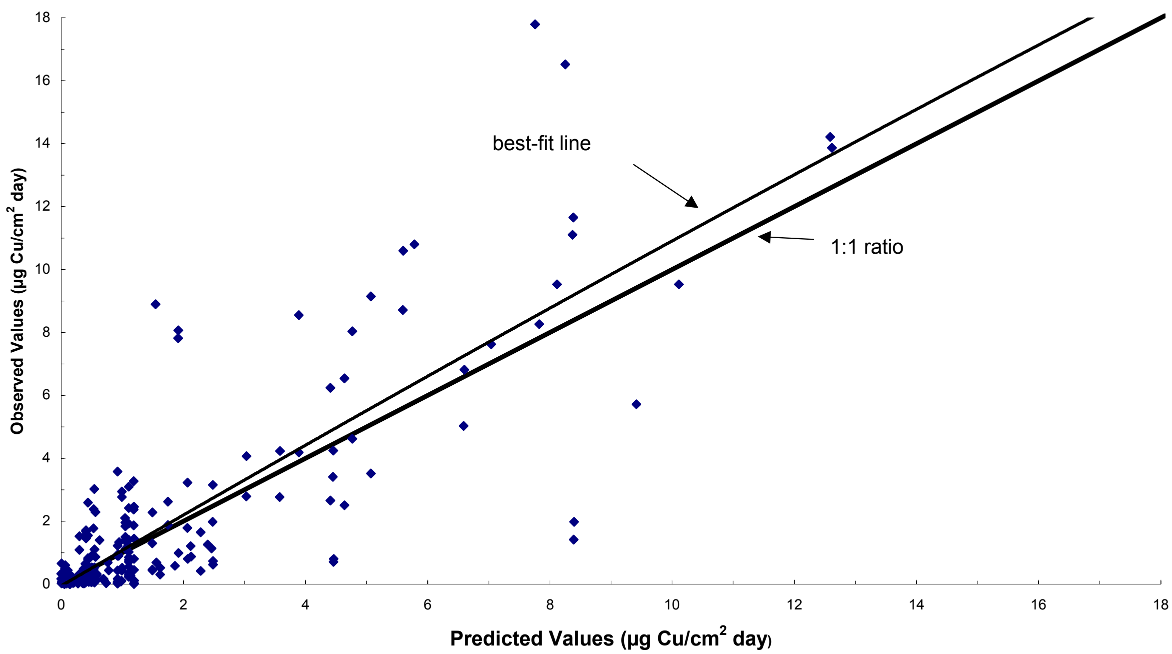

Figure 2.1. Comparison of predicted versus observed Cu leaching rates for the 332 observations upon which the Brooks (2003) CCA leaching model is based.

As shown in the figure, the best-fit line through the data is closely similar to the 1:1 line corresponding to predicted values being equal to observed values. However, substantial variability about the 1:1 line is evident in the plot. No directional bias is evident, with the model providing both under-predictions and over-predictions of the actual values across the range of leaching rates measured. Nevertheless, the plot shows that many of the predicted leaching values differ from the corresponding observed values in the laboratory leaching studies by a factor of two or more. This variation could be caused by differences in specific laboratory testing protocols not captured by the model; other variables related to wood treatment, fixation, and post-fixation processing that influence leaching that are not included in the model; or variations that are inherent in conducting laboratory leaching studies of this nature. Also, wood is a biological material and, as such, its characteristics are subject to natural variation. Nevertheless, the plot in Figure 2.1 indicates that the Brooks (2003) model does a reasonable job of capturing a larger database of laboratory leaching values, but there is substantial uncertainty at the level of the individual study or set of studies that introduce uncertainty in applying the models to predict "worst-case scenario" leaching rates at the level of the individual study.

Brooks' ACZA leaching models

Brooks (1997a) reports on leaching models for Cu, As, and Zn from wood treated with ACZA. Data from laboratory leaching studies conducted on behalf of the J.H. Baxter Company in support of the U.S. Environmental Protection Agency (U.S. EPA) re-registration of ACZA were used to develop the model (Brooks, 1997a). Tests were conducted in saltwater and freshwater, and Brooks (1997a) developed separate equations for the two water types, rather than attempting to include salinity as a variable. The leaching models for Cu from ACZA-treated wood are:

$$\text{For saltwater: Cu loss rate } (\mu g/cm^2 \text{ day}) = 32.5 * e^{(-1.114*\text{Time})}$$

$$\text{For freshwater: Cu loss rate } (\mu g/cm^2 \text{ day}) = 1{,}908.6 * e^{(-0.429*\text{Time} - 0.383*\text{pH})}$$

Thus, the model predicts that Cu loss rate asymptotically approaches zero in both saltwater and freshwater, and that the Cu loss rate is dependent on pH only in freshwater. Furthermore, the initial loss rate is much higher in freshwater than in saltwater under environmentally realistic pH conditions. Brooks (1997a) reports coefficients of determination of 0.758 for Cu loss in saltwater and 0.929 for Cu loss in freshwater. Raw data or detailed comparisons of observed versus predicted values are not provided.

The As leaching model from ACZA-treated wood assumes a time-independent, constant leaching rate of 0.099 $\mu g/cm^2$ day for both saltwater and freshwater (Brooks, 1997a). Brooks (1997a) states that too few data points were available to conduct a meaningful regression analysis for As. The constant rate of 0.099 $\mu g/cm^2$ day is reported by Brooks (1997a) as the average value of all measured leaching rates for As, which ranged from 0.0 to 1.375 $\mu g/cm^2$ day.

The formulations of the leaching models for Zn from ACZA-treated wood are the same as for Cu, but with different parameters (Brooks, 1997a):

$$\text{For saltwater: Zn loss rate } (\mu g/cm^2 \text{ day}) = 31.074*e^{(-2.667*Time)}$$

$$\text{For freshwater: Zn loss rate } (\mu g/cm^2 \text{ day}) = 166.6*e^{(-1.02*Time - 1.054*pH)}$$

Brooks (1997a) reports coefficients of determination of 0.999 for Zn losses in saltwater and 0.85 for Zn losses in freshwater.

Brooks stated that he had never been comfortable with the ability of the dataset used to develop the ACZA model to accurately reflect the dynamics of ACZA leaching under real-world conditions (personal communication, Kenneth Brooks, July 7, 2005). He said that he felt that the data underrepresented leaching compared to real-world situations, because the tests were static rather than flow-through. He said that flow-through tests have recently been completed and that he anticipated revising the ACZA model in the next several months as soon as the analytical data were available.

Brooks' CA-B leaching model

Brooks (2005) also developed a model for the leaching of Cu from wood treated with CA-B. The model was based on data from leaching studies conducted by Brooks on behalf of Arch Treatment Technologies, Inc. The tests were conducted on commodity size wood treated with CA-B according to AWPA specifications, and utilized a flow-through leaching apparatus (Brooks, 2005). The equation to describe the observed Cu loss rates from the CA-B-treated wood is:

$$\text{Cu loss rate } (\mu g/cm^2 \text{ day}) = 6.49/[e^{abs(pH-7.24)}] + 203.12*e^{(-0.14*Time - 0.285*pH + 0.015*T)}$$

The formulation of the model is an exponential decay function with time, asymptotically approaching a constant value that is dependent on receiving water pH. Brooks (2005) reports that the "adjusted" coefficient of determination of the regression is 0.89, that the residuals are normally distributed, and that the variability observed in the data is associated with the range of test conditions and differences in individual boards. Brooks (2003) provides a plot comparing the observed Cu loss rates in the tests with the loss rates predicted by the model. The figure is provided in Figure 2.2, with the 1:1 line added.

The figure, reproduced as supplied, has a feature that could have a bearing on the interpretation of the data. Since the data labels appear above the data points, they may add visual weight to the points above the line.

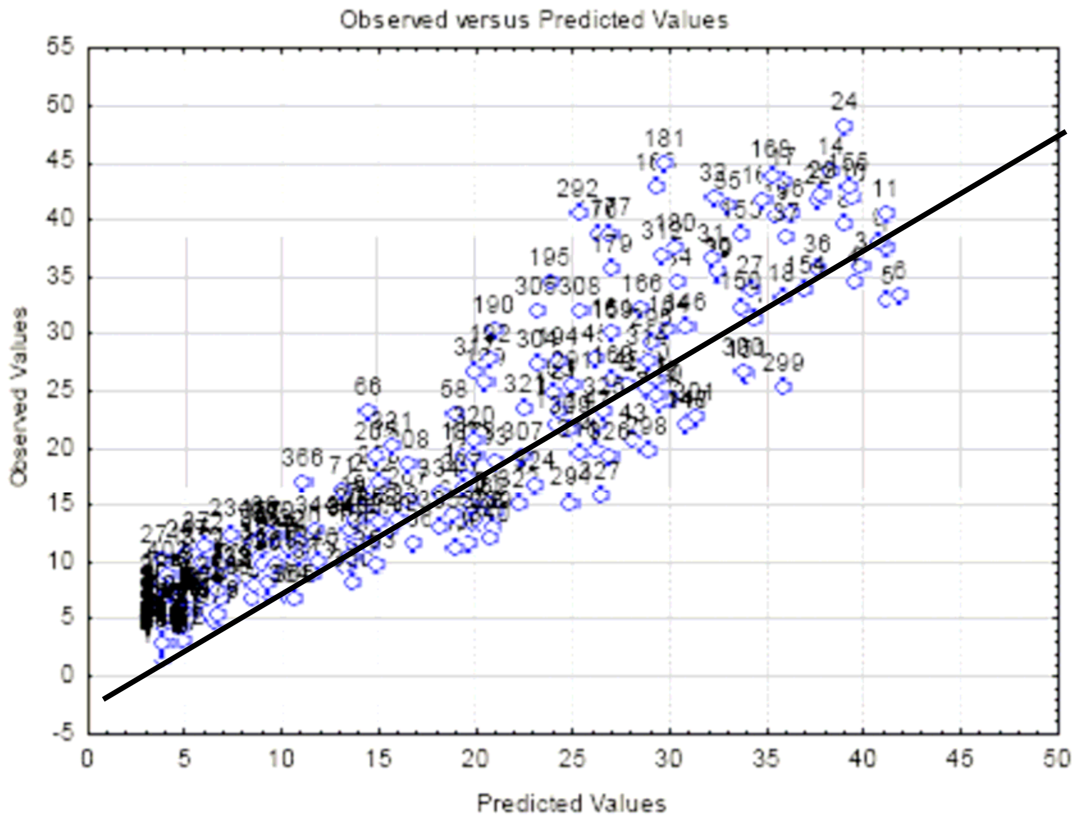

Figure 2.2. Comparison of observed Cu leaching rates from wood treated with CA-B with those predicted by the Brooks model (2005). The model is based on the observed rates shown. Units of both axes are µg/cm^2 per day. From Brooks (2005), with 1:1 prediction:observed line added. Used by permission.

Brooks (2005) notes that the heteroscedasticity in the relationship is minor, and that most of the variability between observed and predicted rates occurs at the high leaching rates that take place at the beginning of the leaching. However, as shown in the figure, although there is a relatively good fit at intermediate leaching rates, and taking into account the data label placement, the model appears to somewhat underpredict leaching rates at both the lowest and highest leaching rates. The manner in which the data points group in this plot suggests that there are systematic ways in which the observations do not match the predictions. There appear to be at least two distinct groupings, which is likely to reflect differences in leaching dynamics in different ranges of leaching rates.

The model's predictions regarding leaching rates appear to be systematically lower than the leaching rates observed in the experiment when predicted leaching rates are below approximately 10 $\mu g/cm^2$ per day. Brooks states that of the 48 cases where the predicted leaching rate is less than 5 ug/cm^2 per day, 23 are long-term loss rates, mostly at pH = 5.5 or pH = 8.5. The remaining 25 cases are intermediate-term loss rates observed on days 15.5 to 30.5. In light of this, long-term leaching of CA-B may be greater than predicted by the model at noncircum-neutral pHs. Also, Brooks states that at circum-neutral pHs, the model slightly overpredicts leaching, presumably over the short- and long-term.

CCA leaching model of Lebow et al. (1999)

Lebow et al. (1999) also developed an empirical model of Cu, Cr, and As leaching from CCA-treated wood based on data obtained in a laboratory leaching study. The laboratory leaching study was conducted on wood treated at two different retention levels and three salinity levels using a static-renewal system.

The leaching model developed by Lebow et al. (1999) is intended to predict the total amount of metal leached from the CCA-treated wood over the long-term (multiple years). The model is formulated to include a term for the amount of metal released in the initial period of higher release (defined to be six months), and a term for the amount of metal released once the release rate has reached an equilibrium value beyond the first six months:

$$\text{Amount of contaminant released } (\mu g/cm^2) \text{ after time m} = B_{LT}\Delta t_m + B_I$$

where:

B_{LT} = long term release rate ($\mu g/cm^2$ per month)
Δt_m = total time interval of estimate, minus initial release time period (six months)
B_I = initial release amount during the first six months ($\mu g/cm^2$).

Therefore, the model is not formulated to provide instantaneous leaching rates within the first six months, but instead calculates the total amount of metal released in the first six months. Beyond six months, the leaching rate is assumed to be constant (at a value of B_{LT}). The parameters of the model were determined from the test data using regression analysis (Lebow et al., 1999). Neither goodness-of-fit of the regression model to the test data are provided, nor are plots of predicted versus observed parameters.

In Table 2.1, the values for the model parameters are used to calculate the average daily release rate over the first six months, and the long-term (beyond six months) daily release rate, using the model of Lebow et al. (1999).

Table 2.1. Daily average release rates (μg/cm^2 day) calculated from the leaching model of Lebow et al. (1999)

Wood type	CCA loading level[a]	Water salinity (ppt)	Cu		As		Cr	
			First 6 months	> 6 months	First 6 months	> 6 months	First 6 months	> 6 months
Lumber	Low	0	0.17	0.010	0.12	0.030	0.033	0.0033
		34	0.26	0.21	0.038	0.010	0.044	0.0033
	High	0	0.18	0.023	0.12	0.063	0.027	0.010
		23	0.30	0.27	0.038	0.010	0.011	0.0033
		34	0.22	0.19	0.038	0.010	0.011	0.0033
Piling	Low	0	0.30	0.020	0.24	0.090	0.049	0.0067
		34	0.82	0.34	0.071	0.020	0.016	0.0033
	High	0	0.40	0.030	0.43	0.14	0.044	0.0067
		23	1.42	0.81	0.26	0.080	0.022	0.0067
		34	0.90	0.62	0.18	0.060	0.016	0.0033

a. Low retention level is approximately equal to 19 kg/m^3 for pilings and 16 kg/m^3 for lumber. High retention level is approximately equal to 41.9 kg/m^3 for pilings and 33.5 kg/m^3 for lumber (Lebow et al., 1999).

The data in Table 2.1 demonstrate that average leaching rates over the first six months are modeled to be much higher than beyond six months, which is consistent with the initial high and rapidly declining leaching rates observed in laboratory leaching experiments. However, the six-month daily average leaching rates are at least an order of magnitude less than the rate measured in the first day of the leaching experiments (Lebow et al., 1999). Therefore, the model cannot be used to provide predictions for the high initial leaching rates that occur immediately after immersion. Another feature of the modeled leaching rates shown in Table 2.1 is that the average leaching rates over the first six months and the long-term leaching rates are much higher in saltwater than in freshwater for pilings, and the long-term leaching rates are much higher in saltwater for lumber (but not the average rates over the first six months). Lebow et al. (1999) report that the model predicts that Cu releases over the initial six months account for approximately 40% of the total release over 10 years in deionized water, whereas the releases during the first six months account for only a small portion of the total releases over 10 years in saltwater.

The model predictions of Lebow et al. (1999) for Cu, As, and Cr leaching from CCA-treated wood can be compared to the predictions of the Brooks (2003) model for the average rate over the first six months, and for the long-term rates. Tables 2.2, 2.3, and 2.4 compare the leaching rates predicted by the two models for Cu, As, and Cr, respectively. The predictions of the Brooks (2003) model were developed by inserting the test conditions used by Lebow et al. (1999) into the Brooks (2003) CCA model described above, and integrating the predicted release over six months to obtain a daily average rate over six months.

Table 2.2. Comparison of daily average Cu leaching rates ($\mu g/cm^2$ day) from CCA-treated wood predicted by Lebow et al. (1999) and Brooks (2003)[a]

CCA loading (kg/m^3)	Water salinity (ppt)	First 6 months		> 6 months	
		Lebow et al. (1999)	Brooks (2003)	Lebow et al. (1999)	Brooks (2003)
16.0	0	0.30	0.57	0.020	0.53
	34	0.82	1.23	0.34	1.19
33.5	0	0.40	0.54	0.030	0.49
	23	1.42	0.97	0.81	0.93
	34	0.90	1.20	0.62	1.16

a. The Lebow et al. (1999) model for pilings was used. Water pH and temperature are not specified in Lebow et al. (1999); a water temperature of 21°C (ambient room temperature), and pH of 6.5 at 0 ppt salinity and 8.0 at 23 and 34 ppt salinity, were assumed for input into the Brooks (2003) model.

Table 2.3. Comparison of daily average As leaching rates ($\mu g/cm^2$ day) from CCA-treated wood predicted by Lebow et al. (1999) and Brooks (2003)[a]

CCA loading (kg/m^3)	Water salinity (ppt)	First 6 months		> 6 months	
		Lebow et al. (1999)	Brooks (2003)	Lebow et al. (1999)	Brooks (2003)
16.0	0	0.24	0.12	0.090	0.074
	34	0.071	0.12[b]	0.020	0.074[b]
33.5	0	0.43	0.12	0.14	0.074
	23	0.26	0.12[b]	0.080	0.074[b]
	34	0.18	0.12[b]	0.060	0.074[b]

a. The Lebow et al. (1999) model for pilings was used.

b. Arsenic leaching rate is not a function of salinity or retention in the Brooks (2003) version of the CCA leaching model, but Brooks plans to change this in the next version of this model (Brooks, 2006).

Table 2.4. Comparison of daily average Cr leaching rates (μg/cm^2 day) from CCA-treated wood predicted by Lebow et al. (1999) and Brooks (2003)[a]

CCA loading (kg/m^3)	Water salinity (ppt)	First 6 months		> 6 months	
		Lebow et al. (1999)	Brooks (2003)	Lebow et al. (1999)	Brooks (2003)
16.0	0	0.049	0.025	0.0067	0.011
	34	0.016	0.016	0.0033	0.011
33.5	0	0.044	0.022	0.0067	0.011
	23	0.022	0.016	0.0067	0.011
	34	0.016	0.015	0.0033	0.011

a. The Lebow et al. (1999) model for pilings was used. Water temperature is not specified in Lebow et al. (1999); a water temperature of 21°C (ambient room temperature) was assumed for input into the Brooks (2003) model.

The predictions of daily average Cu leaching rates over the first six months of exposure are similar between the two models, with all paired comparisons within a factor of two of each other (Table 2.2). Except for wood with a high CCA retention in water of 23 ppt salinity, the Lebow et al. (1999) model predictions are all somewhat less than those of the Brooks (2003) model, although the differences are not substantial. For long-term Cu leaching, the Brooks (2003) model predicts higher daily average rates than the Lebow et al. (1999) model under all conditions that were modeled. The largest difference is in water of 0 ppt salinity, but it should be noted that the Lebow et al. (1999) test at 0 ppt salinity was conducted in deionized water, and the removal of all ions from the water likely would lead to an increase in Cu leaching from the wood over the long-term (Lebow et al., 1999; Hingston et al., 2001).

The predicted daily average leaching rates of As over the first six months are either similar between the two models or higher in the Lebow et al. (1999) model (Table 2.3). The greatest difference is for CCA at high retention in 0 ppt salinity water, where the daily average rate predicted by the Lebow et al. (1999) model is approximately four times higher than the Brooks (2003) prediction. Beyond the first six months, the daily average leaching rates predicted by the two models are similar, with most less than a factor of two apart. The leaching rate of arsenic in the Brooks (2003) CCA leaching model is only dependent on the length of immersion. According to Brooks, arsenic leaching in his earlier (1997) model is sensitive to both salinity and temperature, and he intends to integrate both models; however, only the most recent model (Brooks, 2003) is addressed in this document (Brooks, 2006).

For Cr, daily average leaching rates over the first six months are all similar to each other, falling within a factor of two. Beyond six months, the predicted rates are also generally similar, although the predicted rates at 34 ppt salinity for both high and low CCA retention are approximately three times higher in the Brooks (2003) model.

The comparison of the two models demonstrates that under the test conditions modeled (which were the test conditions of the Lebow et al., 1999 study), the two models produce generally similar results. Most predictions are within a factor of two of each other, with exceptions occurring for some daily average rates for the period beyond the initial six months of leaching. In developing his model, Brooks (2003) included the data from the Lebow et al. (1999) study, so the two models are not completely independent of each other. Nevertheless, the Brooks (2003) model includes data from other laboratory leaching studies, and the two models differ in their underlying formulations. The consistency between the predictions of the two models, at least within the range of parameters tested, enhances the credibility of the models.

However, it should be noted that the model of Lebow et al. (1999) does not attempt to predict the leaching rate during the first days and weeks after immersion, when the rate is highest. Rather, it predicts the average leaching rate over the first six months. Therefore, the leaching rate predicted by the Brooks (2003) model for the initial days after immersion cannot be compared against the Lebow et al. (1999) model.

Other leaching models

Van Eetvelde et al. (1994) developed an empirical model to predict the leaching of metals from wood treated with CCA in accordance with Dutch prestandards for building materials. The leaching tests were conducted using different species and dimensions of wood, fixation methods, and water pH. However, the document describing the details of the model formulation was not available at the time of this draft report preparation.

Waldron et al. (2004) prepared a leaching model for Cu, As, and Cr releases from wood treated with CCA, ACQ, and CA-B. Unlike the other empirical-based models discussed in this section, this model is based on diffusion of the metals within the wood and transfer out of the wood into the surrounding water. Diffusion of preservative within treated wood from the inner to the outer portions has been documented (Hingston et al., 2001). Thus, the model is based on the application of diffusion theory applied to the metals contained within the wood. The authors propose several advantages of leaching models based on diffusion theory over empirical models, including an enhanced ability to extrapolate to different conditions beyond those tested; and the ability to account for variables such as wood specimen dimensions, movement across grain versus with grain, temperature, wood species, and preservative formulation and retention.

The diffusion-theory based model provides an alternative approach to the empirical models, and has the potential capability to predict leaching under a wide range of treatment and environmental conditions. The model was presented at the Environmental Impacts of Preservative-Treated Wood Conference, February 8-11, 2004, in Orlando, Florida, and a draft manuscript is available from the authors. The model is still in a developmental stage and has not yet been published in the peer-reviewed literature, and has not been tested or evaluated in laboratory or field leaching studies.

2.2.2 Conclusions

Models of the rate of metal leaching from treated wood have provided the basis for much of the work conducted to date on evaluating the potential for treated wood to cause environmental harm. The models that have been used to date are empirical models that are based on the results of laboratory leaching studies. The primary models evaluated in this section are the empirical models of Brooks (1997a, 1997b, 2003, 2005) and Lebow et al. (1999). Other models mentioned previously are in various stages of development and are not yet useful for this review.

One aspect of model evaluation is the degree to which the model adequately captures the data upon which the model is based. An evaluation of the Brooks models for metal leaching from CCA shows that while the models appear to capture the overall trend in the observed data fairly well, there is substantial variability about the 1:1 observed:predicted line, indicating that the models do not capture the full range of variability that occurs in different leaching tests. For CA-B, the Brooks (2005) model appears to provide a reasonably accurate set of predictions against the underlying data, although long-term leaching rates are systematically underestimated by the model by a factor of approximately 1.5 to 2 times.

A comparison of the CCA leaching models of Brooks (2003) and Lebow et al. (1999) shows that the two models produce predicted leaching rates for Cu, As, and Cr over the first six months and beyond that are in reasonably good agreement with each other. The two models are not independent of each other, since the Brooks (2003) model includes the data of the Lebow et al. (1999) study in its development, but the reasonable agreement between the two models does provide some measure of confirmation.

2.3 Applying the Results of Laboratory-Derived Leaching Study Results to Field Conditions

The studies and empirical models of metal leaching from wood treated with preservatives described in Section 2.2 are based on laboratory studies of metal leaching where preservative loading, fixing, and leaching are conducted under controlled conditions that typically do not capture all of the variables at work in the environment. The application of these predictions to field conditions involves uncertainty of various kinds, many of which are difficult to assess quantitatively. This section describes the major uncertainties involved in applying laboratory-derived predictions of metal leaching from preserved wood to field conditions. The factors that can cause differences between the results of laboratory studies and leaching in field conditions are listed and summarized in Table 2.5, and described in more detail below.

Table 2.5. Factors potentially affecting the application of laboratory leaching studies to field conditions

Factor	Potential direction of effect	Comment
Preservative application, fixation, and post-treatment	Higher leaching in field vs. lab	Increased variability and decreased control at commercial scale versus laboratory scale.
Physical abrasion, wood borers	Higher leaching in field vs. lab	Abrasion and borers can cause an increase in effective surface area in the field.
Organic acids	Higher leaching in field vs. lab	Organic acids in natural water are not typically included in laboratory studies, and may increase leaching of metals from preserved wood.
Wetting and drying cycles	Higher leaching in field vs. lab	Continual wetting and drying may cause changes to the wood surface that increase leaching.
Production of sawdust and shavings	Higher leaching in field vs. lab	If sawdust and shavings produced during field construction are not contained, leaching from newly built structures would be underestimated based on laboratory studies.
Static or static-renewal laboratory leaching procedures	Higher leaching in field vs. lab	Static or static-renewal test conditions may decrease the concentration gradient in the test chambers. However, available leaching models do not include data from static tests.
Wood fouling	Lower leaching in field vs. lab	Buildup of fouling communities may cause an effective decrease in the wood surface area. Most likely to be relevant at longer exposure times.

2.3.1 Factors affecting the applicability of laboratory study results to field conditions

One of the most important factors that can cause differences in metal leaching between laboratory and field conditions is preservative application and fixation. In laboratory studies, preservative application, fixation, and post-treatment are typically conducted on very small batches of wood using small-scale equipment, and are closely monitored to help reduce variability that could be introduced into the tests from uneven application or fixing. In contrast, wood for commercial use is treated and fixed on large, commercial scales, and the variability in application, retention, degree of fixation, and post-treatment cleanup is much larger than occurs in laboratory studies. For example, the preservative fixation process is complex, and at a commercial scale incomplete fixation is much more likely to occur than in laboratory studies. Likewise, post-fixation treatment to remove excess preservative is likely to be much more closely monitored in small-scale laboratory experiments than at the commercial scale. Since incomplete fixation and excess preservative material produce much higher rates of leaching, especially in the initial days and weeks following immersion, this difference between laboratory studies and in-service conditions is likely to produce much higher leaching rates in the field than what is measured in laboratory studies.

An example of the effect of the application and fixation process on leaching rates is provided in the laboratory leaching data of Brooks (2005) for CA-B. In this study, the Cu leaching rates from a single replicate (out of 27) were observed to be much higher than all other replicates during the first three time periods, 0.5, 1.5, and 2.5 days after immersion, which were monitored. The Cu leaching rates from this single replicate were approximately two times higher than those observed in the other replicates at these sampling times. Brooks (2005) attributed the higher leaching rates in the single replicate to unfixed and excess preservative on the wood surface, and subsequently discarded the data from this replicate in the development of the CA-B leaching model. This example demonstrates both the higher metal leaching rates that can occur from differences in preservative application and fixing, and the tendency of laboratory studies to not include this source of variability. Wood treated and fixed at the commercial scale would most likely include samples such as those observed by Brooks (2005) in the single replicate.

Other differences between laboratory and field conditions that are likely to cause increased leaching in the field include:

- The physical abrasion and impacts of wood borers in the field that can increase the effective surface area of in-service wood or expose deeper grains (Hingston et al., 2001).
- The increased leaching of metals from the wood that can be caused by organic acids (particularly humic and fulvic acids) in natural waters (Lebow, 1996; Hingston et al., 2001).
- The continual wetting and drying cycles that service wood is subject to, such as from tidal flux, which can cause higher metal leaching because of changes to the surface structure of the wood, perhaps because of salt crystal formation (Hingston et al., 2001).
- The production of sawdust and shavings during field construction of structures built from treated wood. Metal leaching from sawdust and shavings occurs at a much higher rate than from whole wood (Brooks, 2003). Where construction does not adequately control the environmental release of shavings and sawdust, the overall metal leaching from a

newly constructed structure will be much higher than that predicted from the application of laboratory studies to the whole wood components of the structure.

An additional factor that could lead to laboratory studies underestimating leaching in the field is that many laboratory leaching studies are conducted using static or static-renewal procedures. Under these procedures, the increase in metal concentrations over time in the ambient water will decrease the concentration gradient across the wood:water interface, thereby decreasing the leaching rate (Hinston et al., 2001). In his development of the empirical leaching models, Brooks (1997a, 1997b, 2003, 2005) excludes static tests from the underlying databases, so this factor should not produce a substantial bias in the models based on laboratory leaching studies.

Other factors may tend to cause a decrease in metal leaching in the field compared to laboratory studies. The factor most commonly cited is the buildup of fouling communities on the wood, which can decrease the effective surface area of the wood exposed to the water (Hingston et al., 2001). However, it should be noted that this factor will have no or little effect during the initial days or weeks after immersion, when the leaching rates of metals are at their highest levels. It can affect the long-term leaching rates, however.

In summary, there are several factors which make it problematic to directly apply leaching rates derived from laboratory studies to field conditions. Most of these factors will tend to result in an increase in leaching in the field compared to that observed in the laboratory, with the most important of these probably being variability in wood treatment, fixation, and post-treatment processing that occurs at the commercial scale. Data with which to evaluate or quantify this difference are unavailable, however.

2.3.2 Field trials of metal leaching from preserved wood

Few studies are available in which metal leaching from wood has been measured in the field (Lebow, 1996; Hingston et al., 2001). Furthermore, the studies that are available monitor the leaching by measuring the amount of metals remaining in the wood, and use those measurements to estimate the amount lost to the environment (Hingston et al., 2001). This methodology is subject to highly variable results because of small errors in the estimate or measurement of the amount of preservative retained in the wood at the outset of the trial (Hingston et al., 2001). Nevertheless, the loss of Cu, Cr, and As from CCA-treated wood has been documented in long-term field trials in aquatic environments, although the results vary substantially (Lebow, 1996). Furthermore, the temporal pattern of the losses in field trials has been observed to be generally similar to that observed in laboratory studies, with loss rates being highest initially, and reducing thereafter (Hingston et al., 2001).

One study on CCA leaching from full-sized pilings in seawater found that after 28 days of exposure, 529 mg of Cu, 60 mg of As, and little or no Cr was released from each square meter of pile surface (Baldwin et al., 1996; as cited in Lebow, 1996). This equates to an average daily leaching rate over the 28 days of 1.89 µg Cu/cm^2 day and 0.21 µg As/cm^2 day. These rates are slightly higher (within a factor or two or less) or approximately equal to the average rates over six months predicted by the leaching models of Lebow et al. (1999) and Brooks (2003) (described in Section 2.2), which predict average leaching rates in seawater over the first six months of approximately 0.9 to 1.2 µg Cu/cm^2 day and 0.07 to 0.18 µg As/cm^2 day. The average daily rates over the first 28 days are expected to be higher than over the first six months since the leaching rates are much higher in the first few days and weeks after immersion. Therefore, the results of this single field leaching study are consistent with the predictions of the empirical models that are based on laboratory leaching studies. However, no other field leaching trial studies were available for additional comparison.

Other field trials have been conducted using small pieces of wood immersed in saltwater. The results of these field trials may be less indicative of leaching of commercial applications of treated wood because of the higher surface area to volume ratio of the small pieces of wood, as well as the reduced physical abrasion that the small pieces may undergo. Furthermore, the studies were monitored by analyzing the metals remaining in the wood at a specific point in time, which makes direct comparison to the results of laboratory leaching studies difficult. Nevertheless, the results of the field studies of small pieces of treated wood confirm that Cu, As, and Cr leach from wood treated with CCA, and that Cu losses are typically greater than As or Cr losses (Lebow, 1996).

2.4 Predicting Environmental Concentrations of Metals from Leaching Models

Another key component in conducting evaluations of the potential for treated wood to cause environmental impacts to NOAA trust resources, in addition to leaching studies and models, is the application of the results of the leaching studies and models to predict resulting environmental metal concentrations. Many factors affect the relationship between metal leaching rates and subsequent environmental concentrations, most of which are highly site-specific. Nevertheless, this section summarizes the models that have been used to predict environmental metal concentrations that result from the leaching of metals from treated wood.

Water column concentrations present in the laboratory leaching study exposure tanks can be derived from the leaching studies presented in Section 2.2 if the leaching rate, surface area of the wood, and volume of leach solution are known. For instance, using data from Lebow et al. (1999), the water column concentrations of Cu in a CCA leaching test for a set of three pilings with a retention of 41.9 kg/m^3 in a closed container of synthetic seawater is calculated to be approximately 2,237 μg/L on day 1, 1,060 μg/L on day 9, 828 μg/L on day 31, and 349 μg/L on day 186. These estimates indicate that in very small, spatially finite systems, without flow, the concentrations of Cu can be quite high.

However, Cu concentrations that build up in the laboratory leaching studies are not representative of concentrations expected in the environment because of the dilution that occurs in the environment. Because these conditions are not representative of actual environmental conditions, it is necessary to develop methods that more accurately predict concentrations found under the dilution that occurs in field flow conditions.

Several transport models that estimate concentrations of contaminants in the water column and sediments resulting from the use of treated wood have been developed. Types of transport models that simulate the advection and dispersion of contaminants in the environment vary from simplistic conceptual models to extremely complex and computationally intensive codes based on high-level fluid mechanics and kinetics research. The models described in this report range from a simple box plume model that estimates mean concentrations based on the ratio of the mass of contaminant leached to the volume of receiving water, to a more rigorous model based on an advection diffusion partial differential equation. Regardless of the particular methods used to develop each model, the underlying goal is the same – to determine the degree and spatial extent of dilution of the leached metals that occur under realistic environmental conditions.

2.4.1 Poston et al. (1996) models of copper concentration in the water column

Poston et al. (1996) developed two models to provide estimates of environmental contaminant concentrations that may result from metals leaching from wood pilings treated with ACZA preservatives: a hydrologic transport model based on a dynamic river plume, and a box model that uses flow, time, and distance to determine mean concentrations of copper. Both models use daily estimates of leachate mass from the treated wood based on the Cu leaching model of Brooks (1994), which has subsequently been updated in Brooks (1997a).

In the Poston et al. (1996) models, simulation results of 2.0 μg/L or greater are considered to exceed a threshold level that may have adverse impacts. This was determined from copper toxicity tests and the best professional judgment of the Northwest Region of the NMFS that a 7.0 μg/L level (the third dilution below the LC_{50} for dissolved copper) would not have a response level that is statistically different than the controls and an assumed background concentration of 5 μg/L.

Hydrologic model based on characterization of a dynamic river plume used for Cu leaching scenarios

The Poston et al. (1996) hydrologic model is based on characterization of a dynamic river plume. Two types of ACZA treated structures were evaluated: shoreline structures and midstream fender structures.

All simulations assumed that the release of copper from a group of wood pilings is modeled using the analytical solution for the diffusion of a line source in a river. The solution is derived from the advection-diffusion equation. In this model, river flow is assumed to be uniformly distributed across the channel because site-specific conditions were not available. Poston et al. (1996) used the equation that gives the concentration distribution downstream from a line source of a contaminant that depends on current velocity (determined by discharge, river width, and mean depth), lateral dispersion, friction velocity, and frictional slope. The frictional velocity is calculated from the Manning equation for uniform flow. The lateral mixing coefficient, which controls the rate at which the plume spreads across the river, is computed from the relationship (Fischer et al., 1979):

$$\varepsilon_T = 0.6\ du^*$$

where:

- d = average river depth
- u^* = frictional velocity.

Shoreline structure scenarios

Model runs were conducted for several shoreline piling configurations. The mean depth was 30 ft and the assumed piling diameter was 30 cm. Concentrations of copper were estimated at a unit distance achieved in 24 hours based on the average current velocity and at fractions of that unit distance. Simulations were performed at mean current velocities of 40.6, 10.0, 1.0, 0.5, and 0.3 cm/s, and pH values of 7.2, 7.5, and 8.0. Estimates of lateral dispersion occur in increments of 50 ft moving in one direction out from the shoreline. The Cu leaching rate was estimated from the Brooks (1994) models.

The first model scenario, a large footprint (i.e., low piling density) of 100 pilings, yielded a maximum estimated copper concentration of 1.4 µg/L on day 1 at a distance of 4 ft from the piling, at a flow rate of 0.3 cm/s.

Higher piling density configurations consisting of 24, 100, and 350 pilings were modeled in subsequent scenarios. The 24 piling simulation did not exceed the threshold value of 2.0 under any flow conditions at pH 7.2. The 100 pilings scenario produced estimated concentrations greater than 2.0 under the lower flow rates of 0.3 and 0.5 cm/s. The lateral and downstream extents of these concentrations were 46 m and 123 m, respectively, at the flow rate of 0.3 cm/s and 46 m, and 14 m at a flow rate of 0.5 cm/s. The 350 piling scenario also exceeded threshold concentrations at flows of 0.3 cm/s and 0.5 cm/s, but also exceeded threshold concentrations at 1.0 cm/s.

These results indicate that modeled concentrations are affected by the density of the pilings in the configuration. The results also suggest that flow is a significant variable that affects both the concentration and the areal extent of the plume.

Fender scenarios

The same flow and pH conditions used in the shoreline simulations were run for the fender structures. Fender structures, however, are placed in the middle of channel and are conceptualized as point sources with a width of 2.5 ft each. In this case, lateral dispersion occurs in two dimensions from the mid-channel position of the fender structure; estimates of lateral dispersion were calculated in 100-foot increments.

Scenarios were modeled for 24, 100, and 350 pilings. The assumed specific configuration was not provided. As with the shoreline simulation, the 24 piling model run did not result in estimated Cu concentrations exceeding 2 µg/L at a distance greater than 4 ft downstream of the structure. The 100 piling simulation produced estimated Cu concentrations greater than 2 µg/L for flow rates of 10 cm/s and lower. All flow rates tested for the 350 piling simulation resulted in estimated concentrations greater than 2 µg/L.

In the fender scenarios, lateral dispersion had a more significant effect on the width of the plume as it was not constrained by the shore. Results showed that as the flow rate decreased, the extent of later dispersion increased.

Box plume model

Poston's second model estimates concentrations of total Cu for one day units of time in a hypothetical box plume using flow, time, and distance. The source of Cu in the simulations is a vertical "footprint" perpendicular to the current representing an assumed number of pilings (24, 100, or 350 compressed into a plane of a given area). Several footprint areas are modeled to simulate differing configurations of the pilings.

The copper loss rate used as input to the model is the same equation used in the hydrodynamic model (Brooks, 1994). The area of the box plume is determined by the surface area of the vertical plane of the source and the flow distance (determined by the current velocity) traveled in one day. Lateral mixing is not considered in this model. Four current velocities, 0.3, 0.5, 1.0, and 10 cm/s, were simulated. Mean copper concentrations were estimated as the amount of contaminant leached from the exposed surface area of all pilings in the scenario in a 24-hour period, divided by the resulting volume of water in the rectangular plume.

Results showed that as flow rate decreased, mean Cu concentrations increased. For the 350 piling scenario at a pH of 7.5, all Cu concentrations exceeded the 2.0 µg/L threshold at flows of 1 cm/s and less; and at a pH of 8, all modeled concentrations exceeded the threshold at velocities of 0.5 cm/s and less. For the 100 and 24 piling scenarios, some of the modeled concentrations exceeded the thresholds at velocities of 1 cm/s at pH values of both 7.5 and 8. Copper concentrations also varied with the size of the footprint. For a constant number of pilings, predicted concentrations decreased with increasing footprint area (i.e., a less dense configuration of pilings).

2.4.2 Brooks (1997b) model of copper concentrations in water and sediments

Brooks (1997b) developed a spreadsheet model to predict concentrations of copper in surface water and sediment resulting from leaching from CCA-treated wood placed in the aquatic environment. The model predicts both water column concentrations and near-field sediment concentrations associated with the use of CCA-treated wood. The input of daily leachate mass appears to be calculated from the Brooks (1997b) model, which was subsequently updated in 2003.

Sixteen parameters are required to run the model:

- Preservative retention (kg/m^3)
- Average piling radius (cm)
- Treated wood age (days)
- Salinity (ppt)
- Settling velocity (0.05 for silt, 0.0005 for clay)
- Average maximum tidal speed (cm/sec)
- Steady state current speed (cm/s, measured at slack tide)
- Marine sediment copper quality standard parts per million (ppm)
- Maximum marine sediment impact zone Cu standard (ppm)
- Freshwater chronic copper standard
- Water hardness (ppm $CaCO_3$)
- Marine water copper standard
- Sediment density (g/cm^3)
- Bulkhead length (cm)
- Bulkhead board width (cm)
- Average water depth (cm).

Recommended input parameters are provided for preliminary evaluations or for use when site-specific information is unavailable.

Water column copper concentrations

The algorithm used in the model to estimate water column copper concentrations associated with CCA treated pilings is:

$$\text{Concentration} = 0.0276\, exp(-0.048*t + 0.02s)*[0.8462 + \ln(0.71*r)]*\, R_p/[(1{,}800*0.0651*V_{max} + 1800V_{ss} + R_p)^2 - R_p^2]$$

where:

t	= time in days
s	= salinity
r	= retention
R_p	= piling radius (cm)
V_{max}	= maximum tidal current speed
V_{ss}	= steady state current speed.

Results from simulations estimate that tidally driven water currents of 2.5 cm/s will cause dispersion of leaching copper into a circle of 586 cm in radius. The dispersion model predicts that for a single piling 30 cm in diameter that leaches 2.86 µg/cm^2-d, the maximum concentration of 34 ng/L of Cu would occur within a half-hour of slack tide on the day of installation.

This model is not valid for small volume closed water body conditions as it assumes that the volume of the water body is large in comparison with the total amount of copper lost from the structure.

Prediction of near-field sediment copper concentrations

The algorithm used in the model to estimate sediment copper accumulation associated with CCA treated pilings is:

$$\text{Copper accumulation} = (1 + V_{model}/10)*\exp(-0.048t)*0.51*\exp(0.02s)*0.65[0.8462 + \ln(0.71r)]*R_p/[(d + R_p)*(V_{model}/V_{vert})]$$

where:

t	= project age, in days
s	= salinity (ppt)
r	= CCA retention (kg/m^3)

R_p = piling radius (cm)
d = distance (cm) from piling perimeter where sediment concentration is measured
V_{model} = model water velocity = $V_{ss} + 0.64V_{max}$
V_{vert} = silt-clay settling velocity (cm/s).

Model results for an installation of 30-cm diameter pilings, treated to a retention of 40 kg/m^3 with poor flushing and a maximum tidal speed of 2.5 cm/s, predicted sediment concentrations of 1.53 $\mu g/cm^2$ within 5 cm of a pile. The predicted concentration decreased to 0.49 $\mu g/cm^2$ at the midpoint between two pilings separated by 2 m.

This model does not address metal loading to sediment that is associated with abrasion of CCA-treated wood. Brooks suggests that pieces of abraded wood could be water logged and sink to the bottom, effectively increasing copper accumulation in the sediments. Another limitation of this model is that it assumes that all copper delivered to the sediments remains in the upper two centimeters of substrate. Dispersion resulting from mechanical disturbances or burial due to high sediment accretion rates are not considered in the model.

2.4.3 Conclusions regarding models to predict environmental metal concentrations from leaching

The development of models that describe the transport of treated wood leachates in the environment is an important task that requires further attention. The models presented here include the models of Poston et al. (1996) and Brooks (1997b). The aquatic systems that these models simulate are highly complex systems that are difficult to quantitatively describe. Therefore, the models and results presented need to be considered in this context.

Without further validation, these models should not be relied upon for accurate representation of actual concentration fields and spatial extent of the plumes because of the highly site-specific nature of individual scenarios. The assumptions required to run these models can have a high degree of uncertainty in them because many of these factors are highly variable. For instance, tidal currents are very complex and influenced by many highly variable factors. Turbulence, the main process by which mixing occurs in these systems (diffusion takes place but is negligible in comparison), is a chaotic three-dimensional process that is very difficult to model. Many simplifying assumptions are made in the development of these models, and furthermore, the leach rates input into the model are model results themselves with their own set of uncertainties associated with them. One way to further investigate and validate these models is to compare the predicted values to observed values. This would require that sampling data from actual installations be collected.

Despite the uncertainties associated with these models, they do have value in qualitatively describing the transport and mixing of the leachate in the environment and the potential effects on biota. For instance, all of the models point to flow rate as a critical variable. As flow rate increases, concentrations of copper are diluted more efficiently. These results of these models suggest that concentrations which may be harmful to biota will accumulate at low velocities. The potential for adverse effects is greatly intensified in systems like canals with low current velocities and limited flushing. The models also suggest that the particular configuration, or more specifically, the piling density of the configuration, can affect concentrations seen in the environment. Lower density configurations appear to be less likely to produce concentration profiles that may adversely affect biota, however, if a threshold level is exceeded due to low flow conditions or other variables, the spatial extent of the effect will be greater.

2.5 Conclusions

The modeling of the metal leaching rates from treated wood and the resulting environmental metal concentrations are key components in the evaluation of environmental risks from treated wood structures. Our conclusions from a review and evaluation of the available information and models on metal leaching and environmental concentrations follow:

- Laboratory studies have shown that many factors affect the rate of metal leaching from treated wood. Some factors are related to the wood treatment, fixation, and post-treatment process, and these factors can have a dramatic influence on the rate and total amount of metals leached. Preservative that is not completely fixed or is left as excess on the treated wood leaches rapidly and substantially from treated wood.

- Environmental factors also affect leaching rates, including ambient water salinity, temperature, and pH, although these factors are somewhat poorly understood and do not produce consistent effects across the different preservative solutions and metals that leach from the solutions.

- Time is a crucial factor in leaching rates. Metal leaching rates are much higher initially, typically within the first few days or weeks, as the more labile or poorly bound metals are leached. Leaching rates then decrease to a more steady state rate that reflects the longer-term exchange and diffusion of the more tightly bound metals.

- Several empirical models have been developed to predict the metal leaching rates from treated wood. These models are based on results from laboratory leaching studies. In general, there is substantial variability between the predictions of the models and the observed leaching rates in laboratory studies, but for the most part there is no or little bias in model under- or overpredictions. The models appear to capture the overall trends in leaching rates reasonably well, despite the variability associated with individual observations.

- In applying the results of the models that are based on laboratory leaching studies to field conditions, there are several factors that make the direct application of laboratory results to field conditions problematic. Most of these factors will tend to result in an increase in leaching in the field compared to that observed in the laboratory, with the most important factors probably being variability in wood treatment, fixation, and post-treatment processing that occurs at the commercial scale. Data to evaluate or quantify this difference are unavailable, however.

- Several models have been developed to predict environmental concentrations of metals in surface water and sediment based on assumptions regarding modeled leaching rates and environmental parameters such as water flow, surface area of treated wood, and sediment settling and movement. The assumptions of these models are, by definition, highly site-specific, and applying them to predict concentrations at a generic or regional level is problematic. The models are useful in providing information on site-specific predictions where site-specific conditions are known, and in evaluating the relative importance of different environmental variables in determining environmental concentrations of metals that result from leaching.

3. Toxicity of Wood Treatment Chemicals to Aquatic Organisms

As described previously, waterborne wood preservatives contain the metals copper, arsenic, zinc, and chromium and these chemicals can be leached from treated wood into the surrounding environment. This chapter discusses the toxicity of these chemicals. We focus on copper because of the higher leaching rates of this metal from treated wood (see Chapter 2), but provide relevant information on the other metals for completeness.

The potential toxicity of wood treatment chemicals to aquatic biota is dependent on the route of exposure. There are three potential exposure pathways: waterborne exposure to chemicals leached from wood and dissolved in the water column; exposure of aquatic biota to chemicals leached from wood that subsequently accumulate in sediments (and sediment pore water); and dietary exposure (e.g., accumulation of leached chemicals in prey items that are subsequently consumed by other aquatic biota; Figure 3.1). It is clearly beyond the scope of this document to provide a comprehensive discussion of the toxicity of each of the above-mentioned compounds to all (or even a subset of) the NOAA trust species. Rather, for each exposure route, we present and describe toxicity reference values that can be used to evaluate the potential toxicity of wood treating chemicals.

3.1 Water Column Exposures

The U.S. EPA has established concentrations of the above metals designed to be protective of aquatic organisms. These Aquatic Life Criteria (ALC) have been developed to protect aquatic organisms in the 5th percentile of sensitivity for each chemical.[1] There are two criteria for each ALC: the criterion maximum concentration (CMC) and the criterion continuous concentration (CCC). The CMC is designed to protect aquatic organisms from short-term, or acute, exposures and permits a relatively higher concentration for a relatively shorter period of time than the CCC, which is designed to protect aquatic organisms from ongoing or chronic adverse effects. The CMC is based on 1-hour average concentrations, whereas the CCC is based on 4-day average concentrations. Table 3.1 presents the ALC for As, Cu, Cr, and Zn in fresh- and saltwater. It should be noted that because of the relative paucity of data on the toxicity of many chemicals

1. Thus, if the first percentile contains the most sensitive, and the 99th percentile contains the least sensitive aquatic organisms, a concentration that is designed to protect organisms in the 5th percentile should protect 95% of aquatic species.

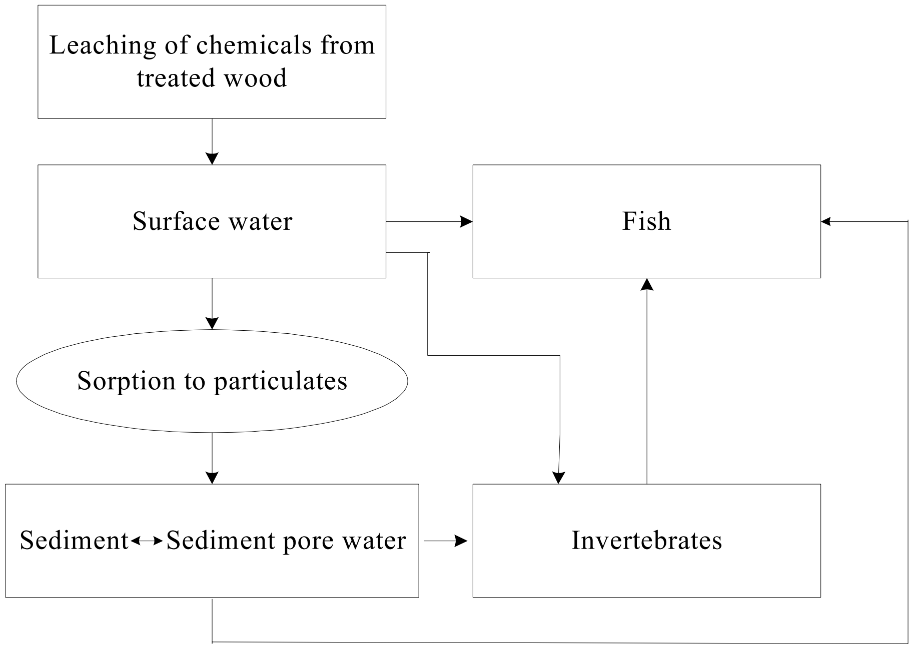

Figure 3.1. Potential exposure routes to aquatic organisms from wood treating chemicals.

over a range of salinity, the U.S. EPA (2002) provided the following guidance for applying freshwater and saltwater criteria:

1. For water in which the salinity is equal to or less than 1 ppt 95% or more of the time, the applicable criteria are the freshwater criteria.

2. For water in which the salinity is equal to or greater than 10 ppt 95% or more of the time, the applicable criteria are the saltwater criteria.

3. For water in which the salinity is between 1 and 10 ppt, the applicable criteria are the more stringent of the freshwater or saltwater criteria. However, alternative freshwater or saltwater criteria may be used if scientifically defensible information and data demonstrate that, on a site-specific basis, the biology of the water body is dominated by freshwater aquatic life and that freshwater criteria are more appropriate, or conversely, the biology of the water body is dominated by saltwater aquatic life and that saltwater criteria are more appropriate.

To aid in the understanding and interpretation of these threshold criteria values, additional information for individual wood treating chemicals is provided below.

Table 3.1. U.S. water quality criteria for the protection of aquatic life (ALC) for water soluble chemicals used in treating wood. Criteria values are presented for a hardness of 50 mg/L (as $CaCO_3$) where criteria are hardness dependent.

Chemical	Freshwater CMC (µg/L)	Freshwater CCC (µg/L)	Saltwater CMC (µg/L)	Saltwater CCC (µg/L)
Arsenic	340	150	69	36
Copper	7.0[a]	5.0[a]	4.8	3.1
Copper (2003)	BLM[b]	BLM[b]	3.1	1.9
Chromium III	323	42	None (850)[c]	None (88)[d]
Chromium VI	16	11	1,100	50
Zinc	65[a]	65[a]	90	81

a. Criteria are hardness-dependent. Criteria values calculated using site-specific hardness based on the equations presented in U.S. EPA (2002).
b. Criteria developed using site-specific chemistry and the Biotic Ligand Model (BLM).
c. No saltwater CMC. As a proxy, we report the lowest reported LC_{50} from the U.S. EPA database (Lussier et al., 1985) divided by a factor of 2. See text for additional details.
d. No saltwater CCC. As a proxy, we report the lowest reported chronic value from the U.S. EPA database (Lussier et al., 1985) divided by a factor of 2. See text for additional details.
Source: U.S. EPA (2002), except as noted.

3.1.1 Copper

Copper has been the subject of much in-depth toxicity research. Standardized acute toxicity test results have been reported for more than 60 genera of North American freshwater organisms and more than 40 genera of marine organisms. In both freshwater and marine environments, the most sensitive species tested are typically invertebrates (see Appendices A and B). In freshwater environments, these are typically cladocerans (*Daphnia*, *Ceriodaphnia*, *Bosmina*) and embryonic/larval forms of freshwater mussels. In marine environments, embryonic and larval stages of mollusks (oysters, mussels, clams) appear to be among the most sensitive organisms.

Because the toxicity of copper (and many other metals) has been shown to be related to some water quality parameters, ALC have been expressed as a function of water hardness (see Table 3.1). This adjustment is a relatively simplistic approach to account for other chemical characteristics of waters, including pH, alkalinity, ionic concentrations of other cations and total ionic strength, all of which co-vary with hardness to some extent.

In U.S. EPA's draft revision of the copper ALC (U.S. EPA, 2003), the Agency proposed a procedure for use in establishing site-specific copper criteria in freshwater based on the concentrations of major chemical ions as well as biosolid organic carbon (each of which can influence the aqueous toxicity of copper). A computer model, the BLM, is used to calculate site-specific criteria based on site-specific water chemistry (U.S. EPA, 2003). The complexity of the model makes it impossible to present a simple equation or single criterion value for comparison with ambient copper concentrations. However, for generally conservative, first order estimates, the previous criteria (U.S. EPA, 2002) shown in Table 3.1 are probably lower than values that would be generated using the BLM and serve as relevant evaluation thresholds. At this time, the scientific applicability of the BLM for marine situations is not known.

In general, copper toxicity typically decreases with increasing salinity, with the relationship between salinity and copper toxicity being species-dependent (U.S. EPA, 2003). Although salmonids are heavily represented in the U.S. EPA's freshwater toxicity database for copper (see Appendix A), there are relatively little data on the toxicity of copper to salmonids in saltwater (see Appendix B). Holland et al. (1960) reported a 5-d LC_{50} for 4.1 cm pink salmon (*Oncorhynchus gorbuscha*) of 563 µg Cu/L at 16.6 ppt salinity. Dinnel et al. (1989) reported a 96-h LC_{50} value for coho salmon (*Oncorhynchus kisutch*) smolts of 601 µg Cu/L at a salinity of 28.6 ppt. Comparing these values with the species mean acute 96-h LC_{50}s of 40.87-108.10 µg Cu/L in freshwater for five species in the genus *Oncorhynchus* (data in Appendix A) indicates that copper is potentially less toxic to salmonids in saltwater than in freshwater. However, there are little data to evaluate the relationship between copper toxicity and salinity in estuarine environments where salinities often fluctuate considerably and where migrating salmonids may be undergoing natural osmoregulatory changes. Per the U.S. EPA guidance described above, however, application of the more stringent freshwater criteria may be more appropriate for such estuarine environments.

Notwithstanding the above, several studies have demonstrated the potential for behavioral avoidance effects in salmonids at copper concentrations at or below criteria levels (see Appendix C). For example, Lorz and McPherson (1976) observed a decrease in successful migration following a one-week exposure to a 50 mg/L hardness-normalized copper concentration of 2.7 µg/L. Copper avoidance has also been observed by Sprague (1964) and Giattina et al. (1982). Hansen et al. (1999a) demonstrated behavioral avoidance at lower copper levels by chinook salmon and rainbow trout (*Oncorhynchus mykiss*) exposed to copper in a series of laboratory studies. Both salmon and trout could detect and avoid dissolved copper at threshold levels between 0.7 and 2.8 µg/L. Chinook salmon and rainbow trout also differed in their ability to detect copper after acclimation for 25-30 days at 2 µg/L copper. Acclimated rainbow trout continued to strongly avoid water containing Cu at levels of 4-24 ppb, but acclimated chinook salmon did not avoid water containing Cu at any concentration (2-24 ppb were tested), indicating a loss of ability to detect Cu after prolonged exposure to low levels. However, in a companion paper, Hansen et al. (1999b) showed significant olfactory neurotoxicity to copper-exposed chinook salmon. Therefore, it is possible that the "acclimation" response represented loss of olfactory function (which may have ecological consequences in field settings) in chinook salmon, rather than a true "acclimation" to the copper. The avoidance studies suggest that fish may avoid copper, potentially causing habitat loss/displacement, or avoidance of dock and pier structures that may provide *refugia* from predators. However, the strength of this response in field settings in which fish may be presented with numerous behavioral cues and, in some cases, gradients of copper-contaminated waters, is unknown.

In addition to observed avoidance responses, recent mechanistic research has demonstrated that exposure to sublethal concentrations of copper can be toxic to the olfactory systems of salmonids (and other fishes). Because salmonids rely on olfaction for a number of important biological functions (e.g., fright responses and predator avoidance, migration, reproductive synchronization), olfactory toxicity may pose risks of higher-order biological effects. Hansen et al. (1999b) observed significant reductions in olfactory neuroreceptors in short-term copper exposures to chinook salmon and rainbow trout. Baldwin et al. (2003) and Sandahl et al. (2004) observed neurotoxic effects on olfactory receptors of juvenile coho salmon and copper concentrations lower than chronic water quality criteria (CCC). Inhibition of olfactory receptor responses was not found to be related to water hardness and occurred at dissolved copper concentrations as low as 1 µg/L and at exposure durations < 1 hour (Baldwin et al., 2003). Similarly, Carreau and Pyle (2005) found that exposure to sublethal copper during embryonic development resulted in a significant reduction in the chemical alarm response of fathead minnows (*Pimephales promelas*). Beyers and Farmer (2001) observed a failure of copper-exposed Colorado pikeminnow (*Ptychocheilus lucius*) to demonstrate a fright response to a chemical stimulus.

Overall, the U.S. EPA ALC appears to be appropriate thresholds for use in evaluating the potential effects of Cu released from treated wood, at least for acute lethality. In certain settings, however, avoidance responses or olfactory neurotoxicity may occur at sub-criteria concentrations, even following brief periods of exposure. The nature and ecological implications of such responses is uncertain without site-specific evaluation, but the potential for neurotoxicologically mediated behavioral changes resulting in adverse biological consequences (e.g., increased susceptibility to predation, impaired migration responses) cannot be discounted. Finally, it should be noted that because olfactory responses do not appear to be mediated by hardness in the same manner as lethality responses, it is possible that the degree to which CMC and CCC may underpredict olfactory neurotoxicity may be greater in higher-hardness waters.

3.1.2 Chromium

Chromium can exist in two valence states in natural waters: trivalent chromium [Cr(III)] and hexavalent chromium [Cr(VI)]. Because the two forms have substantially different toxicity to aquatic organisms, water quality criteria are expressed separately for each form. Cr(III) is less toxic than hexavalent chromium Cr(VI).

Chromium (VI)

As with copper, hexavalent typically is more toxic to invertebrates than fish (Appendices D and E). Although only 19 genera of freshwater organisms are represented in the 1984 criteria acute toxicity data, the 11 most sensitive were invertebrates, and the most sensitive fish species was over 1,000 times less sensitive than the most sensitive invertebrate. Brook trout and rainbow trout are the only two salmonids in the database and they are about 2,500 times less sensitive to Cr(VI) toxicity than the most sensitive invertebrate. In seawater, the most sensitive of three fish species tested ranked 10th of 19 genera.

In both freshwater and saltwater, Cr(VI) criteria sufficiently low to protect invertebrates are likely to provide a large margin of protection against acute toxicity to NOAA trust fish species, and indications are that current U.S. EPA ALC shown in Table 3.1 would be protective of salmonids. However, it might be noted that many invertebrates serve as prey for fish. The current chronic criterion for freshwater (11 µg/L) is greater than concentrations reported to produce chronic toxicity in some cladocerans. Chronic effect thresholds for Cr(VI) and five species of cladocerans in the genera *Ceriodaphnia*, *Daphnia*, and *Simocephalus* include values of < 2.5, 6.1, and 6.1 µg Cr(VI)/L. Thus, it could be prudent to provide an additional safety margin if Cr(VI) concentrations > 6 µg/L were persistent over wide areas where such zooplankters represented an important component of salmonid diets.

There appears to be an effect of salinity on Cr(VI) toxicity based upon a series of tests with the grass shrimp (*Palaemonetes pugio*) with decreasing toxicity at higher salinities. The 48-h LC_{50}s from those tests showing toxic concentrations at 20 ppt were from 1.8 to 3.7 times greater than at 10 ppt (Fales, 1978). Also, tests with the brackish water clam, *Rangia cuneata*, indicated a similar trend with an LC_{50} of 210 µg/L in essentially freshwater (< 1 ppt), 14,000 µg/L at 5.5 ppt, and 35,000 µg/L at 22 ppt (Olson and Harrel, 1973). Whether fish, especially salmonids, would follow a similar pattern and magnitude of effect is unknown. At this time, data were not available to suggest that ALC would not be protective of salmonids or other NOAA trust resources.

Chromium (III)

There is no apparent pattern with respect to the relative sensitivity of invertebrates and fish to Cr(III). Of 17 genera of freshwater organisms, aquatic insects were both the most and least sensitive taxa (Appendix F), with LC_{50} values ranging from 2,200 to 71,600 µg/L. Fish LC_{50}s were intermediate, covering a range of 7,053 to 15,630 µg/L, including a rainbow trout LC_{50} of 9,863 µg/L. The current freshwater acute criterion of 323 µg/L appears to be protective of fish in general, and salmonid fish in particular.

The chronic Cr(III) criterion of 42 µg/L is based on protection of rainbow trout and is the threshold toxicity value from a chronic embryo-larval test (Stevens and Chapman, 1984). The most sensitive effect was on survival over the duration of the test. No growth effects were seen or are expected with older life stages that are more tolerant to Cr(III).

At present, there are no saltwater ALC for Cr(III). Acute LC_{50} and chronic effect thresholds have been observed with mysid shrimp at concentrations above 850 and 88 µg/L, respectively (Lussier et al., 1985). Application of the freshwater Cr(III) criteria to saltwater appears to provide a margin of safety of about two based on the mysid test results. There are no data describing tests with Cr(III) with a single species over a range of salinities, so effects of salinity on Cr(III) toxicity are unknown.

Overall, use of the ALC for chromium as evaluation thresholds for treated wood projects appears appropriate. Absent additional data, application of the freshwater ALC for Cr(III) in saltwater would provide a renewable margin of safety.

3.1.3 Zinc

Like copper, ALC for Zn are hardness-dependent (Table 3.1). The most sensitive invertebrates in the U.S. EPA database are cladocerans, genus *Ceriodaphnia*, with species mean acute values of 51 and 174 µg/L for two species[2]; and *Daphnia*, with species mean acute values of 253 and 355 µg/L (Appendix G). The most sensitive fish species listed in the criteria database are striped bass (119 µg/L) and longfin dace (227.8 µg/L). The anadromous salmonids, chinook salmon, rainbow trout, sockeye salmon, and coho salmon have reported LC_{50}s in the U.S. EPA criterion database of 446, 689, 1,502, and 1,628 µg/L, respectively. Based on these data, both CMC and CCC criteria would be protective of salmonid species from mortality. However, Hansen et al. (2002) reported LC_{50} values for both rainbow trout and the federally listed threatened species bull trout (*Salvelinus confluentus*) that were lower than ALC at hardness concentrations of 30 mg/L. These authors concluded that ALC may not be fully protective of sensitive salmonids

2. Unless otherwise specified, reported values are based on a hardness of 50 mg/L (as $CaCO_3$).

in low hardness water, particularly those with relatively low Ca content. Data for saltwater exposures (Table 3.2; also see Appendix H) suggest that the saltwater ALC (90 and 81 µg/L for the CMC and CCC, respectively) likely are protective of salmonids and other NOAA trust species.

Table 3.2. Toxicity of zinc to salmonids at different salinities

Species	Salinity (ppt)	LC_{50} (µg/L)
Rainbow trout yearlings	5.8	27,000
	11.5	64,000
	16.3	64,000
	24.1	34,000
Atlantic salmon smolt	5.8	16,000
	11.5	35,000
	16.3	32,000
	24.1	27,000

Source: Herbert and Wakeford, 1964.

As with copper, zinc avoidance may be a potentially important sublethal effect. Several tests have shown avoidance of zinc concentrations below, or slightly above criteria levels (Sprague, 1964, 1968; Black and Birge, 1980). However, the potential relevance of such responses in field settings is uncertain and would require site-specific evaluation.

Overall, use of the ALC as evaluation thresholds for zinc is reasonable, particularly for lethality responses. However, in some low hardness waters (e.g., < 50 mg/L, as $CaCO_3$), it is possible that criteria are not fully protective of sensitive salmonids and olfactory-mediated responses, and reviewers may wish to conduct more detailed site-specific evaluations.

3.1.4 Arsenic

Arsenic, like chromium, exists in several different valence states in aqueous environments. Unlike chromium, the two arsenic forms, trivalent As(III) and pentavalent As(V), appear to have generally similar toxicities levels based on a limited freshwater database (Appendix I). Invertebrates show greater sensitivity to the acute effects of arsenic exposure than do fish, and by a large margin. The most sensitive invertebrates are 16 and 29 times more sensitive than the most sensitive fish in freshwater and saltwater, respectively (Appendices I and J). Although some toxicity tests have generated adverse effects values that may be lower than the ALC at 150 µg As/L (freshwater) and 36 µg As/L (saltwater) (see Appendix K), these values were derived from tests not sufficiently standardized to have been included in the derivation of the ALC. Overall, it appears that the ALC values represent reasonable thresholds for evaluation of treated wood impacts.

3.2 Sediment Exposure

Sediment exposures can affect trust fish species through dietary toxicity (i.e., toxic effects caused by consuming contaminated prey items), discussed in the next section, and through toxicity to benthic organisms, thereby reducing food availability to fish. This latter mechanism is discussed in this section.

Contaminants accumulated in sediment can be toxic to aquatic biota through direct contact with sediments through movement of the metals from the sediment into the sediment pore water or water column (Burton, 1992), or through consumption of sediments. Effects of sediment contamination in field settings are generally most evident as changes in benthic community composition. Determination of the concentrations of contaminants that produce various levels of adverse effects is complex and often site-specific.

Absent site-specific study, however, sediment threshold values can be used to evaluate potential effects of contaminants on benthic communities. Although no national sediment quality criteria have been developed, various federal, state, and provincial agencies in North America have developed numerical sediment quality guidelines (SQGs), and several groups have conducted sediment toxicity texts to assess the quality of freshwater and marine sediments. The SQGs currently being used in North America have been developed using a variety of approaches. Approaches used include (MacDonald, 1994):

- **Sediment background approach (SBA)**. Sediment contaminant concentrations are compared to concentrations considered to be representative of background conditions. Using this approach, a site would be considered contaminated if the concentration of the contaminant exceeds the background concentration by a significant margin.

- **Spiked sediment bioassay approach (SSBA).** Clean sediments are spiked with known concentrations of contaminants (or mixtures of contaminants) to establish cause and effect relationships with biological responses.

- **Equilibrium partitioning approach (EqPA).** This approach is based on the assumption that the distribution of contaminants between sediment solids and interstitial water is predictable based on known physical and chemical properties. Sediment criteria are then calculated based on water quality criteria.

- **Tissue residue approach (TRA**). Acceptable sediment concentrations for contaminants are established by determining the chemical concentrations in sediments that are predicted or observed to result in acceptable tissue residues. Acceptable tissue residues are usually based on guidelines for the protection of human health, not the protection of wildlife.

- **Screening level concentration approach (SLCA).** This approach uses matching biological and chemistry data collected in field surveys to calculate an estimate of the highest concentration of a contaminant that can be tolerated by benthic species.

- **Sediment quality triad approach (SQTA)**. The SQTA is based on correspondences between sediment chemistry, sediment bioassays, and *in situ* biological effects. This method can be used to formulate site-specific sediment quality objectives, but SQGs are not developed that would apply on a regional or national basis.

- **Apparent effects threshold approach (AETA).** Relationships between measured concentrations of contaminants in sediments and observed biological effects are used to define the concentration above which significant biological effects are always observed.

- **Weight of evidence approach (WEA).** This approach relies on a compiled database of information generated by EqPA, SSBA, and co-occurrence approaches like the AETA, SLCA, and SQTA. Datasets are screened for applicability and from the compiled database; thresholds are derived based on percentile values for the concentrations associated with biological effects.

The approaches that have been selected by individual jurisdictions depend on the receptors considered, the degree of protection afforded, the geographic area to which the values are intended to apply, and the intended uses of the values.

A compilation of sediment quality benchmarks that apply to freshwater sediments, marine sediments, or both, is presented in Appendix L (MacDonald et al., 2000b). Criteria are presented by contaminant, in ascending order, along with information on the water type for which the criteria were developed, the approach used to develop the threshold, and the area to which the threshold applies. The table includes concentrations recommended to support and maintain general designated uses, concentrations developed to protect designated uses at a specified site, and concentrations established by enforceable environmental control laws (MacDonald et al., 2000b).

MacDonald et al. (2000a) assembled sediment quality guidelines for 28 chemical substances and classified them into two categories according to their original narrative intent: a threshold effect concentration (TEC) and a probable effect concentration (PEC) (MacDonald et al., 2000a). TECs are intended to identify contaminant concentrations below which harmful effects on sediment-dwelling organisms are not expected to occur. TECs include threshold effect levels, effect range low values, lowest effect levels, minimal effect thresholds, and sediment quality advisory levels (Table 3.3). The PECs are intended to identify contaminant concentrations above which harmful effects on sediment-dwelling organisms are expected to occur frequently. PECs include probable effect levels, effect range median values, severe effect levels, and toxic effect thresholds (Table 3.4). These published sediment quality guidelines were then used to develop two consensus-based sediment quality guidelines for each contaminant, a TEC and a PEC. MacDonald et al. (2000a) reported that the consensus PEC numbers for arsenic, chromium, copper, and zinc correctly predicted sediment toxicity in 76.9%, 91.7%, 91.8%, and 90%, respectively, of 150 samples for arsenic and 347 samples for other contaminants from freshwater systems in the United States.

Overall, the sediment quality guidelines presented in Appendix L and Tables 3.3 and 3.4 can serve as useful screening tools to evaluate potential effects of wood-treating chemicals and benthic communities. However, site-specific factors may dictate whether adverse effects are likely at specific project locations. For example, sediment grain size and organic content can influence toxicity fairly dramatically. Furthermore, we were unable to determine the extraction methods (total digestion versus weak acid) used in studies relevant to these guidelines, causing some additional uncertainty in the resulting guidelines.

3.3 Dietary Exposure

Relatively little data are available to evaluate potential toxicity of dietary exposures to fish. Hansen et al. (2004) observed reduced growth in rainbow trout exposed to invertebrate diets with > 170 mg/kg (dry wt) arsenic in sediments, a value considerably higher than the sediment effects thresholds presented above. Similarly, adverse effects from exposures to dietary arsenic have been observed by Cockell et al. (1991) at concentrations between 13 to 33 µg As/g diet; by Cockell and Bettger (1993) at 58 µg As/g diet; and by Pedlar et al. (2002) at concentrations as low as 1 µg As/g diet. Adverse effects of dietary exposures to metals have also been demonstrated for copper at relatively elevated concentrations (Lanno et al., 1985; Julshamn et al., 1988), and metals mixtures (e.g., Mount et al., 1994; Woodward et al., 1995; Farag et al., 1999). Overall, however, because bioaccumulation factors from sediments to benthic invertebrates typically are relatively low for metals (e.g., Farag et al., 1998), it appears that the sediment effects thresholds presented in Tables 3.3 and 3.4 would likely be protective of dietary toxicity because of the limited potential for substantial metals accumulation in invertebrates.

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Table 3.3. Sediment TECs for freshwater sediment

Name	Definition	Basis	Concentration (mg/kg dry wt) As	Cr	Cu	Zn	Reference
Lowest effect level	Level that can be tolerated by the majority of benthic organisms	Field data on benthic communities	6	26	16	120	Persaud et al., 1991
Threshold effect level	Concentrations that are rarely associated with adverse biological effects	Compiled results of modeling, laboratory, and field studies on aquatic invertebrates and fish	5.9	37.3	35.7	123	Smith et al., 1996
Minimal effect threshold	Concentration at which minimal effects are observed on benthic organisms	Field data on benthic communities	7	55	28	150	Environment Canada, 1992
Effects range low[a]	Concentration below which adverse effects would be rarely observed	Field data on benthic communities and spiked laboratory toxicity test data	33	80	70	120	Long and Morgan, 1991
Threshold effect level	Concentration below which adverse effects on survival or growth are expected to occur only rarely	Laboratory toxicity tests on the amphipod *Hyalella azteca* using field-collected sediment	11	36	28	98	Ingersoll et al., 1996; U.S. EPA, 1996
Consensus threshold effect concentration	Concentration below which adverse effects are expected to occur only rarely	Geometric mean of above published effect concentrations	9.79	43.4	31.6	121	MacDonald et al., 2000a

a. Based on data from both freshwater and marine sites.

Table 3.4. Sediment PECs for freshwater sediment

Name	Definition	Basis	Concentration (mg/kg dry wt)				Reference
			As	Cr	Cu	Zn	
Severe effects level	Level at which pronounced disturbance of the sediment-dwelling community can be expected	Field data on benthic communities	33	110	110	820	Persaud et al., 1991
Probable effect level	Concentrations that are frequently associated with adverse effects	Compiled results of modeling, laboratory, and field studies on aquatic invertebrates and fish	17	90	197	315	Smith et al., 1996
Toxic effect threshold	Critical concentration above which major damage is done to benthic organisms	Field data on benthic communities	17	100	86	540	Environment Canada, 1992
Effects range median[a]	Concentration above which effects were frequently or always observed or predicted among most species	Field data on benthic communities and spiked laboratory toxicity test data	85	145	390	270	Long and Morgan, 1991
Probable effect level	Concentration above which adverse effects on survival or growth are expected to occur frequently	Laboratory toxicity tests on the amphipod *Hyalella azteca* using field-collected sediment	48	120	100	540	Ingersoll et al., 1996; U.S. EPA, 1996
Consensus probable effect concentration	Concentration above which harmful effects on sediment-dwelling organisms were expected to occur frequently	Geometric mean of above published effect concentrations	33.0	111	149	459	MacDonald et al., 2000a

a. Based on data from both freshwater and marine sites.

4. Risk Evaluation

This chapter discusses potential impacts to aquatic biota from the use of treated wood. Two alternative lines of evidence are available to evaluate potential impacts: (1) predictive ecological risk assessments of treated wood products, and (2) empirical laboratory and field studies designed to evaluate potential biological and/or ecological effects.

4.1 Predictive Risk Assessments

Several predictive, or modeled, assessments of ecological risks from treated wood have been performed. In general, these model-based evaluations predict little to no ecological risks from the use of most water-based treated wood products (in particular CCA, ACZA, and ACQ), including to sensitive salmonids.

Brooks (1995) developed a spreadsheet model to evaluate potential effects of copper [and polycyclic aromatic hydrocarbon (PAH)] released from CCA- and ACZA-treated wood on T&E species in the Columbia River basin. The author compared predicted copper concentrations against Washington State water quality criteria (6.54 µg Cu/L at an assumed water hardness of 59.5 mg/L), but recommended a lower criterion (5 µg Cu/L) for construction projects built within one week of active salmonid migration (this latter value is assumed in the analysis to be protective of salmonid migration). The author concluded that releases of treated wood are unlikely to cause adverse effects unless water is very poorly circulated (e.g., current velocities < 1 cm/sec). Brooks (1996) also developed an evaluation of risks to aquatic biota from CCA. In this evaluation, the author compares predicted water column and sediment concentrations of copper leached from treated wood. Brooks (1996) concludes that predicted environmental concentrations of copper are likely to be below ALC. However, the model suggests the potential to exceed ALC shortly after installation in poorly flushed environments or when the surface area of the treated wood comprises a significant proportion of the surface area of a water body. Brooks (1997a, 1997b) provides updated risk assessments for CCA and ACZA, respectively; the same conclusions are reached as in the earlier reports. Similar conclusions were also reached by the same author in a risk assessment performed for ACQ (Brooks, 1998) and in a report addressing potential effects of dissolved copper on salmon (Brooks, 2004b).

Sinnott (2000) developed a simulation model to evaluate ecological risks from CCA-treated wood. Using leaching rates from Lebow et al. (1999) and comparisons to New York State water quality standards (similar to the ALC), the author estimated “worst case potential for impacts to aquatic life.” The author concluded that although copper and arsenic are likely to be leached from treated wood, resulting concentrations are not likely to result "in a significant environmental impact."

The predictive risk models of Brooks (1995, 1996, 1997a, 1997b, 1998, 2004b) and of Sinnott (2000) rely on the empirical metal leaching models described in Chapter 2. As discussed in that chapter, the available empirical leaching models appear to capture the available laboratory data on metal leaching reasonably well. However, for a variety of reasons discussed in Chapter 2, there is uncertainty in applying the results of laboratory study-based leaching models to field conditions. Depending on the specific field application, the laboratory-based leaching models appear to be more likely to underpredict leaching under field conditions than overpredict them, at least for the initial leaching period that occurs within the first days and weeks after construction. Furthermore, there is much uncertainty in modeling actual environmental concentrations from the leaching study models, as also described in Chapter 2. Therefore, the results of the predictive risk assessment models should be interpreted carefully, as they may have substantial (and unquantified) uncertainty.

4.2 Laboratory and Field Studies

A number of laboratory and observational field studies also have been performed to evaluate potential impacts of treated wood products on aquatic biota. These studies allow for a direct assessment of the potential adverse effects and ecological risk associated with the use of treated wood in aquatic habitats.

Weis and Weis (1992b) and Weis et al. (1991, 1992) found that CCA leachates caused toxicity to estuarine organisms in laboratory static exposure tests; toxicity was associated with copper in the leachate. In later field studies with oysters, Weis et al. (1993a) found that oysters growing on CCA-treated wood piles had higher metals concentrations in soft tissues and a greater incidence of histopathological lesions than oysters collected from nearby rocks.

Weis et al. (1993b) collected sediments at different distances from CCA-treated wooden bulkheads in estuarine environments in New Jersey and New York. Bulkheads of different ages (several months to several years) were included in the study. The authors found that copper concentrations were higher than concentrations of chromium or arsenic and, at all sites, there was a decrease in the concentration of metals associated with fine-grained sediment as the distance from the bulkhead increased. (However, concentrations of metals in total sediment increased with distance, as did the percentage of fine grained sediments, suggesting the possibility of enhanced scouring associated with the bulkhead.) Immediately adjacent to a bulkhead in a New York marina, the maximum copper concentration in fine sediments was approximately 215 μg/g. At another New York marina, the copper concentrations in fine sediments at 0 m and 1 m from the bulkhead were between 500 and 600 μg/g. These maximum values fall into the range of sediment effects thresholds presented in Chapter 3.

In another study, Weis and Weis (1996) observed mortality in snails exposed to CCA leachates and in snails fed algae grown on CCA-treated docks. Weis and Weis (1992a, 1994) also measured significantly lower biomass and diversity of sessile epifaunal communities on treated wood panels than on untreated wood panels; the response appeared to dissipate over time and was negligible after three months of exposure.

Weis et al. (1998) measured metal concentrations in sediments and marine polychaete worms, and benthic invertebrate community diversity, abundance, and biomass near five CCA-treated wood bulkheads ranging in age from 1 to 8 years, reference bulkheads of concrete and aluminum, and open shoreline. Concentrations of copper and arsenic in sediments were generally elevated within 1 m from the treated wood bulkheads, but diminished to background levels by 3 m. Polychaete worms collected within 1 m of the newest treated wood structure contained elevated copper and arsenic concentrations Benthic community effects on abundance and diversity were noted at all treated wood sites, diminishing with distance from the structure. Effects were negligible by > 1 m from the structures.

Overall, the field and laboratory studies performed by Weis and Weis (1994) present a relatively consistent pattern. Releases of metals from CCA-treated wood can result in both metals accumulation and effects on resident invertebrates in the vicinity of treated structures. However, these effects typically are spatially limited, only observable within several meters of the structures, and appear to diminish with time.

In another field study, Wendt et al. (1996) evaluated the effects of treated wood structures on tidal creeks of the Charleston Harbor Estuary, South Carolina Study endpoints:

- Concentrations of Cu Cr, As, and PAHs in sediments and oysters in natural tidal creeks with and without docks
- Physiological condition and shell thickness of oysters in natural tidal creeks with and without docks
- Survival, growth, and bioaccumulation of metals and PAH in laboratory-reared oysters placed near to and far from newly constructed docks for six weeks
- Survival of estuarine fishes and invertebrates [white shrimp (*Penaeus setifeierus*), red drum (*Sciaenops ocellatus*), mummichog (*Fundulus heteroclitus*), and mud snail (*Ilyanassa obsolete*)] placed near to and far from newly constructed docks for four days
- Microtox and rotifer bioassays with sediments and pore water.

Copper, chromium, and arsenic were measured in composite samples of sediments from creeks with high densities of docks and from nearby reference creeks with no docks. The specific chemicals used to treat the wood were not identified. Average metal concentrations in sediments were higher in samples less than 1 m from the docks than in samples collected from locations greater than 1 m from the docks, but generally within the range of background concentrations (Table 4.1). The highest copper concentration (401 μg/g) was observed in a sample collected immediately adjacent to a dock piling.

Table 4.1. Mean concentrations (± 1 standard error) of copper, chromium, and arsenic in composite sediment samples

Site group	Copper (μg/g dw)	Chromium (μg/g dw)	Arsenic (μg/g dw)
< 1 m from docks	57.7 (± 34.4)	41.1 (± 3.2)	17.0 (± 1.4)
> 10 m from docks	19.1 (± 2.1)	32.4 (± 3.4)	13.9 (± 13.9)
Control	19.8 (± 2.7)	34.8 (± 2.6)	16.5 (± 16.5)

Source: Wendt et al., 1996.

Copper concentrations in field collected oysters were significantly greater near docks than > 10 m from docks and in reference areas. There were no dock-related patterns in tissue concentrations of chromium or arsenic. The physiological condition of field collected oysters did not differ significantly by site group. Mean shell thickness was slightly lower in oysters from dock pilings than in oysters from natural intertidal beds, but the difference was not significant. There were no significant correlations between condition index or shell thickness and tissue copper concentrations. Although survival and growth of oysters after six weeks of exposure was lower at dock sites than at reference sites, these differences were not statistically significant.

Microtox bioassays showed that elevated concentrations of copper and arsenic were associated with toxicity to the luminescent bacterium *P. phosphoreum*, but dock density was not significantly correlated with copper or arsenic concentrations in sediments. Rotifer bioassays showed no significant mortality from exposure to sediment pore water from any of the sites, regardless of dock proximity. Survival of and tissue concentrations of copper, chromium, and arsenic in mud snails, mummichogs, red drum, and white shrimp did not differ between sites near to and distant from newly constructed docks.

Wendt et al. (1996) concluded that wood preservative leachates were not a significant source of metal contamination in sediments in well-flushed tidal creeks, and that there was no evidence of adverse effects on resident biota.

In another study evaluating impacts of docks on tidal creek ecosystems in South Carolina, Sanger et al. (2004) reported that the presence of docks had little influence on sediment metals concentrations. Dock density was found to have small effects on the composition and abundance of benthic invertebrates. However, the authors of this study found that the biological effects were more likely related to habitat disturbance than from toxic effects of leachate.

A field study conducted by Cooper (1991) examined leaching of chromium, copper, and arsenic from CCA-treated lock gates. Water samples were collected downstream of both a newly installed gate and a gate that had been in service for five years. Concentrations of metals were not elevated in water around the five-year old gate. Copper levels in the water were elevated by approximately 200 parts per billion (ppb) adjacent to the new gate and 400 ppb at a location 40 m downstream, chromium levels were elevated by approximately 100 ppb at both locations, and arsenic levels were elevated by approximately 90 ppb near the gate and 60 ppb at the farther downstream location. These concentration increases would be expected to be associated with adverse effects on aquatic biota (see Chapter 3).

Brooks (2004a) collected sediment samples from locations adjacent to three piers in Washington State, which contained wood treated with ACZA [with retentions of 2.5 pounds per cubic foot (pcf)]. In 2002, samples were collected along transects at increasing distances from the piers and analyzed for concentrations of arsenic, copper, and zinc. Copper concentrations at locations near the piers ranged from 5.5 to 77.9 μg/g, while copper concentrations at four reference locations were between 5.8 and 11.2 μg/g. The highest copper concentration (77.9 μg/g) was from a sediment sample collected at a distance of 0.3 m from a pier. Other concentrations of copper were less than 20 μg/g. At one pier where samples were collected at a wide range of distances, the copper concentrations in sediment showed a clear decreasing trend with increasing distance from the pier. However, these concentrations are lower than most of the relevant sediment effects concentrations (see Chapter 3 and Appendix L). Concentrations of arsenic and zinc in sediments near the piers were similar to background concentrations.

Water column metals concentrations were also sampled during the Brooks (2004a) study. Triplicate water samples were collected at 0.5 m depth at two locations under a pier and at a reference station. Concentrations of arsenic, copper, and zinc in water under the piers were not significantly different than those in water from the reference station. The mean copper concentrations under the piers were 1.906 and 0.775 μg/L, compared to 1.690 μg/L at the reference station. The mean arsenic concentrations were 2.257 and 1.293 μg/L, compared to 1.230 μg/L at the reference station. The mean zinc concentrations were 7.233 and 2.113 μg/L, compared to 4.120 μg/L at the reference station.

Brooks (2000) sampled surface water and sediments adjacent to newly installed bridge (the Horseshoe Bayou Bridge) in a bay in Florida. Bridge pilings were constructed of southern yellow pine which had been treated with CCA-C to a retention of 2.5 pcf. The piling cross bracings and other parts were treated to 0.4 pcf. The salinity in this area was 25.5 ppt, the water temperature was 15.8°C, and the pH was 6.9. Samples of sediment and surface water were collected at a variety of distances from the bridge as it was nearing completion in 1998. Surface water samples had concentrations of copper, chromium, and arsenic similar to background concentrations (< 2 µg/L, < 2.1 µg/L, and < 7.5 µg/L, respectively), and did not show a relationship with distance. Concentrations of copper, chromium, and arsenic in sediments were higher within 10 feet of the bridge than at greater distances (Table 4.2). Sediment metal concentrations under and near the bridge were highly variable. Brooks (2000) concluded that these elevated concentrations may have been associated with metals in wood shavings spilled during construction of the bridge, rather than with loss from the preserved wood. The concentrations in Table 4.2, however, are lower than the sediment effects concentrations shown in Chapter 3.

Table 4.2. Metal concentrations in sediment with distance from the Horseshoe Bayou Bridge

Location	Copper (mg/kg)	Chromium (mg/kg)	Arsenic (mg/kg)
Control (175 ft from bridge)	2.43	4.13	0.80
Under bridge	11.87	23.57	9.80
1.5 ft downstream	7.95	17.20	2.65
3.0 ft downstream	4.25	10.80	17.9
6.0 ft downstream	3.80	6.93	3.80
10.0 ft downstream	1.55	2.30	0.85
20.0 ft downstream	2.15	3.20	0.70
33.0 ft downstream	1.80	3.15	0.55

Source: Brooks, 2000.

At a nearby bridge in a freshwater marsh (Fountains Bridge), Brooks (2000) conducted a similar study. This bridge was constructed with CCA-C-treated wood two years prior to the evaluation, and was located in an area with minimal water movement. Dissolved concentrations of copper and arsenic in the surface water were not significantly higher near the bridge than concentrations in reference samples. At locations up to 33 feet from the bridge, copper concentrations were equal to or below 2.62 µg/L, and arsenic concentrations were 7.48 µg/L or lower. Dissolved chromium concentrations increased with distance from the bridge, with the highest concentration (2.08 µg/L) observed at over 100 feet from the bridge, which was considered to be a reference location. Sediment concentrations of copper, chromium, and arsenic from locations between 10 and 20 feet from the bridge were slightly higher than concentrations at the reference station (Table 4.3). Again, these concentrations are lower than the sediment effects thresholds shown in Chapter 3.

Table 4.3. Metal concentrations in sediment with distance from the Fountains Bridge

Location	Copper (mg/kg)	Chromium (mg/kg)	Arsenic (mg/kg)
Under bridge	2.1	3.23	1.50
1.5 ft downstream	1.65	2.60	4.30
3.0 ft downstream	2.20	2.90	1.45
6.0 ft downstream	2.10	2.00	1.25
10.0 ft downstream	1.40	1.10	0.45
20.0 ft downstream	0.70	1.30	0.40
33.0 ft downstream	1.20	1.05	0.63
100.0 ft downstream	0.63	1.00	0.57

Source: Brooks, 2000.

Tarakanadha et al. (Date unknown) evaluated the effect of wood preservatives on settlement, abundance, growth, and biomass of non-target organisms in an Indian harbor. Algal and bryozoan settlement was common initially on all treated and untreated panels; these communities were replaced after a month by calcareous organisms (barnacles, oysters, and serpulids). Panels were examined for barnacles, serpulids, bryozoans, and oysters after 1, 2, 6, 12, 18, and 24 months. Total colony number, spread, growth by species, and biomass were recorded.

Biomass buildup by fouling organisms was greater on CCD-, CCA-, and CDDC-treated panels than on ACZA-, ACQ-, and ACC-treated panels. The authors concluded that CCA had no impact on epi-biotic communities. Biomass and growth and total number of individuals were greater on CCA-, CCA-, and CDDC-treated panels than on control panels. ACZA-, ACQ-, and ACC-treated panels had lower numbers, growth, and biomass.

Forest Products Laboratory (2000) reported on a study of contaminant leaching and environmental effects of a wetland boardwalk constructed in Oregon. Sections of the boardwalk were treated with four different preservatives (CCA-C, ACZA, ACQ-B, and CDDC); a control boardwalk of untreated Douglas fir was also constructed. The preservatives were applied to different wood species to address treated wood applications on the Pacific Coast. CCA-C and ACQ-B were applied to Western hemlock, ACZA to Douglas fir, and CDDC to Southern pine. Preservative retentions for the wood were as follows: ACQ-B, 7.04 kg/m^3 and 8.16 kg/m^3; ACZA, 7.04 kg/m^3; CCA-C, 11.68 kg/m^3; and CDDC 7.20 kg/m^3. Samples of riparian soils, wetland sediments, and water were collected at a series of points at increasing distances from the boardwalks. Samples were collected prior to construction, and at 2 weeks, 2 months, 5.5 months, and 11 months.

Copper concentrations in sediment sampled from locations around the boardwalk treated with CCA-C are presented in Table 4.4. Samples were separated into an upper zone (0-2.5 cm) and a lower zone (2.5-10 cm). In general, copper concentrations in both the upper zone and the lower zone increased with time. After 11 months, copper concentrations appeared to stabilize at locations near the boardwalk, but continued to increase at locations farther away (30 and 60 cm), suggesting that releases from the boardwalk had slowed, but that copper was being redistributed through the sediment. It should be noted that surficial samples in this study were collected as the top 2.5 cm, and therefore evaluation of the accumulation of copper in the surficial sediments at a scale finer than the top 2.5 cm is not possible. Nevertheless, the results of the study document that increased copper accumulation in sediment was observed downstream of the boardwalk.

Table 4.4. Copper concentrations in sediment samples near a boardwalk treated with CCA-C

Time (months)	Distance from boardwalk (cm)[a]	Sample size	Range (ppm)	Geometric mean (ppm)[b]
Upper 2.5 cm of sediment				
Preconstruction	Intended path	9	19-24	22
0.5	Under	6	28-49	**34**
	0	6	25-48	33
	30	6	18-43	27
	60	6	21-48	31
	150	6	14-34	22
	300	2	27-28	28
	Control	4	19-29	23
2	Under	6	45-201	**75**
	0	6	36-55	**43**
	30	6	24-54	35
	60	6	24-48	31
	150	6	17-32	24
	300	2	22-35	28
	Control	4	21-26	24
5.5	Under	6	35-219	**98**
	0	6	21-138	**58**
	30	6	20-64	**40**
	60	6	21-59	30
	150	6	21-28	23
	300	2	22-27	25
	Control	4	15-23	20

Table 4.4. Copper concentrations in sediment samples near a boardwalk treated with CCA-C (cont.)

Time (months)	Distance from boardwalk (cm)[a]	Sample size	Range (ppm)	Geometric mean (ppm)[b]
11	Under	6	34-115	**73**
	0	6	39-95	**63**
	30	6	32-83	**57**
	60	6	26-61	**44**
	150	6	19-51	33
	300	6	23-28	30
	Control	10	18-60	31
Lower 2.5-10 cm of sediment				
Preconstruction	Intended path	9	17-21	19
0.5	Under	6	21-29	24
	0	6	17-55	27
	30	6	18-35	23
	60	6	17-34	24
	150	6	16-24	20
	300	2	14-17	16
	Control	4	20-25	22
2	Under	5	22-121	**73**
	0	6	20-79	**36**
	30	6	22-53	30
	60	6	21-24	22
	150	6	17-22	19
	300	2	20-23	21
	Control	4	14-22	19
5.5	Under	6	20-59	31
	0	6	20-29	23
	30	5	17-36	22
	60	6	15-26	19
	150	6	13-23	18
	300	2	19-24	21
	Control	3	13-21	18

Table 4.4. Copper concentrations in sediment samples near a boardwalk treated with CCA-C (cont.)

Time (months)	Distance from boardwalk (cm)[a]	Sample size	Range (ppm)	Geometric mean (ppm)[b]
11	Under	6	23-46	**31**
	0	6	20-45	28
	30	6	21-83	30
	60	6	20-34	25
	150	6	14-22	18
	300	6	21-26	22
	Control	10	16-37	24

a. 0 = edge of boardwalk.
b. Values in bold are elevated above background and control levels at a 95% tolerance level.
Source: Forest Products Laboratory, 2000.

Concentrations of copper in the water column under the boardwalk treated with CCA-C and at distances up to 10 m were elevated compared to the concentration in water collected prior to the construction of the boardwalk. The maximum concentration (1.55 μg/L) was observed after 162 days at a distance of 1 m from the boardwalk. No significant changes in invertebrate communities were reported beneath or adjacent to any of the treatment locations.

In a followup publication (Lebow et al., 2002), sediment metals and invertebrate community measurements were taken for 24 months following boardwalk construction. For CCA-treated sections, copper concentrations in sediment generally increased throughout the 24-month study period at all distances from the boardwalk. At 24 months, sediment-Cu ranged from approximately 80 ppm beneath the boardwalk to approximately 40 ppm 3 m from the boardwalk. A similar pattern of increasing Cu was found at ACZA- and ACQ-treated sections, with maximum Cu concentrations exceeding 100 ppm beneath the ACZA-treated section and exceeding 120 ppm for the ACQ-treated sections. Despite the copper accumulation observed in the wetland sediments, the authors concluded that there were no measurable impacts to benthic invertebrate communities.

Poston (2001) reported the results of a "New York Mussel Study" (Adler-Ivanbrook and Breslin, 1999; as cited in Poston, 2001). The study was aimed at evaluating responses of blue mussels (*Mytilus edulis*) to CCA-treated wood in laboratory and field assays. At the dilution rates used in the study, no adverse effects on mussels were observed.

Overall, the field studies described above indicate that treated wood structures can leach metals, particularly copper, into the environment. However, the degree of metal accumulation associated with these structures appears to be relatively minor in most settings, particularly in well-circulated waters. Metal accumulation also appears to be relatively limited spatially (within 10 m of the structure) and has not generally been associated with significant biological effects except in close proximity to the structures. The duration of any biological effects appears to become attenuated within several months of construction (the time period when leaching rates are likely to be highest).

Notwithstanding the above, there are several factors that suggest that a precautionary principle might be applied to certain treated wood uses. First, the above studies typically have evaluated responses at the community level (e.g., benthic invertebrate studies) or to tolerant life stages (e.g., adult oysters and mussels). However, the level of environmental protectiveness applied to T&E species (such as endangered salmonids) should occur at the *individual* rather than the *population* or *community* level. Moreover, field studies have indicated that metals can accumulate to potentially deleterious concentrations in poorly circulated water bodies or when the density of treated wood structures is high compared to the overall surface area of the water body. As a result, we recommend that site-specific evaluations of risk be conducted for treated wood projects that are proposed for areas containing sensitive life stages of special concern species and where water circulation and dilution is potentially low. We discuss considerations associated with such site-specific risk assessments below.

4.3 Factors to be Considered in Aquatic Risk Assessments

Although the risks of treated wood to aquatic biota appear, based on the above studies, to be relatively low (and both temporally and spatially limited in extent), in certain settings site-specific risk assessments likely should be performed to ensure protectiveness of projects. Conditions that should prompt consideration of a site-specific risk assessment include:

- Low current velocities (e.g., current speeds < 1 cm/sec) and/or relatively little expected mixing coupled with a relatively high density of construction materials
- The presence of sensitive life stages (typically larvae and juveniles) of aquatic organisms, particularly T&E or special status species, in the project location.

When conducting such site-specific risk assessments, Hutton and Samis (2000) identify the following factors that should be considered:

- **Background water quality variables, including temperature, hardness, pH, salinity**

  As noted previously, the toxicity of copper (and zinc) is dependent on water hardness (and other related water quality variables). Although the absolute concentration of copper leached from treated wood may not be great (see Chapter 2 and field studies discussed above), low levels of copper can cause toxicity in soft, calcium-poor waters. Moreover, hardness can vary at locations over the annual hydrograph. As a result, site-specific risk assessments should consider the projected ambient hardness (and calcium) concentrations relative to the timing of releases. Worst-case scenarios (i.e., lowest annual hardness) may be used to ensure adequate protectiveness.

  The salinity of the receiving environment should also be considered. Leaching rates of some chemicals from treated wood are salinity dependent (Chapter 2), with most metals leaching at increased rates compared to freshwater. Although the toxicity of copper, chromium VI, and zinc to salmonids generally is reduced in saltwater, the toxicity of arsenic appears to be greater in saltwater (Chapter 3). In estuarine environments with fluctuating salinities, a worst-case scenario could be used to ensure protectiveness of special status species.

- **Current velocity and direction**

  Although total leach rates from treated wood can be relatively low, potential environmental effects will be dictated by local water mixing, with poorly-mixed waters being at greater risk. Information on current velocities – at the specific micro-environment – of the project location (including the influence of the structure itself on ambient current velocities) should be developed and integrated into a site-specific risk evaluation.

- **Proximity to sensitive fish habitat**

  The presence of sensitive life stages, especially T&E species or their essential prey species, should prompt an evaluation of potential risks at that location. Essential fish habitats for Pacific salmon include all streams, lakes, and other water bodies currently or historically accessible to salmon. This includes essentially all estuarine and marine waters of the Pacific Coast. The most sensitive life stages for these species are fry (particularly post swim-up) and juveniles. Because the initial leach rates are higher for treated wood, risk assessments should consider the timing of metals releases relative to periods when sensitive life stages of fish are not present.

- **Timing of proposed construction**

  Because initial leach rates tend to be greater, the timing of the proposed construction should be considered with respect to the presence of sensitive life stages of aquatic receptors; water flow rates; environmental/climatic factors that can influence mixing and dilution; and the relationship between season, annual hydrograph, and water quality conditions.

- **Size of proposed structure**

  As discussed previously, environmental effects are likely to be greatest when the size of the proposed structure is large relative to the receiving environment. Factors to consider include the number and size of pilings; the surface area of exposed wood area relative to a mixing zone; the density of pilings relative to the mixing zone (to evaluate potential behavioral avoidance responses), and potential effects of structure size on current flows.

- **Treatment chemicals and application methods**

  As discussed in greater detail in Chapter 7, treatment chemicals should be identified and treatment/application methods must be confirmed to meet industry BMPs.

- **Proximity of other preserved-wood structures and other sources of contamination which may contribute to cumulative effects**

  In evaluations of site-specific risks, assessments should consider potential effects in light of the cumulative effect of the proposed structure relative to other existing environmental perturbations at the site.

5. Alternative Materials

Materials other than treated wood are increasingly used in aquatic environment construction projects. Alternative materials are often favored because of their expected longevity and increased strength, as well as their minimal leaching characteristics. A comprehensive evaluation of these alternative construction materials is beyond the scope of this document. However, to provide perspective regarding potential costs and benefits of treated wood relative to other materials, this chapter describes materials that can be used as alternatives to wood in marine and freshwater construction, provides a brief summary of information on the potential toxicity of the chemical compounds in these materials, and discusses economic cost considerations.

5.1 Material Types

5.1.1 Galvanized steel

Steel used for construction in aquatic environments is usually galvanized to retard corrosion. Galvanizing forms a metallurgical bond between the protective material and the underlying steel, creating a barrier incorporated into the metal itself (AGA, 2000). Typically, sacrificial cathodic protection, impressed-current cathodic protection, or coating materials are used (Hutton and Samis, 2000). Cathodic protection uses zinc, aluminum, or magnesium anodes, which are corroded in preference to steel, and can be released into water (AGA, 2000).

Many types of coating materials are available to reduce steel corrosion. Common coating materials include thick galvanic zinc, high-zinc acrylic, epoxy polyamide, and coal tar epoxies. New technologies include fusion bonded epoxy coatings, moisture cured urethane coatings, coatings of epoxy aliphatic polyurethane with polypropylene fibers, polyurea coatings similar to utility truck bed linings and industrial floor coatings, and glass flake resin coatings. U.S. Army Corps of Engineers (USACE) evaluations have demonstrated that the performance of coatings varies by type, combination, and formulation (USACE, 1978).

5.1.2 Concrete

Concrete structural pilings and bulkheads are used in many commercial and industrial applications. Concrete can be manufactured as one or more of the following types: pre-stressed or partially pre-stressed, pre-cast, cast-in-place, and reinforced, usually with steel reinforcing bars (rebar) embedded in the concrete (USACE, 1978). Concrete spalling (fragmentation around corroding rebar) may occur in some marine environments in as little as five years, although concrete formulations and rebar placement can be designed to resist spalling for 50 to 100 years (Sagues et al., 1994).

5.1.3 Recycled plastic lumber

Recycled plastic lumber (RPL) is a wood-like product made from recovered plastic or recovered plastic mixed with other materials (CIWMB, 1997b). Recycled plastic feedstocks are usually high density polyethylene (HDPE) from items such as discarded milk and bulk water jugs. Plastic lumber is moisture and chemical resistant, does not need sealants or preservatives, does not crack, and is flexible enough to be curved or shaped (CIWMB, 1997a).

The California Recycling Company (CRC) conducted an evaluation of recycled plastic materials, including tests to determine strength, creep (gradual bending or sagging under weight), serviceability, biological compatibility, and toxicity of the plastic lumber (R.W. Beck and Associates, 1993; as cited by Breslin et al., 1998). These tests were selected to characterize the behavior of the material for marine application. Results indicated that plastic lumber has significant creep characteristics. This can be a major obstacle to the use of plastic lumber in marine construction, particularly for load bearing applications (Breslin et al., 1998). To reduce creep and increase strength and flexure, recycled HDPE is blended with glass fibers for additional strength and rigidity. Without reinforcement, polyethylene-based plastic lumber is at least an order of magnitude less flexible than natural wood of the same dimensions (Krishnaswamy and Lampo, 2001).

5.1.4 Combination materials

Many of the new alternative construction materials are products made from a combination of source materials. Examples include pilings made from wood plastic composites (WPCs), fiber reinforced polymer (FRP) tubes filled with concrete, and wood or steel pilings wrapped with ultraviolet (UV) radiation and abrasion resistant plastic coatings.

WPCs consist of 50% wood fiber and 50% thermoplastic polymer plastic material (EPIC, 2003). WPCs are as durable as wood in terms of resistance to weathering, moisture, marine borers, and other decay factors. WPCs also require little maintenance and show high resistance to moisture and marine borers. WPCs are also relatively durable. Toxic chemicals are not required for resistance to decay. The U.S. Navy has been conducting research on the use of WPCs in naval wharves and piers (Sorathia et al., 2002).

FRPs are composed of structural fibers in a plastic matrix, and generally consist of a hollow FRP tube, a concrete core, and a durable, environmentally neutral coating (USACE, 1997). The components of FRPs tend to be inert materials that do not leach into the environment. FRPs also have many desirable structural properties. The tensile strength of FRPs can range from the strength of mild reinforcing steel to that of prestressing steels (USACE, 1997). Because of their high strength and low density, FRP composites have specific strengths that are up to 60 times greater than high strength steels (USACE, 1997). FRPs also show good fatigue resistance.

Steel, concrete, and wood pilings can be wrapped with UV and abrasion resistant plastic sheeting of polyvinyl chloride (PVC) or spray-on coatings of materials such as polyurea to produce a piling designed to have lower degradation or leaching than traditional pilings (USACE, 1978).

5.2 Toxicity Considerations

As with treated wood products, chemical leaching may occur from alternative construction materials, and none should be considered to be entirely non-toxic.

5.2.1 Galvanized steel

Zinc corrosion from galvanized steel and cathodic protection of steel can pose a significant source of zinc releases in aquatic environments. The USACE has documented that zinc can corrode from galvanized steel in freshwater at a rate sufficient to be an effective biocide against zebra mussels in the Great Lakes (Race and Kelly, 1994). Leach rates immediately after installation were 15 $\mu g/cm^2/day$, dropping to about 5 $\mu g/cm^2/day$ for the first two years after installation. The corrosion rate for zinc from galvanized steel may increase in saltwater splash zones, although the toxicity of zinc is considerably reduced in seawater. Magnesium and calcium ions in seawater have a strong inhibiting effect on zinc corrosion (AGA, 2000). However, zinc corrosion and toxicity could be a factor in estuarine waters impaired for zinc, such as portions of San Francisco Bay and estuaries in Southern California. The cumulative effect of multiple steel projects should be considered in evaluating projects.

5.2.2 Concrete

Pouring concrete in place requires measures to prevent highly alkaline water from entering and damaging the surrounding environment. Water surrounding piles immediately after curing should be tested for pH and neutralized with acid prior to being released back into the receiving water.

Recent advances in concrete technology include the addition of coal ash to concrete mixes. Coal fly ash, a by-product of the burning of coal in electric utilities, may be incorporated into cement at levels as high as 20-30%, reducing the amount of cement used (Zhang et al., 2001). Coal fly ash can contain heavy metals such as arsenic, cadmium, copper, antimony, selenium, molybdenum, lead, and zinc. As a result, concerns have been raised that leaching of these elements from the fly ash could impact aquatic biota.

5.2.3 Plastic lumber and recycled plastic

Although unpolymerized plastic monomers, plasticizers, and other adjuvants to plastic formulations are well-documented endocrine disruptors, few evaluations of the leaching risks of plastics have been published, especially for structures made from recycled materials.

Weis et al. (1992) compared the toxicity of leachate from CCA-treated wood to the toxicity of leachate from recycled plastic lumber fabricated in the Rutgers University Center for Plastic Recycling. In the laboratory, CCA-treated wood leachates were found to be more acutely toxic, but the plastic leachates caused accelerated rates of leg regeneration of fiddler crabs. The plastic leachates also caused reduced fertilization of sea urchin eggs. Chemical identification of the plastic leachate revealed at least 14 phthalate isomers, plus nonyl-phenol, although only qualitative estimates of each leached component were made. The physiological or hormonal mechanisms of the stimulation of crab regeneration and sea urchin egg infertility were not evaluated, and the chemical agents responsible were not identified.

Xie et al. (1997) evaluated leachates from CCA-treated wood and recycled plastic used for construction of a pier in the East River, New York City. Laboratory studies were conducted to evaluate effects of organic chemicals and metals leached from both products. Phthalates were leached from both treated wood and plastic, although at higher concentrations from the plastic. The presence of phthalates in treated wood was unexpected, and the source was not identified. Phthalate contamination of solvents, analytical glassware, and instrumentation in the laboratory could not be ruled out. Copper was also leached from the recycled plastic at low levels, which indicates that recycled plastic may contain many unexpected substances, including metals as well as endocrine disruptors.

Xie et al. (1997) also identified many phthalates in samples of water taken from the East River, and concluded that the plastic of the pier was contributing only a small additional load of phthalates. Neither Weis et al. (1992) nor Xie et al. (1997) addressed the risks of phthalates and other potential endocrine disruptors.

5.2.4 Plastic coatings

The recent technological advancements in polyurea coatings have resulted in a greatly expanded use of these materials in spray coatings for many applications, including pilings for aquatic construction use (Broekaert, 2003). Polyurea coatings are solvent and catalyst free systems produced by mixing two reactive components at the time of spray application, resulting in a 100% solids coating with no leachable chemicals. The reactive components may be toxic, and the mixing ratio is critical in producing a complete reaction with no residual reactants (Broekaert, 2003).

5.3 Economic Considerations

This section presents a discussion of economic considerations related to the use of treated wood and alternative materials for fender pilings and pier supports.[1] Fenders are cushioning devices on the side of a pier or wharf that dissipate the impact of docking boats, wind, waves, and current. Based on the availability of cost information, the alternative materials considered in this analysis include galvanized steel, concrete, and plastic pipes (reinforced plastic, plastic coated steel, and fiber reinforced plastic).

First, we summarize material and installation costs for a hypothetical 80-foot piling constructed with each alternative piling material. Costs are based on material costs, spacing requirements of the piles, and the expected service life of the construction material.

This summary of material and installation costs for alternative construction materials is followed by the calculation of annualized costs for each alternative for a hypothetical pile fender. Omitted from this annualized cost analysis are any removal and disposal costs. These costs are likely to vary widely based on local disposal opportunities and restrictions. Maintenance costs are also excluded based on the assumption that minimal effort is devoted to fender pier maintenance on an annual basis.

Finally, the relative environmental and disposal costs of each alternative are discussed qualitatively.

1. The analysis does not consider pier costs because of a lack of comparable estimates for materials other than treated wood. The limited information available from KPFF (2004) suggests that the installation cost of concrete is not disproportionate to the installation cost of treated wood, and that concrete has lower maintenance costs. However, concrete may not be a suitable alternative for some pier structures.

5.4 Summary of Costs for Alternative Piling Materials

5.4.1 Treated timber

A treated 80-foot timber pile typically costs from $750 to $1,000, and these piles typically are spaced at 10-foot intervals in a fender (KPFF, 2004). Although environmental and use conditions will affect the service life of a treated timber fender piling, 15 years represents a probable maximum project life (personal communication, K. Nikzad, KPFF Consultant Engineers, December 29, 2004).

Estimates of the typical installation costs for a treated timber pile range from approximately $500 (personal communication, D. Yingling, Sales Manager, American Piledriving Equipment, Inc., January 4, 2005) to $770 (Alling, 1996, p. 70). The midpoint of this range of installation costs, $635, is incorporated in our economic evaluation.

5.4.2 Galvanized steel

Galvanized steel and plastic coated galvanized steel are both options for fender pilings. An 80-foot galvanized steel fender piling typically costs about $2,400, and a similar plastic pile with a plastic coating costs about $4,400 (KPFF, 2004). Both are usually spaced at 20-foot intervals when used for fenders (KPFF, 2004) and are expected to last up to 20 years in this application (personal communication, K. Nikzad, KPFF Consulting Engineers, December 29, 2004).

Installation costs for both the plastic coated and uncoated galvanized steel piles are higher than for treated timber because they are relatively heavy and are non-buoyant. Because of these characteristics, galvanized steel piles can require specialized design, larger handling and pile driving equipment, and specialized tools for cutting and trimming after installation Therefore, the installation cost for a steel pile is approximately 25% more than for treated timber (personal communication, D. Yingling, Sales Manager, American Piledriving Equipment, Inc., January 4, 2005). Using the midpoint from the range of installation costs for a treated timber pile, this is equivalent to an installation cost of $794 for both the coated and uncoated galvanized steel pile.

5.4.3 Concrete

An 80-foot reinforced concrete pile to be used as a fender costs around $2,800, is typically spaced at 20-foot intervals (KPFF, 2004), and is expected to last up to 20 years (personal communication, K. Nikzad, KPFF Consulting Engineers, December 29, 2004).

For the same reasons as for the galvanized steel alternatives, installation costs for reinforced concrete piles are roughly 50% more than for treated timber (personal communication, D. Yingling, Sales Manager, American Piledriving Equipment, Inc., January 4, 2005). Using the midpoint from the range of installation costs for a treated timber pile, this is equivalent to an installation cost of $953 for a reinforced concrete pile.

5.4.4 Composite and plastic pipes

Composite and plastic pipes that can be used as fender pilings include structural pipe cores, concrete filled fiberglass shells, fiberglass and steel reinforced plastic piles, and combinations of these materials (Warren, 1996; as cited in Alling, 1996).

The cost for an 80-foot pile is roughly $4,000 for a steel-reinforced plastic pile and $3,200 for a fiber-reinforced plastic pile; both are typically placed at 10-foot intervals and are assumed to have an effective project life of 10 years (KPFF, 2004). Plastic pilings are usually installed using the same hardware, tools, equipment, and methods as timber piles, so the same installation cost estimate of $635 is incorporated for this evaluation.

5.5 Annualized Costs for a Hypothetical Fender Piling Project

Because the combination of anticipated years of effective service, purchase and installation costs, and installation spacing vary among construction materials, we evaluate the costs of the different alternatives for a hypothetical project to create a comparative cost analysis.

For our hypothetical project, we assume fender pilings need to be installed over 100 linear feet of waterway on a pier or shore structure. To account for the differences in the anticipated project life using different construction materials, we first assume the maximum project life is realized, and then consider the cost if the project lasted only half as long. All values are rounded to the next whole year.

Having defined the number of years the project will be in place before it needs to be replaced, the equivalent annual cost of the project can be calculated. The equivalent annual cost represents the amount of money that would need to be paid each year of the project life that provides a present value equal to the installation cost. The present value calculations assume a discount rate of 3% with no discounting in the first year (i.e., total payment is made on the first day of the first year of the project life). The results of this cost evaluation are presented in Table 5.1.

Table 5.1. Annualized cost estimates for alternative pile materials. Data sources are provided in the text.

Pile material	Cost per pile	Cost to place piles	Spacing between piles (feet)	Number of piles for project	Total cost to install project[a]	Maximum project life (years)	Equivalent annual cost over maximum project life
Treated timber	$875	$635	10	10	$15,100	15	$1,228
Galvanized steel	$2,400	$794	20	5	$15,969	20	$1,042
Plastic coated galvanized steel	$4,400	$794	20	5	$25,969	20	$1,695
Reinforced concrete	$2,800	$953	20	5	$18,763	20	$1,224
Steel-reinforced plastic	$4,000	$635	10	10	$46,350	15	$3,769
Fiber-reinforced plastic	$3,200	$635	10	10	$38,350	15	$3,119

a. Does not include costs for material removal and disposal at end of project life.

The results in Table 5.1 show the importance of accounting for project life and the required spacing of pilings. Treated timber piles are by far the least expensive in terms of unit costs for the purchase of the material and for installation, totaling roughly half the cost of the next least expensive alternative (galvanized steel). However, this cost advantage is lost once the longer expected life of other materials is accounted for, along with the need for fewer piles. As a result, Table 5.1 shows that there is likely to be little cost difference between the use of treated timber, galvanized steel, or reinforced concrete piles in terms of the equivalent annualized cost of a fender project. Estimated costs for both steel-reinforced and fiber-reinforced plastic piles are about three times more than for the other alternatives.

Accounting for disposal costs is likely to do little to change the overall or relative rankings of the construction materials in Table 5.1. Although disposal costs will reflect local opportunities and constraints, some general trends may exist. Disposal costs for treated timber are likely to be relatively high compared to the other materials because they have limited alternative uses and may have to be disposed of as hazardous waste in some locations. The next highest disposal cost may be that for reinforced concrete because of its weight and likely limited options for alternative uses and recycling. Disposal of the other construction materials may include high landfill fees, but lower overall costs because their components can be recycled or may have active scrap markets.

6. Current Regulations and Best Management Practices

This chapter summarizes existing regulations and BMPs for waterborne preservatives in treated wood. These regulations and BMPs were developed to try to minimize the potential for adverse impacts to surface water, sediments, and aquatic biota from the leaching of chemicals into the aquatic environment. Local, state, federal, international, and industry regulations and BMPs are reviewed. The objective of this chapter is to provide a synthesis that highlights key provisions and key differences of regulations and BMPs from different sources.

Relevant regulations and BMPs cover three main issues related to waterborne preservatives:

- How should treated wood be produced and monitored?
  - What practices should treatment plants follow?
  - What types of quality assurance/quality control should be in place?
  - How should independent verification take place?
- What types of construction practices are appropriate for treated wood?
  - How should wood be handled in the field?
  - How can environmental risks be minimized?
- When should treated wood be used?
  - What environmental characteristics (e.g., flow rates, pH) make treated wood appropriate or inappropriate for aquatic use?
  - Which types of wood treatments are appropriate for different uses?

The remainder of this chapter discusses the regulations and BMPs related to each of these three issues.

6.1 How Should Treated Wood be Produced and Monitored?

Potential environmental risks from treated wood are minimized when the wood is produced according to the highest technical standards. Using the minimum amount of chemical necessary to achieve the desired performance, achieving full fixation of the chemical into the wood cells before the time of installation, and minimizing surface residues are all key practices for reducing environmental risks when the treated wood is in an aquatic environment. In recent years, the wood treatment industry has worked to reduce the amount of preservatives used in treated lumber, and has developed procedures and BMPs to reduce the amount of excess surface residues during treatment.

Key BMPs addressing these technical issues are issued by the WWPI and the CITW (WWPI and CITW, 1996), and updated by the WWPI (2002a). These BMPs cover each of the waterborne preservatives (plus creosote), specifying chemical-specific uses and specifications, treatment procedures, and post-treatment procedures to achieve fixation. Some key features of these BMPs are highlighted in Table 6.1. In addition, the BMPs for all the chemicals state that treatment should "minimize the amount of chemical placed into the wood while assuring conformance with the AWPA retention and penetration requirements," and that the product should be visually inspected "to insure that no excessive residual materials or preservative deposits exist" (WWPI and CITW, 1996).

The BMPs issued by the WWPI and the CITW do not provide specific guidance on when it is or is not appropriate to use treated wood versus a different material (e.g., steel, concrete). The BMPs simply state "Projects calling for large volumes of treated wood immersed in . . . poorly circulating bodies of water should be evaluated on an individual basis using risk assessment procedures" (WWPI and CITW, 1996).

The BMPs described above reference the standards produced by the AWPA (e.g., AWPA, 2003). The AWPA issues standards that specify the chemicals and treatments to be used for waterborne preservatives according to the designated end use for the treated wood product. The retention levels of chemicals in treated wood are specified by the AWPA as the minimum that will protect wood from decay and borer damage. For example, the required amount of preservative retention is higher for wood directly in contact with soil or water compared to wood used in exterior construction that is not in direct contact with soil or water.

The current version of the AWPA standards uses a system of five "use categories" that define the conditions that treated wood will encounter. The required amount of preservative retention, and sometimes the depth of penetration, increases as the use category number rises and the potential for chemical leaching increases (i.e., Use Category 5 requires more chemical retention than Use Category 1) (Table 6.2).

Table 6.1. Selected key provisions of BMPs for treated wood according to the WWPI and the CITW

Preservative	Use/ specifications	Treatment procedures	Post-treatment procedures
Chromated copper arsenate	Full range of salt and freshwater applications Not recommended for Douglas fir for marine pilings because chemical does not penetrate well	Use treating solutions in accordance with relevant AWPA standards Minimize sawdust and surface residue	Required to achieve fixation – air seasoning, kiln drying, steaming, or hot water bath Fixation should be confirmed with a chromotropic acid test (the absence of CrVI indicates the reaction is complete)
Ammoniacal copper zinc arsenate	Full range of salt and freshwater applications Useful for Douglas fir	Use treating solutions in accordance with relevant AWPA standards Minimize sawdust and surface residue Apply a final vacuum process for a minimum of 2 hours	Required to achieve fixation – air seasoning for a minimum of 3 weeks, kiln drying, "in-retort ammonia removal" followed by air seasoning for a minimum of 1 week, or "aqua-ammonia steaming cycle"
Alkaline copper quat	Salt and freshwater applications but not recommended for saltwater immersion Useful for Douglas fir	Use treating solutions in accordance with relevant AWPA standards Minimize sawdust and surface residue	Required to achieve fixation – air seasoning, kiln drying

Sources: WWPI and CITW, 1996; as updated by WWPI, 2002a.

Table 6.2. Specification guide to treated wood end uses showing the amount of waterborne preservatives required for each use category relevant to aquatic systems

	AWPA standard		Waterborne preservatives (minimum retentions – pound per cubic foot)			
Use	**Use category system**	**C standard[a]**	**ACQ[b]**	**ACZA[c]**	**CA-B[d]**	**CCA[e,f]**
Decking						
Highway bridge	4B	C2, C14	0.60	0.60	NL	0.60
Highway material						
Lumber and timbers for bridges, structural members, decking, cribbing, and culverts	4B	C2, C14	0.60	0.60	NL	0.60
Structural lumber and timbers:						
– In saltwater use and subject to marine borer attack	5A, 5B, 5C	C3, C14	NL	2.50	NL	2.50
– Piles, foundation, land, and freshwater use	4C	C3, C14	NL	0.80-1.0	NL	0.80
– Piling in saltwater use and subject to marine borer attack	5A, 5B, 5C	C3, C14	NL	1.5-2.5[g]	NL	1.5-2.5[g]
– Posts: round, half-round, quarter-round	4A	C5, C14	0.40	0.40	NL	0.40
– Posts: sawn	4A	C2, C14	0.40	0.40	NL	0.40
– Handrails and guardrails	3B	C2, C14	0.25	0.25	NL	0.25
Lumber						
Ground contact and freshwater use	4A	C2	0.40	0.40	0.21	0.40
Marine lumber and timbers						
Freshwater	4A	C2	0.40	0.40	0.21	0.40
Members out of water but subject to saltwater splash or ground contact	4B	C2, C18	0.60	0.60	NL	0.60
In brackish or saltwater use and subject to marine borer attack	5A, 5B, 5C	C2, C18	NL	2.50	NR	2.5

Table 6.2. Specification guide to treated wood end uses showing the amount of waterborne preservatives required for each use category relevant to aquatic systems (cont.)

Use	AWPA standard		Waterborne preservatives (minimum retentions – pound per cubic foot)			
	Use category system	C standard[a]	ACQ[b]	ACZA[c]	CA-B[d]	CCA[e,f]
Piles						
Land and freshwater use (round)	4C	C3	0.80	0.80-1.0	NL	0.80
Marine (round) in salt or brackish and subject to marine borer attack	5A, 5B, 5C	C3, C18	NL	1.5-2.5[g]	NL	1.5-2.5[g]
Marine, dual treatment (round)	5A, 5B, 5C	C3, C18	NL	1.0	NL	1.0
Sawn timber piles	4B	C24	0.60	0.60-0.80	NL	0.60-0.80

a. The C standards are being phased out in favor of the use category system.
b. Alkaline copper quat.
c. Ammoniacal copper zinc arsenate.
d. Copper azole.
e. Chromated copper arsenate.
f. It is generally recognized that Douglas fir is extremely difficult to treat with CCA to required penetration and retention.
g. The lower retentions for creosote, ACZA, and CCA are for waters from New Jersey North on the East Coast and North of the San Francisco Bay on the West Coast.

Source: AWPA, 2003.

6.1.1 Quality assurance/quality control

Quality assurance and quality control for the wood treatment BMPs is important to ensure that the treated wood was produced in compliance with standards. The WWPI has produced a document entitled “Quality Assurance Inspection Procedures for Best Management Practices (BMP’s) for the Use of Treated Wood in Aquatic Environments” (WWPI, 2001). These inspection procedures are designed to be followed by an independent, third-party inspection agency that is accredited by the American Lumber Standard Committee. A “check mark” logo is assigned to structural materials that have been inspected in compliance with AWPA standards. In addition, for treated wood that is intended for aquatic or wetland applications, third-party inspection and certification is documented by the presence of the WWPI “BMP Certification Mark,” a logo showing three wooden pilings with a fish swimming nearby. This logo (or written certification) certifies that the wood products were produced in accordance with the industry BMPs for treated wood in aquatic environments.

6.1.2 Adoption of BMPs

To assure that treated wood has been produced in compliance with the BMPs for aquatic applications, WWPI recommends a four-step process for project regulators and designers:

- Specify the appropriate material in terms of performance defined in the book of standards from the AWPA
- Specify that the material be produced in compliance with the BMPs
- Require assurance that the products were produced in conformance with these BMPs (i.e., third-party inspection agency certification)
- Provide for on-site inspection before installation and conformance with recommended installation practices (WWPI and CITW, 1996).

The WWPI and CITW have promoted their BMPs for widespread adoption. For example, the State of Michigan produced a booklet entitled “Best Management Practices for the Use of Preservative-Treated Wood in Aquatic Environments in Michigan” (Michigan Timber Bridge Initiative, 2002) that is almost wholly adapted from the WWPI and CITW BMP guide. The USFWS Division of Engineering produced a guidance for wooden bridge design that includes furnishing a BMP compliance certificate to the Contracting Officer on delivery of treated wood to the job site (USFWS, 2001). The USFWS guidance also recommends the adoption of the four steps listed above to assure that treated wood products used in aquatic environments incorporate the industry BMPs.

6.2 Construction Specifications

Minimizing the effect of treated wood on aquatic environments requires implementing specific construction practices. The USDA Forest Service Forest Products Laboratory developed a "Guide for Minimizing the Effect of Preservative-Treated Wood on Sensitive Environments" that includes recommendations for construction practices (Lebow and Tippie, 2001). Key recommendations include:

- On-site storage of treated wood free from standing water or wet soil; ideally, the wood is supported off the ground and covered until used.
- Minimize the amount of field fabrication by careful prefabrication before treatment.
- Collect construction debris, including all wood sawdust and shavings, which can have a disproportionate impact on releases compared to solid wood (tarps can be spread under structures before cutting).
- Apply field treatment preservatives sparingly and with care to avoid spillage. Field treatment is necessary for wood exposed during field fabrication; usually copper naphthenate (CuN) is used. AWPA Standard M4 gives requirements for field treatments.
- Carefully apply water repellants and stains, which can increase longevity and reduce leaching from treated wood.

6.3 When Should Treated Wood be Used?

A number of guidelines and policy directives have been developed within the last 10 years to address the issue of when and what types of treated wood are appropriate for use in aquatic environments. These guidelines have been produced by different local, state, federal, and international agencies, as well as by industry. For the most part, these guidelines assume that the BMPs for treated wood production have been followed and that appropriate construction practices have been used (see Sections 6.1 and 6.2). Table 6.3 summarizes selected key guidelines and directives that focus on when to use treated wood; a discussion of these guidelines and directives follows the table.

Table 6.3. Summary of guidelines and directives focused on when to use treated wood in aquatic environments

Author, date, source agency (if different from author)	Title	Habitats	Elements in guidelines/directives
	Federal agencies		
USACE, 1996	Biological Assessment (BA) for the use of Treated Wood Products in the Columbia River	Columbia River	Uses number of piles, cross-sectional area, and current velocity to predict water column Cu concentrations in exceedence of thresholds
NMFS, 1998	Position Document for the Use of Treated Wood in Areas within Oregon Occupied by Endangered Species Act Proposed and Listed Anadromous Fish Species	Lower Columbia River and other Oregon waters	Uses number of piles, pH, cross-sectional area, and current velocity to predict water column and sediment Cu concentrations in exceedence of thresholds
U.S. Department of Defense, 2001	Replacement of up to Eighteen (18) Existing Piling, Version: May 30, 2001	Washington State	No treated wood for freshwater or for the Columbia River mainstem including Snake River and Baker Bay No piles treated with creosote or pentachlorophenol in marine/estuarine waters, excluding Baker Bay
NOAA Fisheries, 2003	Draft NOAA Fisheries Northwest Region Habitat Conservation Division Guidance for Projects that Propose the Use of Treated Wood	Northwest Region habitat relevant to ESA or MSA species	Preference for alternative materials Treated wood must adhere to WWPI BMPs if used Removal of treated wood debris could be used as mitigation
USACE, 2004	Department of the Army General Permit New Jersey SPGP-19: General permit for the construction and maintenance of piers, docks, mooring piles, boat lifts, timber breakwaters, and replacement bulkheads	Navigable waters in the State of New Jersey	Use of creosote and pressure-treated lumber (including CCA-C, ACZA, CC, ACQ, etc.) is not acceptable for any structures covered under the permit

Table 6.3. Summary of guidelines and directives focused on when to use treated wood in aquatic environments (cont.)

Author, date, source agency (if different from author)	Title	Habitats	Elements in guidelines/directives
State and local agencies			
Oregon State Marine Board, 2002	Best Management Practices for Environmental and Habitat Protection in Design and Construction of Recreational Boating Facilities	Boating facility projects in Oregon	Avoid treated wood for inwater structures for boat ramps Round steel pipe is preferred for piling Use of treated wood for piling should be avoided and existing wooden piles should be removed when possible
California Coastal Commission, 2003	Coastal Development Permit Application: Port San Luis Five-year Operations & Maintenance	Port San Luis Harbor in San Luis Obispo County, CA	Phase out ACZA treated pilings Require new fender pilings to be made of plastic
Dickey, 2003 (prepared for the San Francisco Department of the Environment)	Guidelines for Selecting Wood Preservatives	Waters in the City of San Francisco, including San Francisco Bay	Prohibits use of ACA, ACZA, and CCA except for saltwater immersion environments
International (non-U.S.) agencies			
Hutton and Samis, 2000 (Fisheries and Oceans Canada)	Guidelines to Protect Fish and Fish Habitat from Treated Wood Used in Aquatic Environments in the Pacific Region	Shoreline projects using treated wood	Alternatives to treated wood should be used wherever practicable Only wood treated to BMP specifications will be acceptable
Government of Newfoundland and Labrador, 2001	Policy for Treated Utility Poles in Water Supply Areas	Water supply areas	Avoid treated poles within buffer zone of water supply areas

Table 6.3. Summary of guidelines and directives focused on when to use treated wood in aquatic environments (cont.)

Author, date, source agency (if different from author)	Title	Habitats	Elements in guidelines/directives
	Other recommendations		
Poston, 2001	Treated Wood Issues Associated with Overwater Structures in Marine and Freshwater Environments	Marine and freshwater environments in the State of Washington	Use of treated wood for over- and in-water structures can result in the release of wood preservatives into aquatic habitats
WWPI, 2002b	Treated wood in aquatic environments	All aquatic environments	Project-specific risk assessment only if current > 1.0 cm/sec, pH < 5.5, or more than 100 piling for CCA-C, 25 for ACZA, or two piling for ACQ-B.
Michigan Timber Wood Initiative, 2002	Best Management Practices for the Use of Preservative-Treated Wood in Aquatic Environments in Michigan	Aquatic habitats in Michigan	Conform to AWPA BMPs when using treated wood in aquatic habitats

U.S. Army Corps of Engineers, 1996

The USACE (1996) issued a BA for the use of treated wood products in waters of the Columbia River basin that affect listed salmonids and are under the jurisdiction of the Portland District of the Corps (Corps). The BA was limited to creosote, ACZA, and CCA that have been treated according to the WWPI BMPs. The purpose of the BA was to address treated wood in areas with a current faster than 0.5 cm/sec, to determine the amount of treated wood that would not adversely affect listed salmonids.

Model analyses used by the Corps indicated that impacts associated with ACZA-treated wood were higher than impacts associated with CCA-treated wood, so ACZA was used as a conservative analysis. The document states that the water quality criterion for copper was based on the U.S. EPA ALC for Cu of 7 mg/L (ppm) at 55 mg/L total hardness; this is likely an error, however, because the U.S. EPA ALC at the time was 7 μg/L (ppb) Cu at 55 mg/L total hardness. The document also states that the background concentration for Cu in the lower Columbia River was 2 mg/L; this is also likely an error and should be 2 μg/L. Model outputs were subjected to a multiple regression analysis to develop an equation relating water column copper concentration in μg/L to flow rate, number of piles, and cross-sectional area. The equation developed was:

$$Cu = 10^{[-0.9988 \log \text{flow (cm/s)} + 0.99846 \log \text{piles} - 0.9984 \log \text{XS } (m^2) + 0.8833]}$$

where:

Cu	= copper concentration (μg/L)
flow	= current velocity in m/sec
piles	= number of piles
XS	= cross sectional areas in m^2.

The conclusions of the analysis for waterborne preservatives were that no adverse impacts to listed salmonids would occur for ACZA and CCA treated pilings if average flows exceeded 1.0 cm/sec and there were fewer than 100 pilings involved in the structure. Also, no adverse impacts were expected if average flows exceeded 10.0 cm/sec and there were between 101 and 350 pilings involved in the structures.

National Marine Fisheries Service, 1998

The NMFS summarized its approach in 1998 for analyzing aquatic projects that propose the use of treated wood: “Position Document for the Use of Treated Wood in Areas within Oregon Occupied by Endangered Species Act Proposed and Listed Anadromous Fish Species” (NMFS, 1998). This document focuses on the chemicals ACZA, CCA, and creosote and their effects on listed salmonids and their habitat in the mainstem Columbia River from McNary Dam to the mouth, as well as other areas where there are proposed and listed salmonids. For ACZA and CCA, copper was the main metal of concern. The analysis focused on the juvenile rearing life history stage and on potential chronic effects.

The framework for the guidelines proceeds as follows (excluding the guidelines focused on creosote):

- Water column guidance for copper was set at 7 ppb at 55 mg/L hardness for behavioral avoidance
- Background water column concentration for copper in the lower Columbia River was determined to be 2 ppb, resulting in a maximum allowed contribution of 5 ppb from treated wood
- pH in receiving waters was required to be 7.0 or above, because of the risk of increased leaching for pH below 7.0
- The sediment guideline was based on the "effects range-low" value of 34 ppm for copper developed by U.S. EPA's Office of Solid Waste and Emergency Response
- Background sediment concentration was determined to be 20 ppm, resulting in a maximum allowed contribution of 14 ppm from treated wood.
- Simple models developed by Ted Poston, Battelle Pacific Northwest National Laboratory, were used "to conservatively predict the level of leachates from ACA, CCA, and creosote treated wood" (NMFS, 1998)
- Factors in the water column models include the number of pilings, pH, cross-sectional area, and current velocity
- Factors in the sediment models include the number of pilings, cross-sectional area, and current velocity.

The following parameters were used in the models to predict water column and sediment copper concentrations:

- Number of pilings: 350, 100, or 24
- Water column pH: 7.2, 7.5, or 8.0
- Cross-sectional area (m^2): 200, 400, 800, or 1,600
- Velocity (cm/sec): 10, 1, 0.5, or 0.3.

All combinations of the parameters listed above were modeled to predict the total water column and sediment copper concentrations, including background plus expected contributions from treated wood. All of the scenarios with either 24 pilings or a current velocity of 10 cm/sec produced acceptable water column and sediment concentrations for copper. Figure 6.1 summarizes the combinations of parameters that resulted in exceedences of the NMFS standards for copper. All of the scenarios that resulted in exceedences of copper for the water column also resulted in exceedences of copper for sediment concentrations.

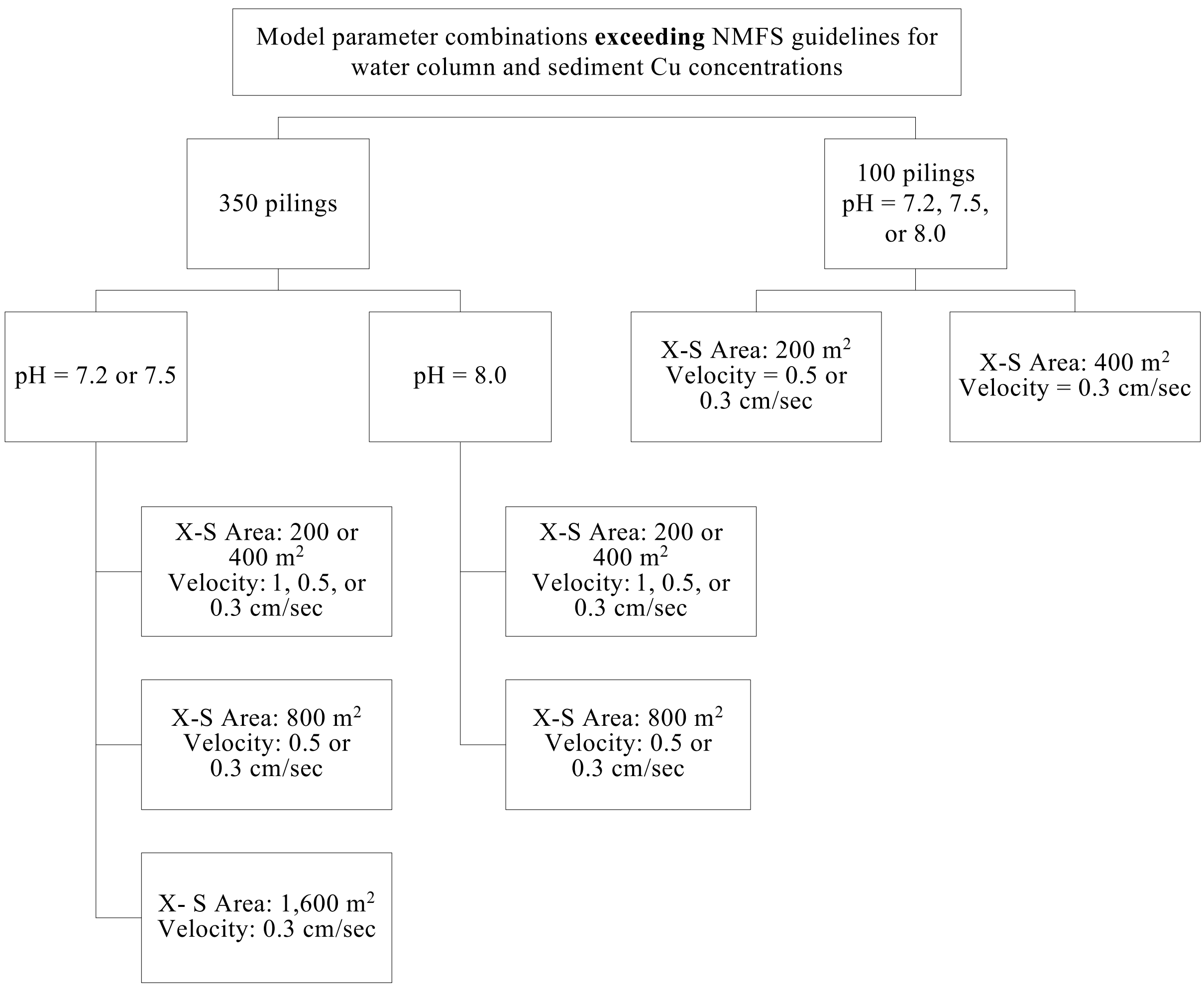

Figure 6.1. Combination of model parameters that would exceed NMFS guidelines for safe water column or sediment copper concentrations.

In the guidelines, NMFS offers the alternatives of either using the tabulated scenarios in the document to apply to a proposed treated wood project or to use multiple regression equations based on site-specific data. According to the guidelines, the multiple regression equation for water column copper is:

$$\mathrm{Cu} = 10^{[-\log \mathrm{flow} + \log \mathrm{piles} - \log \mathrm{XS} + 0.883]}$$

For sediment copper, the multiple regression equation is:

$$\mathrm{Cu} = 10^{[-\log \mathrm{piles} - \log \mathrm{XS} - \log \mathrm{flow} - 0.667]}$$

where for both equations:

Cu = copper concentration (ppb for water column, ppm for sediment)
piles = number of piles
XS = cross sectional areas in m^2
flow = current velocity in m/sec.

Although the document states that the current velocity should be given in m/sec, the results obtained with these values appear unreasonable. If current velocity is given in cm/sec, water concentrations close to those shown from the box models (given in Tables 1 and 2 of NMFS, 1998). These equations are similar to those in the USACE 1996 Biological Assessment.

U.S. Department of Defense, 2001

The U.S. Department of Defense issued an informal programmatic consultation for replacement of up to 18 pilings in Washington State. For projects that conform to a specific set of conditions (discussed below), the NMFS and USFWS concurred that the project will not adversely affect listed fish species or critical habitat. Project conditions relevant to wood treatment include:

- For freshwater excluding the Columbia River mainstem, only nontreated pilings may be used

- In the Columbia River mainstem including Snake River and Baker Bay, only non-treated pilings may be used

- In marine/estuarine waters excluding Baker Bay, no piles treated with creosote or pentachlorophenol may be used.

NOAA Fisheries, 2003

The NOAA Fisheries Northwest Region issued a draft guidance in 2003 for projects that propose the use of treated wood. This draft guidance provides a supplement to the protocols outlined in the 1998 position document, with additional information on potential methods to improve project design, BMPs for general construction, and piling removal. The draft guidance focuses solely on creosote and ACZA. The document discusses the role of NOAA Fisheries in the context of the ESA and the MSA.

The draft guidance includes the statement that "While this guidance contains recommendations on project design and construction, in most settings, a site specific assessment will be needed to determine if the use of treated wood will be consistent with adequate protection of species covered by the ESA and the MSA, their prey base and habitat." This statement is significant because the desire to avoid a formal consultation for a treated wood project can lead an applicant to use alternative materials such as steel or concrete, even if the proposed use of treated wood products would not have exceeded water column or sediment guidelines.

The following design considerations that are relevant to ACZA-treated wood are listed as measures that would minimize or avoid impacts relevant to the ESA and MSA:

- "Use of alternative materials that are less or non-toxic and are also stronger than wood . . ."
- Use untreated wood for temporary structures or as impact cushions
- Seal treated or untreated wood structures with an impervious polymer coating
- Any cutting of treated wood during construction should occur away from aquatic environments
- Wood should be visually inspected to avoid wood with surface residues or bleeding of preservative
- Wood should be certified according to the WWPI BMPs, with the additional requirement of specifying aqua ammonia steaming as the post-treatment process for ACZA
- Removal of remnant treated wood debris could be used as mitigation for new projects that incorporate treated wood.

Oregon State Marine Board, 2002

The Oregon State Marine Board issued BMPs for the design and construction of recreational boating facilities to minimize project impacts to water quality and habitat. The BMPs cover a wide range of topics including facility siting and construction practices, and include several recommendations relevant to treated wood:

- For boat ramp construction or replacement, wood treated with oil-borne preservatives (CuN or creosote) or “other environmentally harmful substances should not be used in the construction of inwater structures”
- For piling material, “round steel pipe is the preferred material”
- “Use of treated, wooden piling should be avoided”
- “Existing wooden piles on the site should be removed whenever possible.”

The document states that the “use of piling with chemical treatment or coatings that leach into the water may be harmful to the aquatic environment,” but does not give citations or offer any specific water column or sediment criteria.

California Coastal Commission, 2003

The California Coastal Commission issued a Coastal Development Permit to Port San Luis Harbor in San Luis Obispo County. The Commission’s recommendations related to treated wood included the provision to phase out ACZA-treated pilings and to require new fender pilings to be made of plastic. The commission noted that plastic pier pilings appeared superior to plastic wrapped chemically treated wood because the plastic wrap used previously had torn off in places and was likely to become floatable marine plastic debris. The justification for limiting the use of ACZA pilings was: “In each of the studies, measurable amounts of preservatives were shown to be released into the environment. While the degree of environmental accumulation and biological impacts appear to be low, some release does occur. Recognizing the potential impacts of using ACZA-treated wood products in the marine environment, a precautionary approach is warranted.”

Dickey, 2003 – San Francisco Department of the Environment

The San Francisco Department of the Environment contracted with the Washington Toxics Coalition “to investigate factors or criteria that might be used to select preferred wood preservative treatments or alternative construction materials for city projects.” The objective of this project was to identify an acceptable list of wood preservatives for each proposed use.

The document developed nine selection criteria for acceptable wood treatments, including:

- Any pressure treated products require standardized treatment by the AWPA.
- Product must not be used in ways that the U.S. EPA prohibits or discourages.
- Products or use must not violate state or local law, policy, or published BMPs.
- Product may not result in the release or creation of dioxins.
- Product or constituents must not be listed on the U.S. EPA priority persistent bioaccumulative toxins list or the U.S. EPA waste minimization priority chemicals list.
- Product or components should not contain known, likely, or probably human carcinogens.
- Product or components should not be listed as reproductive or developmental toxicants by the State of California.
- "For structures built in or over water, or where significant runoff is likely to occur, the use of copper should be minimized. If copper-based products are used, products with the lowest leaching potential should be chosen."
- Product must not be designated as a hazardous waste using California criteria.

Because of the requirement that products must not violate state or local law, ACA, ACZA, and CCA were categorically excluded from use, with an exception made only for saltwater immersion environments. This exclusion results from a resolution passed by the San Francisco Board of Supervisors on November 20, 2001, with several provisions relating to treated wood:

- Arsenic-treated wood is not be used on city projects, with an exception for saltwater immersion environments.
- The Port of San Francisco is required to report back annually on alternatives to arsenic-treated wood for use in saltwater immersion.
- The Port of San Francisco is required to ensure that treated wood used for Port facilities is produced in accordance with the WWPI BMPs.

For freshwater environments, the wood treatment compounds that passed the criteria screening were CuN, ACQ-B (ammoniacal copper quat), ACQ-D (amine copper quat), and CA-B.

Hutton and Samis, 2000 (Fisheries and Oceans Canada)

Fisheries and Oceans Canada staff developed guidelines to assist in the review of shoreline projects involving treated wood. Fisheries and Oceans Canada administers Section 35 of the Canadian Fisheries Act “which prohibits the harmful alteration, disruption or destruction of fish habitat without authorization.” According to the guidelines document, “because the aquatic use of treated wood may have adverse environmental effects,” consultation with appropriate regulatory agencies is required when treated wood projects are proposed.

The overall conclusions of the report included:

- “Alternatives to treated wood should be used wherever practicable”
- “Only wood treated to BMP specifications will be acceptable in or adjacent to aquatic areas.”

The following specific points were listed as the steps needed to conduct a water quality assessment of a proposal for a structure made from treated wood:

- “Consider the environmental risks associated with all types of construction materials”
    - For example, precast concrete structures may be advisable in areas with low current velocities and fine-textured sediments with slow degradation rates for creosote/PAHs, but concern is raised over greater physical impacts on fish habitat
    - Steel can be built with fewer supporting members, but may require repainting and may include sacrificial zinc anodes that may elevate zinc levels
    - Damage to docking ships can be a concern with steel and concrete structures
    - Untreated wood may be suitable for short-term freshwater use
    - Full-pile polyurethane wraps can be used on untreated or treated wood, but ensuring the integrity of the wrapping over time is required
    - Top caps may shield creosote-treated wood from solar heating
    - Consider new technology such as plastic piling, using anchors instead of piling, Superwood (a plastic timber), and Trex (a wood/plastic composite).
- When unacceptable environmental risks to the environment are expected from treated wood, engineers will help field staff evaluate the potential for alternatives.

- If treated wood is necessary, the most appropriate type of treated wood should be identified, considering existing environmental conditions (e.g., use metal oxide treated wood if PAHs are already elevated).
- Metal-oxide treated wood (e.g., CCA, ACZA) is not recommended if water hardness is low (15-25 mg/L), pH is < 5.5, or elevated background metal levels exist.
- Creosote-treated wood discouraged in areas with anaerobic sediment, low TOC, or elevated background PAHs.
- Over-water boring of treated wood should be minimized, with all debris collected and deposited at an approved upland facility.
- Losses of treated wood through abrasion can be minimized with protective wear strips.
- Timing restrictions may help reduce exposure of sensitive life stages to contamination during the initial leaching period.
- All treated wood should be treated according to WWPI BMPs with particular attention to following post-treatment procedures for fixation.

Government of Newfoundland and Labrador, 2001

The Water Resources Management Division in the Government of Newfoundland and Labrador, Canada, issued a policy directive related to treated utility poles in water supply areas. The policy guidelines are focused on public health, not on environmental effects. The guidelines prefer the use of untreated poles, or steel or concrete structures, and require that treated poles be placed outside of a specified buffer zone from the high water mark of any body of water. Appendix A to the guidelines notes that the wood treatment chemicals that will leach from treated wood products "may result in both short-term and long-term environmental and human health hazards."

Western Wood Preservers Institute, 2002b

WWPI produced a 30-page extensively illustrated booklet entitled "Treated Wood in Aquatic Environments." The booklet suggest five steps for the appropriate use of treated wood in aquatic environments, including:

- Selecting the proper preservative and retention levels
- Environmental considerations and evaluations
- Specifying BMPs
- Providing quality assurance and certification
- Appropriate handling, installation, and maintenance.

Selection of the proper preservative and retention levels references the AWPA standards discussed in Section 6.1. Specifying BMPs and providing quality assurance and certification were also discussed in Section 6.1. Appropriate handling, installation, and maintenance were discussed in Section 6.2.

For environmental considerations and evaluations, the document provides guidelines for the number of pilings that could be placed “without jeopardizing the environment,” according to the maximum current speed (Table A in WWPI, 2002b). For example, for a hypothetical project with a pH of 6.5, hardness of 75 mg/L, background copper concentration of 1.5 ppb, and current speed of 0.5 cm/sec, the WWPI risk assessment model would allow 116 CCA-C treated pilings, 25 ACZA treated pilings, 2 ACQ-B treated pilings, and 6 CuN pilings. The guidelines are based on WWPI Risk Assessment models that are not described in the booklet. The WWPI also recommends a full risk assessment for waterborne treatments if the maximum current speed is less than 1.0 cm/sec, if the pH of the receiving water is less than 5.5, or if the project involves more than 100 pilings in parallel with the current for CCA-C, 25 pilings for ACZA, or two pilings for ACQ-B.

6.4 Conclusions

The large number and variety of existing regulations and BMPs for waterborne preservatives in treated wood attest to the importance of minimizing the potential for adverse impacts to surface water, sediments, and aquatic biota from the leaching of chemicals into the aquatic environment. All parties appear to agree that following industry BMPs for treatment procedures in the treating facility, together with third-party certification, are an important first step when considering using treated wood in aquatic environments. Careful construction practices also minimize environmental risks.

The greatest variety in regulations and BMPs relate to guidance on when or whether to use treated wood in aquatic environments. The California Coastal Commission, for example, adopted a precautionary approach to the potential risk from treated wood, and required a complete phaseout of ACZA-treated wood for the Port of San Luis. In contrast, the industry recommendations for treated wood suggest a full risk assessment only for a large number of pilings, slow current speeds, or low pH.

7. Conclusions and Use Recommendations

The results of our literature review suggest that appropriately processed treated wood products[1] can be used in most aquatic environments. Hazardous chemicals, particularly copper, can be leached from treated wood, both from precipitation and submersion. Leaching occurs primarily within the first several months following project completion, and leaching rates often are not sufficiently high to pose ecological risks in well-mixed waters. Environmental accumulation of copper and other leached substances generally is only observable within several meters of the wood structure. Some adverse biological responses, particularly to benthic invertebrates, have been associated with treated wood structures. However, these effects typically attenuate relatively rapidly in space (i.e., within several meters of the structure) and in time (i.e., within several months of project implementation).

Notwithstanding these generalized conclusions, uses of treated wood may not be appropriate in certain circumstances, particularly those in which special status species (including salmonids) – and, in particular, juvenile life stages of salmonids – may come into contact with treated wood leachate, in locations that are poorly mixed or with low current velocities (e.g., < 1 cm/sec), or for projects in which the density of the wood structures is high relative to the surface area of the water body. Given these caveats regarding potential safe uses of treated wood products, application of guidelines and BMPs to ensure environmental protectiveness is warranted.

As described in the previous chapter, a number of governmental and industry guidelines and BMPs already have been developed to address potential hazards of treated wood for aquatic uses. These guidelines and BMPs address the three critical, but distinct, elements related to treated wood use: product manufacture, construction practices, and project appropriateness. The remainder of this chapter contains use recommendations based on our review of existing guidelines and the underlying scientific literature.

Recommendation #1: To ensure environmental protectiveness, guidelines should comprise all three elements related to the safe use of treated wood: product manufacture, construction practices, and project appropriateness

We recommend that all treated wood products used in aquatic habitats conform with manufacturing BMPs (see Recommendation #2), that all projects conform to safe construction BMPs to minimize environmental releases of more readily leached sawdust and construction fragments (see Recommendation #3), and that a project evaluation review be conducted for all

1. Again, it is emphasized that this report focuses on water-based wood treating products, particularly CCA, ACZA, and ACQ.

projects (Recommendation #4), often in conjunction with a site-specific ecological risk assessment (see Recommendation #5), depending on project conditions.

Recommendation #2: All treated wood products used for in- and over-water uses should conform to applicable manufacturing/processing standards

We recommend that all treated wood products used in projects that could affect the aquatic environment conform to the WWPI BMPs described in Chapter 6 of this report (WWPI and CITW, 1996). These industry BMPs specify the treating solutions, treatment procedures, and post-treatment procedures required to maximize fixation of chemicals to the wood. BMP certification requires independent auditing by accredited third-party agencies, which is a key step in ensuring that the BMP practices have been followed. Because the BMPs are linked to the AWPA book of standards that are continuously updated, treated wood BMPs should consistently be improving to track the state-of-the-art methods for wood preservation.

We also recommend improved monitoring of BMP compliance. Proper adherence to the BMPs that have been developed for preservative loading, fixing, and post-fixing processing has been shown to be crucial in reducing the rate and amount of metals that leach from treated wood in service. Preservative that is poorly or incompletely fixed, or excess preservative that is not removed prior to installation, can leach rapidly and in large amounts from wood in service. Therefore, one important recommendation is for adequate monitoring of treated wood processes and products to evaluate the degree to which BMPs are being followed by the industry. The lower the compliance with BMPs related to wood treatment, fixing, and post-fixing processing, the more important site-specific evaluations of the potential for adverse effects to NOAA trust resources becomes.

We recommend that project managers be required to keep records of all BMP certification paperwork, including results of the chromotropic acid test for CCA-treated wood, which tests whether the CCA fixation reaction is complete. In addition, project managers should not request that the wood treater increase the preservative retention rate beyond the standard specified by the AWPA. Project managers also should reject the use of wood with any visible surface residue or wood that has been retreated after initially failing to meet penetration or retention requirements.

Conformance with manufacturing BMPs is a first step toward minimizing potential hazards of treated wood in aquatic environments. Other recommendations must also be followed, however, including conformance with good construction practices and proper project evaluation review.

Recommendation #3: All treated wood projects should conform to construction BMPs

We recommend the development of and adherence to BMPs for construction. Field and laboratory studies have clearly demonstrated that the rate and amount of metals that leach from wood shavings and sawdust are much greater than from whole wood. The available models of metal leaching and subsequent environmental concentrations of metals are based on the assumption that only whole wood is exposed in the environment. Therefore, in cases where shavings and sawdust are being released into aquatic environments during construction, the models on metal leaching and concentrations will underestimate the concentrations of metals in the environment.

BMPs for the prevention of the release of shavings and sawdust during construction are lacking, and should be developed, implemented, and their adherence monitored to reduce the need for site-specific risk assessments and environmental monitoring. The recommendations developed by Lebow and Tippie (2001) provide an excellent framework for development of these BMPs. Construction BMPs should focus on the practices that minimize discharge of debris and field preservative treatment into the environment. Suggested BMPs include:

- Store treated material at the job site above the ground and covered from precipitation. If any ammoniacal preservative odor is noted for ACZA-, ACQ-B-, or CC-treated wood, the wood should be stacked with spacers between layers to aid in volatilization of the ammonia.

- Prefabricate wood away from the field to the maximum extent possible. Ideally, untreated wood should be fabricated and then treated after fabrication. For example, decking and rail posts can be cut or bored before treatment.

- Conduct field fabrication (e.g., drilling or cutting) away from the aquatic environment wherever possible.

- Require a debris collection plan to be included in the construction management plan. This plan should include methods such as tarps and buckets to collect wood sawdust and shavings at the construction site.

- Develop BMPs for the application of field treatment preservatives to untreated wood that is exposed during field fabrication. Field treatment preservatives should be applied sparingly and with a collection device to collect any spills or drips. The BMPs should specify the weather conditions under which it is permissible to apply field preservatives. AWPA Standard M4 gives requirements for field treatment.

Recommendation #4: A screening-level project evaluation review should be conducted to determine whether a site-specific ecological risk assessment is required

We recommend that an initial screening-level project evaluation review be conducted. The purpose of this screening-level review should be to establish conditions – within a conservative, reasonable worst-case framework – that should be met for a project to be acceptable without conducting a site-specific risk assessment. The following conditions should be met to enable a project to proceed without conducting a site-specific risk assessment:

- The project conforms to product and construction BMPs, as identified in Recommendations #2 and #3
- The project location should not provide critical habitat for NOAA trust resources, in particular, early life stages of sensitive salmonids or other anadromous species
- The minimum water hardness at the project location should be greater than 50 mg/L (as $CaCO_3$)
- The pH of the water at the project location should fall within the range of 6.0-8.5
- The water body should not have any pre-existing impairment [e.g., 303(d) or TMDL-listed] for compounds of concern
- Typical minimum current velocities should be greater than 2 cm/sec.

In the event that the proposed project meets any of the above elements, we recommend that a site-specific risk evaluation be conducted to ensure environmental protectiveness.

Recommendation #5: When dictated by the results of the screening-level project evaluation, a site-specific ecological risk assessment should be conducted to ensure environmental protectiveness

Based on the results of the screening-level project evaluation, site-specific ecological risk assessments may be necessary to ensure environmental protectiveness. Such site-specific risk assessments should include the following elements:

- *Project description*.
- *Description of ecological receptors*. This description should include identification of habitat types, biological receptors, and life history descriptions that identify the life stages of all NOAA trust species within the project location.
- *Exposure evaluation*. This evaluation should include quantitative modeling of both short- and long-term leaching and accumulation of metals using the models described in Chapter 2. We recommend that reasonable worst-case scenarios be used for parameter inputs when conducting such evaluations (e.g., minimum current velocities, maximum temperature, minimum site hardness).

- *Hazard evaluation*. This should include quantitative numerical hazard evaluation by screening predicted concentrations against the benchmarks described in Chapter 3. If behavioral avoidance is a potential concern for salmonids, a qualitative discussion of potential avoidance responses within the project location should be included.

- *Uncertainty analysis*. This should include a discussion of key uncertainties in the risk assessment.

As discussed in Chapter 4, the following factors also should be considered when conducting site-specific risk assessments:

- *Background water quality variables, including temperature, hardness, pH, and salinity*. The toxicity of copper (and zinc) is dependent on water hardness (and other related water quality variables). Although the absolute concentration of copper leached from treated wood may not be substantial, low levels of copper can cause toxicity in soft, calcium-poor waters. Moreover, hardness can vary at locations over the annual hydrograph. As a result, site-specific risk assessments should consider the projected ambient hardness (and calcium) concentrations and pH relative to the timing of releases. Worst-case scenarios (i.e., lowest annual hardness) may be used to ensure adequate protectiveness. The salinity of the receiving environment should also be considered. Leaching rates of some chemicals from treated wood are salinity dependent, with most metals leaching at increased rates compared to freshwater. In estuarine environments with fluctuating salinities, a worst-case scenario should be used to ensure protectiveness of special status species.

- *Current velocity and direction.* Although total leach rates from treated wood can be relatively low, potential environmental effects will be dictated by local water mixing, with poorly-mixed waters being at greater risk. Information on current velocities at the specific micro-environment of the project location (including the influence of the structure itself on ambient current velocities), should be developed and integrated into a site-specific risk evaluation.

- *Proximity to sensitive fish habitat.* The presence of sensitive life stages, especially T&E species or their essential prey species, should prompt an evaluation of potential risks at that location. Essential fish habitats for Pacific salmon include all streams, lakes, and other water bodies currently or historically accessible to salmon. The most sensitive life stages for these species are fry (particularly post swim-up) and juveniles. Because the initial leach rates are higher for treated wood, risk assessments should consider the timing of metals releases from construction projects relative to the time periods when sensitive life stages of fish are present.

- *Timing of proposed construction.* Because initial leach rates tend to be greater, the timing of proposed construction should be considered with respect to the presence of sensitive life stages of aquatic receptors; water flow rates; environmental/climatic factors that can influence mixing and dilution; and the relationship between season, annual hydrograph, and water quality conditions.

- *Size of proposed structure.* As discussed previously, environmental effects are likely to be greatest when the size of the proposed structure is large relative to the receiving environment. Factors to consider include number and size of pilings; surface area of exposed wood area relative to the mixing zone; density of pilings relative to the mixing zone (to evaluate potential behavioral avoidance responses); and potential effects of structure size on current flows.

- *Proximity of other preserved-wood structures and other sources of contamination that may contribute to cumulative effects.* In evaluations of site-specific risks, assessments should consider potential effects in light of the cumulative effect of the proposed structure relative to other existing environmental perturbations at the site.

Recommendation #6: Additional research is needed to address outstanding scientific uncertainties

The results of our information review indicate that a number of uncertainties remain regarding potential environmental effects of treated wood. We recommend that research continue to better address these uncertainties. Key areas where additional research is needed are described below.

- Factors that affect the variability of metals leaching from treated wood:

    - Research is needed to investigate how the leaching of metals varies under environmentally realistic water conditions (e.g., the presence of fulvic and humic acids, fluctuating hardness levels, wetting and drying cycles).

    - Research is needed to investigate how the leaching of metals varies as a function of commercial variation in treatment, fixation, and post-fixation processing, especially where multiple techniques are allowed as part of BMPs.

    - Research is needed to evaluate the potential effects of physical abrasion on metals releases and whether abraded wood can be leached more readily under site-specific redox conditions in field sediments.

- Factors that affect the toxicity of releases in field conditions:

    - Research is needed to determine which metals species are released during the leaching process. Toxicity can vary as a function of metal speciation.

    - Research is needed to determine the ecological importance of behavioral avoidance responses in settings where treated wood can be used. For example, releases may result in potential avoidances of overhanging and submerged structures that could provide refugia from predators. Avoidance of overwater structures may reduce the use of certain habitats.

- Potential for use of alternative materials:

    - Ongoing research is needed to characterize the relative environmental benefits and risks of alternative materials to treated wood. Novel products such as the use of sodium silicate to create nontoxic pressure treated wood should be investigated to determine if these products are suitable for use in aquatic environments.

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A. The Relative Sensitivity to Copper of 62 Genera of Freshwater Organisms, Ranked by GMAV

Fish species rank numbers are in bold and genus *Oncorhynchus* species data are also in bold. Acute values based on a hardness of 50 mg/L (as $CaCO_3$).

Rank	Genus mean acute value (µg/L)	Species	Species mean acute value (µg/L)
62	59,490	Golden shiner, *Notemigonus crysoleucas*	59,490
61	9,382	Stonefly, *Acroneuria lycorias*	9,382
60	3,543	Crayfish, *Cambarus robustus*	3,543
59	3,223	American eel, *Anguilla rostrata*	3,223
58	1,881	Snail, *Campeloma decisum*	1,881
57	1,771	Crayfish, *Procambarus clarkii*	1,771
56	1,711	Bluegill, *Lepomis macrochirus*	1,711
55	1,373	Crayfish, *Orconectes rusticus*	1,373
54	1,290	Amphipod, *Crangonyx pseudogracilis*	1,290
53	804.8	Mosquitofish, *Gambusia affinis*	804.8
52	789.6	Banded killifish, *Fundulus diaphanus*	789.6
51	675.1	Mozambique tilapia, *Tiliapia mossambica*	675.1
50	631.3	Pumpkinseed, *Lepomis gibbosus*	631.3
		Midge, *Chironomus decorus*	835.3
49	411.3	Midge, *Chironomus tentans*	202.6
48	386.7	Asiatic clam, *Corbicula fluminea*	386.7
47	324.4	Striped shiner, *Notropis chrysocephalus*	324.4
46	288.9	Goldfish, *Carassius auratus*	288.9
45	207.4	White sucker, *Catostomus commersoni*	207.4
44	206.9	Longfin dace, *Agosia chrysogaster*	206.9
43	189.9	Flagfish, *Jordanella floridae*	189.9
		White perch, *Morone americana*	5,852
42	168.1	Striped bass, *Morone saxatilis*	168.1
41	149.8	Common carp, *Cyprinus carpio*	149.8

Rank	Genus mean acute value (µg/L)	Species	Species mean acute value (µg/L)
40	138.2	Snail, *Goniobasis livescens*	138.2
		Rainbow darter, *Etheostoma caeruleum*	84.73
		Fantail darter, *Etheostoma flabellare*	110.8
		Johnny Darter, *Etheostoma nigrum*	163.8
39	136.4	Orangethroat darter, *Etheostoma spectabile*	225.1
38	135.6	Copepod, *Tropocyclops prasinus mexicanus*	135.6
37	132.8	Chiselmouth, *Acrocheilus alutaceus*	132.8
36	131.9	Bryozoan, *Lophopodella carteri*	131.9
		Brown bullhead, *Ictalurus nebulosus*	68.02
35	131.2	Channel catfish, *Ictalurus punctatus*	252.9
34	111.9	Atlantic salmon, *Salmo salar*	111.9
33	111.2	Freshwater mussel, *Anodonta imbecilis*	111.2
32	110.6	Brook trout, *Salvelinus fontinalis*	110.6
31	97.50	Razorback sucker, *Xyrauchen texanus*	97.50
30	84.73	Blacknose dace, *Rhinichthys atratulus*	84.73
29	82.51	Guppy, *Poecilia reticulata*	82.51
28	82.08	Creek chub, *Semotilus atromaculatus*	82.08
27	77.15	Bonytail, *Gila elegans*	77.15
26	76.79	Central stoneroller, *Campostoma anomalum*	76.79
25	72.34	Worm, *Lumbriculus variegatus*	72.34
24	70.28	Freshwater mussel, *Pygranodon grandis*	70.28
		Bluntnose minnow, *Pimephales notatus*	69.66
23	69.58	Fathead minnow, *Pimephales promelas*	72.30
22	63.06	Cladoceran, *Simocephalus vetulus*	63.06
		Cutthroat trout, *Oncorhynchus clarki*	**68.64**
		Coho salmon, *Oncorhynchus kisutch*	**108.10**
		Rainbow trout, *Oncorhynchus mykiss*	**41.51**
		Sockeye salmon, *Oncorhynchus nerka*	**48.46**
21	**57.16**	**Chinook salmon, *Oncorhynchus tshawytscha***	**40.87**
20	55.57	Snail, *Gyraulus circumstriatus*	55.57

Rank	Genus mean acute value (µg/L)	Species	Species mean acute value (µg/L)
19	52.49	Tubificid worm, *Limnodrilus hoffmeisteri*	52.49
		Colorado squawfish, *Ptychocheilus lucius*	120.40
18	47.7	Northern squawfish, *Ptychocheilus oregonensis*	18.90
		Snail, *Physa heterostropha*	35.51
17	39.14	Snail, *Physa integra*	43.14
16	36.20	Bryozoan, *Pectinatella magnifica*	36.20
15	36.20	Bryozoan, *Plumatella emarginata*	36.20
14	35.75	Tubificid worm, *Tubifex tubifex*	35.75
13	35.41	Amphipod, *Hyallela azteca*	35.41
12	34.45	Snail, *Juga plicifera*	34.45
11	24.77	Amphipod, *Gammarus pseudolimnaeus*	24.77
10	24.13	Freshwater mussel, *Actinonaias pectorosa*	24.13
9	23.42	Arctic grayling, *Thymallus arcticus*	23.42
8	19.88	Freshwater mussel, *Villosa iris*	19.88
7	18.38	Snail, *Lithoglyphus virens*	18.38
6	17.34	Cladoceran, *Scapholeberis* spp.	17.34
5	14.87	Freshwater mussel, *Lampsilis fasciola*	14.87
		Cladoceran, *Ceriodubia reticulata*	12.47
4	13.21	Cladoceran, *Ceriodaphnia dubia*	13.99
3	13.02	Freshwater mussel, *Medionidus conradicus*	13.02
		Cladoceran, *Daphnia magna*	13.16
		Cladoceran, *Daphnia pulex*	13.05
2	11.60	Cladoceran, *Daphnia pulicaria*	9.085
1	2.329	Cladoceran, *Bosmina longirostrus*	2.329

B. Relative Sensitivity of Various Saltwater Species and Genera to Copper Acute Toxicity

Fish species rank numbers are in bold as are coho salmon species data.

Rank	Genus mean acute value (µg/L)	Species	Species mean acute value dissolved (µg/L)
44	6,386	Common rangia, *Rangia cuneata*	6,386
43	4,697	Atlantic croaker, *Micropogon undulatus*	4,697
42	2,282	Pinfish, *Langodon rhomboides*	2,282
41	1,690	Killifish, *Fundulus heteroclitus*	1,690
40	1,418	Mangrove rivulus, *Rivulus marmoratus*	1,418
39	946.2	Sheepshead, *Archosargus probatocephalus*	946.2
38	745.3	Sand shrimp, *Crangon* spp.	745.3
37	**498.8**	**Coho salmon (smolt), *Oncorhynchus kisutch***	**498.8**
36	498.0	Amphipod, *Corophium insidiosum*	498.0
35	498.0	Green crab, *Carcinus maenas*	498.0
34	396.4	Polychaete worm, *Hediste diversicolor*	396.4
33	346.9	Shiner perch, *Cymatogaster aggregata*	346.9
32	341.7	Florida pompano, *Trachinotus carolinus*	341.7
31	305.4	Sheepshead minnow, *Cyprinodon variegatus*	305.4
30	256.5	Squid, *Loligo opalescens*	256.5
29	232.4	Spot, *Leiostomus xanthurus*	232.4
28	215.8	Nematode (3-4 day), *Caenorhabditis elegans*	215.8
27	207.5	Amphipod, *Elasmopus bampo*	207.5
26	201.7	Topsmelt, *Atherinops affinis*	201.7
25	195.4	Copepod, *Pseudodiaptomus coronatus*	195.4
24	178.0	Copepod, *Tisbe furcata*	178.0
23	130.8	Mysid, *Mysidopsis bahia*	146.3
		Mysid, *Mysidopsis bigelowi*	117.0
22	125.0	Polychaete worm, *Neanthes arenaceodentata*	125.0
21	113.2	Tidewater silverside, *Menidia peninsulae*	116.2
		Atlantic silverside, *Menidia menidia*	112.5

Rank	Genus mean acute value (µg/L)	Species	Species mean acute value dissolved (µg/L)
		Inland silverside, *Menidia beryllina*	111.1
20	112.6	Mysid, *Neomysis mercedis*	112.6
19	107.0	Winter flounder, *Pseudopleuronectes americanus*	107.0
18	99.60	Polychaete worm, *Phyllodoce maculata*	99.60
17	78.85	Cabezon, *Scorpaenichthys marmoratus*	78.85
16	56.93	Dungeness crab, *Cancer magister*	56.93
15	54.45	Red abalone, *Haliotis rufescens*	71.44
		Black abalone, *Haliotis cracherodii*	41.50
14	42.33	Surf clam, *Spisula solidissima*	42.33
13	32.37	Soft-shell clam, *Mya arenaria*	32.37
12	40.67	American lobster, *Homarus americanus*	40.67
11	40.50	Copepod, *Acartia clausi*	40.50
10	25.50	Copepod, *Acartia tonsa*	25.50
9	24.07	Bay scallop, *Argopecten irradians*	24.07
8	23.95	Copepod, *Eurytemora affinis*	23.95
7	17.70	Coot clam, *Mulinia lateralis*	17.70
6	14.11	Mysid, *Holmesimysis costata*	14.11
5	13.40	Sea urchin, *Strongylocentrotus purpuratus*	13.40
4	11.99	Eastern oyster, *Crassostrea virginica*	14.35
		Pacific oyster, *Crassostrea gigas*	10.01
3	11.56	Summer flounder, *Paralichthys dentatus*	11.56
2	7.89	Blue mussel, *Mytilus edulis*	7.89
1	5.81	Bivalve mollusk, *Isognomen californicum*	5.81

C. Other Effects of Copper on Species in the Genus *Oncorhynchus,* Including Avoidance, Growth, Incipient Lethal Levels, Olfaction, and Physiology

Species	Effect	Concentration (µg/L)[a]	Duration	Reference
Cutthroat trout (3-5 mo), *Oncorhynchus clarkii*	Avoidance of copper	7.4	20 min	Woodward et al., 1997
Rainbow trout (7 cm), *Oncorhynchus mykiss*	Avoidance	10.5	20 min	Giattina et al., 1982
Rainbow trout, *Oncorhynchus mykiss*	Avoidance threshold	33.1	80 min	Black and Birge, 1980
Coho salmon (7.8 cm), *Oncorhynchus kisutch*	15% reduction in growth	13.4	14 wk	Buckley et al., 1982
Coho salmon (parr), *Oncorhynchus kisutch*	NOEC growth and survival	41.5	61 days	Mudge et al., 1993
Coho salmon (parr), *Oncorhynchus kisutch*	NOEC growth and survival	49.7	61 days	Mudge et al., 1993
Coho salmon, *Oncorhynchus kisutch*	NOEC growth and survival	27.0	60 days	Mudge et al., 1993
Rainbow trout (parr), *Oncorhynchus mykiss*	NOEC growth and survival	135.6	62 days	Mudge et al., 1993
Chinook salmon (alevin), *Oncorhynchus tshawytscha*	LC_{10}	29.9	200 hr	Chapman, 1978
Chinook salmon (parr), *Oncorhynchus tshawytscha*	LC_{10}	33.9	200 hr	Chapman, 1978
Chinook salmon (smolt), *Oncorhynchus tshawytscha*	LC_{10}	35.9	200 hr	Chapman, 1978
Chinook salmon (swimup), *Oncorhynchus tshawytscha*	LC_{10}	27.9	200 hr	Chapman, 1978
Rainbow trout (1.7-3.3 g), *Oncorhynchus mykiss*	Incipient lethal level	47.4	144 hr	Dixon and Sprague, 1981
Rainbow trout (1.7-3.3 g), *Oncorhynchus mykiss*	Incipient lethal level	48.0	144 hr	Dixon and Sprague, 1981
Rainbow trout (1.7-3.3 g), *Oncorhynchus mykiss*	Incipient lethal level	44.8	144 hr	Dixon and Sprague, 1981

Species	Effect	Concentration (µg/L)[a]	Duration	Reference
Rainbow trout (1.7-3.3 g), *Oncorhynchus mykiss*	Incipient lethal level	39.5	144 hr	Dixon and Sprague, 1981
Rainbow trout (1.7-3.3 g), *Oncorhynchus mykiss*	Incipient lethal level	53.5	144 hr	Dixon and Sprague, 1981
Rainbow trout[b] (1.7-3.3 g), *Oncorhynchus mykiss*	Incipient lethal level	38.3	144 hr	Dixon and Sprague, 1981
Rainbow trout[b] (1.7-3.3 g), *Oncorhynchus mykiss*	Incipient lethal level	50.3	144 hr	Dixon and Sprague, 1981
Rainbow trout[b] (1.7-3.3 g), *Oncorhynchus mykiss*	Incipient lethal level	74.2	144 hr	Dixon and Sprague, 1981
Rainbow trout[b] (1.7-3.3 g), *Oncorhynchus mykiss*	Incipient lethal level	81.3	144 hr	Dixon and Sprague, 1981
Rainbow trout[b] (1.7-3.3 g), *Oncorhynchus mykiss*	Incipient lethal level	102.1	144 hr	Dixon and Sprague, 1981
Rainbow trout (alevin), *Oncorhynchus mykiss*	LC_{10}	36.4	200 hr	Chapman, 1978
Rainbow trout (embryo), *Oncorhynchus mykiss*	EC_{10} (death or deformity)	8.2	28 days	Birge et al., 1980
Rainbow trout (parr), *Oncorhynchus mykiss*	LC_{10}	14.8	200 hr	Chapman, 1978
Rainbow trout (smolt), *Oncorhynchus mykiss*	LC_{10}	12.9	200 hr	Chapman, 1978
Rainbow trout (swimup), *Oncorhynchus mykiss*	LC_{10}	17.3	200 hr	Chapman, 1978
Coho salmon (smolts > 10 cm), *Oncorhynchus kisutch*	Decrease in downstream migration after release	2.7	165 days	Lorz and McPherson, 1976
Rainbow trout (18.5-26.5 cm), *Oncorhynchus mykiss*	55% depressed olfactory response	27.6	2 hrs	Hara et al., 1976
Rainbow trout (2.9 g), *Oncorhynchus mykiss*	Inhibited avoidance of serine	10.2	ca. 2 hr	Rehnberg and Schreck, 1986
Rainbow trout (embryo), *Oncorhynchus mykiss*	Inhibited olfactory discrimination	17.0	7-9 mo	Saucier et al., 1991a
Rainbow trout (embryo), *Oncorhynchus mykiss*	Lesions in olfactory rosettes	17.0	7-9 mo	Saucier et al., 1991b
Rainbow trout (swimup), *Oncorhynchus mykiss*	Inhibited olfactory discrimination	15.9	13-40 wk	Saucier and Astic, 1995

Species	Effect	Concentration (µg/L)[a]	Duration	Reference
Rainbow trout (yearling), *Oncorhynchus mykiss*	Olfactory receptor degeneration	15.4	15 days	Julliard et al., 1993
Rainbow trout (200-250 g), *Oncorhynchus mykiss*	Altered liver and blood enzymes and mitochondrial function	5.0	4 mo	Arillo et al., 1984
Rainbow trout (53.5 g), *Oncorhynchus mykiss*	Transient decrease in food consumption	14.8	15 days	Lett et al., 1976
Rainbow trout (yearling), *Oncorhynchus mykiss*	Elevated plasma cortisol returned to normal	58.1	21 days	Munoz et al., 1991
Sockeye salmon (yearling), *Oncorhynchus nerka*	Drastic increase in plasma corticosteroids	235.7	1-24 hr	Donaldson and Dye, 1975
Coho salmon (smolts), *Oncorhynchus kisutch*	Decrease in survival upon transfer to 30 ppt seawater	10.9	144 hr	Lorz and McPherson, 1976

a. Adjusted to a hardness of 50 mg/L.
b. Fish were pre-acclimated to copper.

D. The Relative Sensitivity to Cr(VI) of 19 Genera of Freshwater Organisms

Fish species rank numbers and genus *Oncorhynchus* species data are in bold.

Rank	Genus mean acute value (µg/L)	Species	Species mean acute value (µg/L)
19		Green sunfish, *Lepomis cyanellus*	114,700
	123,500	Bluegill, *Lepomis macrochirus*	132,890
18	116,500	Goldfish, *Carassius auratus*	116,500
17	**69,000**	**Rainbow trout, *Oncorhynchus mykiss***	**69,000**
16	59,000	Brook trout, *Salvelinus fontinalis*	59,000
15	57,300	Midge, *Tanytarsus dissimilis*	57,300
14	43,100	Fathead minnow, *Pimaphales promelas*	43,100
13	30,450	Striped bass, *Morone saxatilis*	30,450
12	30,000	Guppy, *Poecilia reticulate*	30,000
11	23,010	Snail, *Physa heterostropha*	23,010
10	9,921	Rotifer, *Philodina acuticornis*	9,921
9	1,560	Bryozoan, *Lophopodelia carteri*	1,560
8	1,440	Bryozoan, *Pectinatella magnifica*	1,440
7	650	Bryozoan, *Plumatella emarginata*	650
6	630	Scud, *Hyalella azteca*	630
5	163	Scud, *Crangonyx pseudogracilis*	163
4	67.1	Scud, *Gammarus pseudolimnaeus*	67.1
3	45.2	Cladoceran, *Ceriodaphnia reticulata*	45.2
		Cladoceran, *Simocephalus serrulatus*	40.9
2	36.35	Cladoceran, *Simocephalus vetulus*	32.3
		Cladoceran, *Daphnia pulex*	36.3
1	29.63	Cladoceran, *Daphnia magna*	24.2

E. The Relative Sensitivity to Cr(VI) of 17 Genera of Seawater Organisms

Fish species rank numbers are in bold.

Rank	Genus mean acute value (µg/L)	Species	Species mean acute value (µg/L)
17	105,000	Mud snail, *Nassarium obsoletus*	105,000
16	93,400	Blue crab, *Callinectes sapidus*	93,400
15	91,000	Mummichog, *Fundulus heteroclitus*	91,000
14	57,000	Soft-shell clam, *Mya arenaria*	57,000
13	32,000	Starfish, *Asterias forbesi*	32,000
12	30,500	Speckled sanddab, *Citharicthys stigmaeus*	30,500
11	22,140	Brackish-water clam, *Rangia cuneata*	22,140
10	15,280	Atlantic silverside, *Menidia menidia*	15,280
9	10,000	Hermit crab, *Pagurus longicarpus*	10,000
8	7,500	Plychaete worm, *Orphryotrocha diadema*	7,500
7	6,600	Copepod, *Acartia clause*	6,600
6	6,325	Polychaete worm, *Capitella capitata*	6,325
5	4,300	Polychaete worm, *Ctenodrillus serratus*	4,300
4	3,650	Copepod, *Pseudodiaptomus coronatus*	3,650
		Mysid, *Mysidopsis bahia*	2,033
3	2,991	Mysid, *Mysidopsis bigelowi*	4,400
2	3,100	Polychaete worm, *Neanthes arenaceodentata*	3,100
1	2,390	Polychaete worm, *Nereis virens*	2,390

F. The Relative Sensitivity to Cr(III) of 17 Genera of Freshwater Organisms

Fish species rank numbers are in bold and genus *Oncorhynchus* species data are also in bold.

Rank	Genus mean acute value (µg/L)	Species	Species mean acute value (µg/L)
17	71,600	Caddisfly, *Hydropsyche betteni*	71,600
16	43,100	Damselfly, Unidentified	43,100
15	16,010	Cladoceran, *Daphnia magna*	16,010
14	15,630	Banded killifish, *Fundulus diaphanus*	15,630
		Pumpkinseed, *Lepomis gibbosus*	15,720
13	15,370	Bluegill, *Lepomis macrochirus*	15,020
		White perch, *Morone Americana*	13,320
12	14,770	Striped bass, *Morone saxitili*s	16,370
11	13,230	Common carp, *Cyprinus carpio*	13,230
10	12,860	American eel, *Anguilla rostrata*	12,860
9	11,000	Midge, *Chironomus* spp.	11,000
8	10,320	Fathead minnow, *Pimephales promelas*	10,320
7	**9,863**	**Rainbow trout, *Oncorhynchus mykiss***	**9,863**
6	9,300	Worm, *Nais* spp.	9,300
5	8,684	Goldfish, *Carrasius auratus*	8,684
4	8,400	Snail, *Amnicola* spp.	8,400
3	7,053	Guppy, *Poecilia reticulata*	7,053
2	3,200	Scud, *Gammarus* spp.	3,200
1	2,200	Mayfly, *Ephemerella subvaria*	2,200

G. The Relative Sensitivity to Zn of 35 Genera of Freshwater Organisms

Fish species rank numbers are in bold and genus *Oncorhynchus* species data are also in bold. Acute values normalized to a hardness of 50 mg/L (as $CaCO_3$).

Rank	Genus mean acute value (µg/L)	Species	Species mean acute value (µg/L)
35	88,960	Damselfly, *Argia* spp.	
34	19,800	Amphipod, *Crangonyx pseudogracilis*	
33	18,400	Worm, *Nais* spp.	
32	17,940	Banded killifish, *Fundulus diaphanus*	
31	16,820	Snail, *Amnicola* spp.	
30	13,630	American eel, *Anguilla rostrata*	
29	10,560	Pumpkinseed, *Lepomis gibbosus*	18,790
		Bluegill, *Lepomis macrochirus*	5,937
28	10,250	Goldfish, *Carrasius auratus*	
27	9,712	Worm, *Lumbriculus variegatus*	
26	8,157	Isopod, *Asellus bicrenata*	5,751
		Isopod, *Asellus communis*	11,610
25	8,100	Amphipod, *Gammarus* sp.	
24	7,233	Common carp, *Cyprinus carpio*	
23	6,580	Northern squawfish, *Ptychocheilus oregonensis*	
22	6,053	Guppy, *Poecilia reticulata*	
21	6,000	Golden shiner, *Notemigonus crysoleucas*	
20	5,228	White sucker, *Catostomus commersoni*	
19	4,900	Asiatic clam, *Coricula fluminea*	
18	4,341	Southern platyfish, *Xiphophorus maculatus*	
17	3,830	Fathead minnow, *Pimephales promelas*	
16	3,265	Isopod, *Lirceus alabamae*	
15	2,176	Atlantic salmon, *Salmo salar*	
14	2,100	Brook trout, *Salvelinus fontinalis*	
13	1,707	Bryozoan, *Lophopodella carteri*	

Rank	Genus mean acute value (µg/L)	Species	Species mean acute value (µg/L)
12	1,672	Flagfish, *Jordanella floridae*	
11	1,607	Bryozoan, *Plumatella rostrata*	
10	1,578	Snail, *Heliosoma campanulatum*	
9	1353	Snail, *Physa gyrina*	1,683
		Snail, *Physa heterostropha*	1,088
8	1,307	Bryozoan, *Pectinetella magnifica*	
7	>1,264	Tubificid worm, *Limnoddrilus hoffmeisteri*	
6		**Rainbow trout, *Oncorhynchus mykiss***	**689.3**
		Coho salmon, *Oncorhynchus kisutch*	**1,628**
		Sockeye salmon, *Oncorhynchus nerka*	**1,502**
		Chinook salmon, *Oncorhynchus tshawytscha*	**446.4**
5	790	Mozambique tilapia, *Tilapis mossambica*	
4	299.8	Cladoceran, *Daphnia magna*	355.5
		Cladoceran, *Daphnia pulex*	252.9
3	227.8	Longfin dace, *Agosia chrysogaster*	
2	119.4	Striped bass, *Morone saxitilis*	119.4
1	93.95	Cladoceran, *Ceriodaphnia dubia*	174.1
		Cladoceran, *Ceriodaphnia reticulate*	50.70

H. The Relative Sensitivity to Zn of 28 Genera of Saltwater Organisms

Fish species rank numbers are in bold and genus *Oncorhynchus* species data are also in bold.

Rank	Genus mean acute value (µg/L)	Species	Species mean acute value (µg/L)
28	320,400	Clam, *Macoma balthica*	320,400
27	50,000	Mud snail, *Nassarius obsoletus*	50,000
26	39,000	Starfish, *Asterias forbesii*	39,000
25	38,000	Spot, *Leiostomus zanthurus*	38,000
24	36,630	Mummichog, *Fundulus heteroclitus*	36,630
23	9,467	Winter flounder, *Pseudopleuronectes americanus*	9,467
22	7,100	Polychaete worm, *Ctenodrilus* spp.	7,100
21	6,328	Soft-shell clam, *Mya arenaria*	6,328
20	4,683	Amphipod, *Corohium volutator*	4,683
19	4,515	Atlantic silverside, *Menidia menidia*	3,640
		Tidewater silverside, *Menidia peninsulae*	5,600
18	8,856	Polychaete worm, *Nereis diversicolor*	9,682
		Polychaete worm, *Nereis virens*	8,100
17	4,074	Copepod, *Eurytemora affinis*	4,074
16	3,934	Blue mussel, *Mytilus edulis*	3,934
15	2,439	Polychaete worm, *Capitella capitata*	2,439
14	>1,920	Squid, *Loligo opalescens*	>1,920
13	1,450	Copepod, *Nitocra spinipes*	1,450
12	1,400	Polychaete worm, *Ophryotroha diadema*	1,400
11	1,273	Polychaete worm, *Neathes arenaceodentata*	1,273
10	1,000	Green crab, *Carcinus maenus*	1,000
9	665.9	Copepod, *Acartia clause*	1,507
		Copepod, *Acartia tonsa*	294.2
8	586.1	Dungeness crab, *Cancer magister*	586.1
7	543.2	Mysid, *Mysidosis bahia*	499
		Mysid, *Mysidopsis bigelowi*	591.3

Rank	Genus mean acute value (µg/L)	Species	Species mean acute value (µg/L)
6	430	Striped bass, *Morone saxatilis*	430
5	400	Hermitcrab, *Pagurus longicarpus*	400
4	380.5	Lobster, *Homarus americanus*	380.5
3	247.5	Pacific oyster, *Crassostrea gigas*	233.3
		Eastern oyster, *Crassostrea virginica*	262.5
2	195	Quahog clam, *Mercenaria mercenaria*	195
1	191.4	Cabezon, *Scorpaenichthys marmoratus*	191.4

I. The Relative Sensitivity to As(III) of 12 Genera of Freshwater Organisms [two values for As(V) are included]

Fish species rank numbers are in bold and genus *Oncorhynchus* species data are also in bold.

Rank	Genus mean acute value (µg/L)	Species	Species mean acute value (µg/L)
12	41,700	Bluegill, *Lepomis macrochirus*	41,700
11	28,130	Flagfish, *Jordanella floridae*	28,130
10	26,042	Goldfish, *Carrasium auratus*	26,042
9	22,040	Stonefly, *Pteronarcys californica*	22,040
8	18,096	Catfish, *Ictalurus punctatus*	18,096
7	15,660	Fathead minnow, *Pimephales promelas*	15,660
6	14,964	Brook trout, *Salvelinus fontinalis*	14,964
5	**13,340**	**Rainbow trout, *Salmo gairdneri***	**13,340**
	As(V)	**Rainbow trout, *Salmo gairdneri***	**10,800**
4	5,278	Cladoceran, *Daphnia magna*	5,278
	As(V)	Cladoceran, *Daphnia magna*	7,400
3	1,348	Cladoceran, *Daphnia pulex*	1,348
2	879	Scud, *Gammarus pseudolimnaeus*	879
1	812	Cladoceran, *Simocephalus serrulatus*	812

J. The Relative Sensitivity to As(III) of 5 Genera of Saltwater Organisms

Fish species rank numbers are in bold.

Rank	Genus mean acute value (µg/L)	Species	Species mean acute value (µg/L)
5	16,033	Atlantic silverside, *Menidia menidia*	16,033
4	14,953	Fourspine stickleback, *Apeltes quadracus*	14,953
3	7,500	American oyster, *Crassostrea virginica*	7,500
2	3,490	Bay scallop, *Argopecten irradians*	3,490
1	508	Copepod, *Acartia clausi*	508

K. Other Effects of As(III) on Aquatic Organisms

Species	Effect	Conc. (µg/L)	Duration	Reference
Amphipod, *Gammarus pseudolimnaeus*	80% mortality	961	7 d	Spehar et al., 1980
Toad (embryo-larval), *Gastrophryne carolinensis*	LC_{50}	40	7 d	Birge, 1978
Rainbow trout (embryo-larval), *Oncorhynchus mykiss*	LC_{50}	540	28 d	Birge, 1978
Goldfish (embryo-larval), *Carrasius auratus*	LC_{50}	490	7 d	Birge, 1978
Bluegill (fingerling), *Lepomis macrochirus*	LC_{50}	290	48 h	Hughes and Davis, 1967
Pink salmon (saltwater), *Oncorhynchus gorbuscha*	LC_{54}	3,787	10 d	Holland et al., 1960
Chum salmon (saltwater), *Oncorhynchus keta*	LC_{50}	8,330	48 h	Alderdice and Brett, 1957

L. A Summary of the Available Sediment Quality Criteria and Guidelines for the Protection of Aquatic Life

Table L.1. A summary of the available sediment quality criteria and guidelines for the protection of aquatic life

Chemical name	Water type	Guideline (mg/kg)	Approach	Application	Jurisdiction	Reference
Freshwater						
Arsenic	FW/SW	3	SBA	Sediment Quality Criterion; No Effect Threshold; Dry weight	St. Lawrence River, Canada	MENVIQ/EC, 1992
	FW	3	SQG	Non-polluted; USEPA Region 5 Harbour Classification; Dry weight	United States	U.S. EPA, 1977
	FW	3-8	SQG	Moderately Polluted; USEPA Region 5 Harbour Classification; Dry weight	United States	U.S. EPA, 1977
	FW	4.25	SLCA	Sediment Quality Guideline	Canada	Hart et al., 1988
	FW	5	SBA	USEPA Region VI Proposed Guideline for Sediment Disposal	United States	Pavlou and Weston, 1983
	FW	5.9	ERA	Interim Guideline; Threshold Effect Level; Dry weight	Canada	Environment Canada, 1999
	FW	5.9	ERA	Interim Guideline; Threshold Effect Level; Dry weight	Nova Scotia	NSDOE, 1998
	FW	6	SLCA	Guideline; Lowest Effect Level; Dry weight	Ontario	Persaud et al., 1993
	FW/SW	6	SLCA	Criterion; Lowest Effect Level; Dry weight	New York State	NYSDEC, 1994
	FW	6.9	SQG	Sediment Quality Guideline; NRCC 85th percentile level in streams; Dry weight	Texas	TNRCC, 1996
	FW/SW	7	SLCA	Sediment Quality Criterion; Minimal Effect Threshold; Dry weight	St. Lawrence River, Canada	MENVIQ/EC, 1992
	FW	8	SBA	Criterion; MOE Dredged Material Classification; Open water disposal	Ontario	OMOE, 1987
	FW	8	SQG	Heavily Polluted; USEPA Region 5 Harbour Classification; Dry weight	United States	U.S. EPA, 1977
	FW	8.25	EqPA	Chronic EqP Threshold; @ 1% OC; Dry weight	United States	Bolton et al., 1985
	FW	10	SBA	WDNR Criterion; Interim for In-water Disposal of Dredged Sediments	Wisconsin	Sullivan et al., 1985
	FW	11	ELA	Threshold Effect Level for Hyalella azteca 28-d test; Dry weight	United States	Ingersoll et al., 1996
	FW	13	ERA	Effect Range Low for Hyalella azteca 28-d test; Dry weight	United States	Ingersoll et al., 1996
	FW	15.7	SBA	Texas Water Commission Screening Levels	United States	Davis, 1987
	FW	16.6	EqPA	Ecotoxicological value; Dry weight; @ 1% OC	Netherlands	Stortelder et al., 1989
	FW	17	ERA	Probable Effect Level; Dry weight	Canada	Environment Canada, 1999
	FW	17	ERA	Probable Effect Level; Dry weight	Nova Scotia	NSDOE, 1998
	FW/SW	17	SLCA	Sediment Quality Criterion; Toxic Effect Threshold; Dry weight; @ 1% OC	St. Lawrence River, Canada	MENVIQ/EC, 1992
	FW	17.6	SQG	Sediment Quality Guideline; NRCC 85th percentile level in reservoirs; Dry weight	Texas	TNRCC, 1996

Table L.1. A summary of the available sediment quality criteria and guidelines for the protection of aquatic life (cont.)

Chemical name	Water type	Guideline (mg/kg)	Approach	Application	Jurisdiction	Reference
	FW	19	PAETA	Sediment Quality Value; Hyalella; Dry weight	Washington	Cubbage et al., 1997
	FW	19	PAETA	Sediment Quality Value; Microtox; Dry weight	Washington	Cubbage et al., 1997
	FW/SW	20	SQG	Sediment Quality Guideline; Low level; Dry weight	Australia and New Zealand	ANZECC, 1998
	FW	33	SLCA	Guideline; Severe Effect Level; Dry weight	Ontario	Persaud et al., 1993
	FW/SW	33	SLCA	Criterion; Severe Effect Level; Dry weight	New York State	NYSDEC, 1994
	FW	35		Recommended Target for classification (see reference) of freshwater and dredged sediments; Not polluted	Netherlands	NIPHEP, 1989
	FW	40	AETA	Sediment Quality Value; Microtox; Dry weight	Washington	Cubbage et al., 1997
	FW/SW	40		Interim Sediment Quality Guidelines; Potential adverse effects on water quality	Oregon	ODEQ, 1989a
	FW	45		Recommended Directive for classification (see reference) of freshwater and dredged sediments; Slightly polluted	Netherlands	NIPHEP, 1989
	FW	48	ELA	Probable Effect Level for Hyalella azteca 28-d test; Dry weight	United States	Ingersoll et al., 1996
	FW	50	ERA	Effect Range Median for Hyalella azteca 28-d test; Dry weight	United States	Ingersoll et al., 1996
	FW/SW	70	SQG	Sediment Quality Guideline; High level; Dry weight	Australia and New Zealand	ANZECC, 1998
	FW	100	AETA	No Effect Concentration for Hyalella azteca 28-d test; Dry weight	United States	Ingersoll et al., 1996
	FW	100		Recommended Limit for classification (see reference) of freshwater and dredged sediments; Polluted	Netherlands	NIPHEP, 1989
	FW	150	AETA	Sediment Quality Value; Hyalella; Dry weight	Washington	Cubbage et al., 1997
Arsenic total	FW	5.9		Working Sediment Quality Guideline; Interim; Dry weight	British Columbia	Nagpal et al., 1998
	FW	6	SLCA	Criterion; Lowest Effect Level	British Columbia	Nagpal et al., 1995
	FW	17		Working Guideline; Probable Effect Level; Dry weight	British Columbia	Nagpal et al., 1998
	FW	33	SLCA	Criterion; Severe Effect Level	British Columbia	Nagpal et al., 1995
Chromium	FW	6.25	EqPA	Chronic EqP Threshold; @ 1% OC; Dry weight	United States	Bolton et al., 1985
	FW	20	SQG	Sediment Quality Guideline; NRCC 85th percentile level in streams; Dry weight	Texas	TNRCC, 1996
	FW/SW	20-300		Interim Sediment Quality Guidelines; Potential adverse effects on water quality	Oregon	ODEQ, 1989a

Table L.1. A summary of the available sediment quality criteria and guidelines for the protection of aquatic life (cont.)

Chemical name	Water type	Guideline (mg/kg)	Approach	Application	Jurisdiction	Reference
	FW	25	SBA	Criterion; MOE Dredged Material Classification; Open water disposal	Ontario	OMOE, 1987
	FW	25	SQG	Non-polluted; USEPA Region 5 Harbour Classification; Dry weight	United States	U.S. EPA, 1977
	FW	25-75	SQG	Moderately Polluted; USEPA Region 5 Harbour Classification; Dry weight	United States	U.S. EPA, 1977
	FW	26	SLCA	Guideline; Lowest Effect Level; Dry weight	Ontario	Persaud et al., 1993
	FW/SW	26	SLCA	Criterion; Lowest Effect Level; Dry weight	New York State	NYSDEC, 1994
	FW	34	SQG	Sediment Quality Guideline; NRCC 85th percentile level in reservoirs; Dry weight	Texas	TNRCC, 1996
	FW	36	ELA	Threshold Effect Level for Hyalella azteca 28-d test; Dry weight	United States	Ingersoll et al., 1996
	FW	37.3	ERA	Interim Guideline; Threshold Effect Level; Dry weight	Canada	Environment Canada, 1999
	FW	37.3	ERA	Interim Guideline; Threshold Effect Level; Dry weight	Nova Scotia	NSDOE, 1998
	FW	39	ERA	Effect Range Low for Hyalella azteca 28-d test; Dry weight	United States	Ingersoll et al., 1996
	FW/SW	55	SBA	Sediment Quality Criterion; No Effect Threshold; Dry weight	St. Lawrence River, Canada	MENVIQ/EC, 1992
	FW/SW	55	SLCA	Sediment Quality Criterion; Minimal Effect Threshold; Dry weight	St. Lawrence River, Canada	MENVIQ/EC, 1992
	FW	72.1	SBA	Texas Water Commission Screening Levels	United States	Davis, 1987
	FW	> 75	SQG	Heavily Polluted; USEPA Region 5 Harbour Classification; Dry weight	United States	U.S. EPA, 1977
	FW/SW	80	SQG	Sediment Quality Guideline; Low level; Dry weight	Australia and New Zealand	ANZECC, 1998
	FW	90	ERA	Probable Effect Level; Dry weight	Canada	Environment Canada, 1999
	FW	90	ERA	Probable Effect Level; Dry weight	Nova Scotia	NSDOE, 1998
	FW	95	AETA	No Effect Concentration for Hyalella azteca 28-d test; Dry weight	United States	Ingersoll et al., 1996
	FW	96.6	EqPA	Ecotoxicological value; Dry weight; @ 1% OC	Netherlands	Stortelder et al., 1989
	FW	100	SSBA	Sediment Quality Guideline	Canada	Hart et al., 1988
	FW	100	SBA	WDNR Criterion; Interim for In-water Disposal of Dredged Sediments	Wisconsin	Sullivan et al., 1985
	FW/SW	100	SLCA	Sediment Quality Criterion; Toxic Effect Threshold; Dry weight; @ 1% OC	St. Lawrence River, Canada	MENVIQ/EC, 1992
	FW	100	SBA	USEPA Region VI Proposed Guideline for Sediment Disposal	United States	Pavlou and Weston, 1983
	FW	110	SLCA	Guideline; Severe Effect Level; Dry weight	Ontario	Persaud et al., 1993

Table L.1. A summary of the available sediment quality criteria and guidelines for the protection of aquatic life (cont.)

Chemical name	Water type	Guideline (mg/kg)	Approach	Application	Jurisdiction	Reference
	FW/SW	110	SLCA	Criterion; Severe Effect Level; Dry weight	New York State	NYSDEC, 1994
	FW	115		Recommended Target for classification (see reference) of freshwater and dredged sediments; Not polluted	Netherlands	NIPHEP, 1989
	FW	120	ELA	Probable Effect Level for Hyalella azteca 28-d test	United States	Ingersoll et al., 1996
	FW	155		Recommended Directive for classification (see reference) of freshwater and dredged sediments; Slightly polluted	Netherlands	NIPHEP, 1989
	FW	270	ERA	Effect Range Median for Hyalella azteca 28-d test; Dry weight	United States	Ingersoll et al., 1996
	FW/SW	370	SQG	Sediment Quality Guideline; High level; Dry weight	Australia and New Zealand	ANZECC, 1998
	FW	600		Recommended Limit for classification (see reference) of freshwater and dredged sediments; Polluted	Netherlands	NIPHEP, 1989
Chromium total	FW	26	SLCA	Criterion; Lowest Effect Level	British Columbia	Nagpal et al., 1995
	FW	37		Working Sediment Quality Guideline; Interim; Dry weight	British Columbia	Nagpal et al., 1998
	FW	90		Working Guideline; Probable Effect Level; Dry weight	British Columbia	Nagpal et al., 1998
	FW	110	PAETA	Sediment Quality Value; Hyalella; Dry weight	Washington	Cubbage et al., 1997
	FW	110	SLCA	Criterion; Severe Effect Level	British Columbia	Nagpal et al., 1995
	FW	280	AETA	Sediment Quality Value; Hyalella; Dry weight	Washington	Cubbage et al., 1997
Copper	FW	8.4	EqPA	Ecotoxicological value; Dry weight; @ 1% OC	Netherlands	Stortelder et al., 1989
	FW	16	SLCA	Guideline; Lowest Effect Level; Dry weight	Ontario	Persaud et al., 1993
	FW/SW	16	SLCA	Criterion; Lowest Effect Level; Dry weight	New York State	NYSDEC, 1994
	FW	19.2	SQG	Sediment Quality Guideline; NRCC 85th percentile level in streams; Dry weight	Texas	TNRCC, 1996
	FW	25		Criterion; MOE Dredged Material Classification; Open water disposal	Ontario	OMOE, 1987
	FW	25	SQG	Non-polluted; USEPA Region 5 Harbour Classification; Dry weight	United States	U.S. EPA, 1977
	FW	25-50	SQG	Moderately Polluted; USEPA Region 5 Harbour Classification; Dry weight	United States	U.S. EPA, 1977
	FW	28	ELA	Threshold Effect Level for Hyalella azteca 28-d test; Dry weight	United States	Ingersoll et al., 1996
	FW/SW	28	SBA	Sediment Quality Criterion; No Effect Threshold; Dry weight	St. Lawrence River, Canada	MENVIQ/EC, 1992
	FW/SW	28	SLCA	Sediment Quality Criterion; Minimal Effect Threshold; Dry weight	St. Lawrence River, Canada	MENVIQ/EC, 1992

Table L.1. A summary of the available sediment quality criteria and guidelines for the protection of aquatic life (cont.)

Chemical name	Water type	Guideline (mg/kg)	Approach	Application	Jurisdiction	Reference
	FW	33	SQG	Sediment Quality Guideline; NRCC 85th percentile level in reservoirs; Dry weight	Texas	TNRCC, 1996
	FW	34	EqPA	Chronic EqP Threshold; @ 1% OC; Dry weight	United States	Bolton et al., 1985
	FW	35		Recommended Target for classification (see reference) of freshwater and dredged sediments; Not polluted	Netherlands	NIPHEP, 1989
	FW	35.7	ERA	Interim Guideline; Threshold Effect Level; Dry weight	Canada	Environment Canada, 1999
	FW	35.7	ERA	Interim Guideline; Threshold Effect Level; Dry weight	Nova Scotia	NSDOE, 1998
	FW	40	SBA	Texas Water Commission Screening Levels	United States	Davis, 1987
	FW	41	ERA	Effect Range Low for Hyalella azteca 28-d test; Dry weight	United States	Ingersoll et al., 1996
	FW	50		Interim guideline; Potential adverse effects on water quality; Dry weight	Oregon	ODEQ, 1989b
	FW	50	SBA	USEPA Region VI Proposed Guideline for Sediment Disposal	United States	Pavlou and Weston, 1983
	FW	> 50	SQG	Heavily Polluted; USEPA Region 5 Harbour Classification; Dry weight	United States	U.S. EPA, 1977
	FW/SW	65	SQG	Sediment Quality Guideline; Low level; Dry weight	Australia and New Zealand	ANZECC, 1998
	FW	85	SSBA	Sediment Quality Guideline	Canada	Hart et al., 1988
	FW/SW	86	SLCA	Sediment Quality Criterion; Toxic Effect Threshold; Dry weight; @ 1% OC	St. Lawrence River, Canada	MENVIQ/EC, 1992
	FW	90		Recommended Directive for classification (see reference) of freshwater and dredged sediments; Slightly polluted	Netherlands	NIPHEP, 1989
	FW	100	ELA	Probable Effect Level for Hyalella azteca 28-d test; Dry weight	United States	Ingersoll et al., 1996
	FW	100		Interim criteria; For In-water disposal of dredged sediments; Dry weight	Wisconsin	Sullivan et al., 1985
	FW	110	SLCA	Guideline; Severe Effect Level; Dry weight	Ontario	Persaud et al., 1993
	FW/SW	110	SLCA	Criterion; Severe Effect Level; Dry weight	New York State	NYSDEC, 1994
	FW	190	ERA	Effect Range Median for Hyalella azteca 28-d test; Dry weight	United States	Ingersoll et al., 1996
	FW	197	ERA	Probable Effect Level; Dry weight	Canada	Environment Canada, 1999
	FW	197	ERA	Probable Effect Level; Dry weight	Nova Scotia	NSDOE, 1998
	FW/SW	270	SQG	Sediment Quality Guideline; High level; Dry weight	Australia and New Zealand	ANZECC, 1998

Table L.1. A summary of the available sediment quality criteria and guidelines for the protection of aquatic life (cont.)

Chemical name	Water type	Guideline (mg/kg)	Approach	Application	Jurisdiction	Reference
	FW	340	PAETA	Sediment Quality Value; Hyalella; Dry weight	Washington	Cubbage et al., 1997
	FW	400		Recommended Limit for classification (see reference) of freshwater and dredged sediments; Polluted	Netherlands	NIPHEP, 1989
	FW	580	AETA	No Effect Concentration for Hyalella azteca 28-d test; Dry weight	United States	Ingersoll et al., 1996
	FW	840	AETA	Sediment Quality Value; Hyalella; Dry weight	Washington	Cubbage et al., 1997
Copper total	FW	16	SLCA	Criterion; Lowest Effect Level	British Columbia	Nagpal et al., 1995
	FW	36		Working Sediment Quality Guideline; Interim; Dry weight	British Columbia	Nagpal et al., 1998
	FW	110	SLCA	Criterion; Severe Effect Level	British Columbia	Nagpal et al., 1995
	FW	197		Working Guideline; Probable Effect Level; Dry weight	British Columbia	Nagpal et al., 1998
Zinc	FW	50	SBA	Federal Water Quality Administration Criterion for Dredged Material; Maximum Acceptable Concentration	United States	Pavlou and Weston, 1983
	FW	75	SBA	USEPA Region VI Proposed Guideline for Sediment Disposal	United States	Pavlou and Weston, 1983
	FW	90	SBA	Light Pollution; 1968 FWPCA Chicago Guideline	Chicago, Illinois	Pavlou and Weston, 1983
	FW	90	SQG	Non-polluted; USEPA Region 5 Harbour Classification; Dry weight	United States	U.S. EPA, 1977
	FW	90-200	SQG	Moderately Polluted; USEPA Region 5 Harbour Classification; Dry weight	United States	U.S. EPA, 1977
	FW	93	SQG	Sediment Quality Guideline; NRCC 85th percentile level in streams; Dry weight	Texas	TNRCC, 1996
	FW	95.4	EqPA	Ecotoxicological value; Dry weight; @ 1% OC	Netherlands	Stortelder et al., 1989
	FW	98	ELA	Threshold Effect Level for Hyalella azteca 28-d test; Dry weight	United States	Ingersoll et al., 1996
	FW	100	SBA	Criterion; MOE Dredged Material Classification; Open water disposal	Ontario	OMOE, 1987
	FW/SW	100	SBA	Sediment Quality Criterion; No Effect Threshold; Dry weight	St. Lawrence River, Canada	MENVIQ/EC, 1992
	FW	100		Interim criteria; For In-water disposal of dredged sediments; Dry weight	Wisconsin	Sullivan et al., 1985
	FW	110	ERA	Effect Range Low for Hyalella azteca 28-d test; Dry weight	United States	Ingersoll et al., 1996
	FW	120	SLCA	Guideline; Lowest Effect Level; Dry weight	Ontario	Persaud et al., 1993
	FW	120	SBA	Texas Water Commission Screening Levels	United States	Davis, 1987

Table L.1. A summary of the available sediment quality criteria and guidelines for the protection of aquatic life (cont.)

Chemical name	Water type	Guideline (mg/kg)	Approach	Application	Jurisdiction	Reference
	FW	120	SQG	Sediment Quality Guideline; NRCC 85th percentile level	Texas	TNRCC, 1996
	FW/SW	120	SLCA	Criterion; Lowest Effect Level; Dry weight	New York State	NYSDEC, 1994
	FW	123	ERA	Interim Guideline; Threshold Effect Level; Dry weight	Canada	Environment Canada, 1999
	FW	123	ERA	Interim Guideline; Threshold Effect Level; Dry weight	Nova Scotia	NSDOE, 1998
	FW	143	SBA	Sediment Quality Guideline	Canada	Hart et al., 1988
	FW	145	SBA	Moderate Pollution; 1968 FWPCA Chicago Guideline	Chicago, Illinois	Pavlou and Weston, 1983
	FW/SW	150	SLCA	Sediment Quality Criterion; Minimal Effect Threshold; Dry weight	St. Lawrence River, Canada	MENVIQ/EC, 1992
	FW	190	EqPA	Chronic EqP Threshold; @ 1% OC; Dry weight	United States	Bolton et al., 1985
	FW	200	SBA	Heavy Pollution; 1968 FWPCA Chicago Guideline	Chicago, Illinois	Pavlou and Weston, 1983
	FW/SW	200	SQG	Sediment Quality Guideline; Low level; Dry weight	Australia and New Zealand	ANZECC, 1998
	FW	> 200	SQG	Heavily Polluted; USEPA Region 5 Harbour Classification; Dry weight	United States	U.S. EPA, 1977
	FW	250		Interim guideline; Potential adverse effects on water quality; Dry weight	Oregon	ODEQ, 1989b
	FW	300		Recommended Target for classification (see reference) of freshwater and dredged sediments; Not polluted	Netherlands	NIPHEP, 1989
	FW	315	ERA	Probable Effect Level; Dry weight	Canada	Environment Canada, 1999
	FW	315	ERA	Probable Effect Level; Dry weight	Nova Scotia	NSDOE, 1998
	FW/SW	410	SLCA	Criterion; Severe Effect Level; Dry weight	New York State	NYSDEC, 1994
	FW/SW	410	SQG	Sediment Quality Guideline; High level; Dry weight	Australia and New Zealand	ANZECC, 1998
	FW	500	PAETA	Sediment Quality Value; Microtox; Dry weight	Washington	Cubbage et al., 1997
	FW	520	AETA	Sediment Quality Value; Microtox; Dry weight	Washington	Cubbage et al., 1997
	FW	540	ELA	Probable Effect Level for Hyalella azteca 28-d test; Dry weight	United States	Ingersoll et al., 1996
	FW/SW	540	SLCA	Sediment Quality Criterion; Toxic Effect Threshold; Dry weight; @ 1% OC	St. Lawrence River, Canada	MENVIQ/EC, 1992
	FW	550	ERA	Effect Range Median for Hyalella azteca 28-d test; Dry weight	United States	Ingersoll et al., 1996
	FW	820	SLCA	Guideline; Severe Effect Level; Dry weight	Ontario	Persaud et al., 1993
	FW	1,000	PAETA	Sediment Quality Value; Hyalella; Dry weight	Washington	Cubbage et al., 1997

Table L.1. A summary of the available sediment quality criteria and guidelines for the protection of aquatic life (cont.)

Chemical name	Water type	Guideline (mg/kg)	Approach	Application	Jurisdiction	Reference
	FW	1,000		Recommended Directive for classification (see reference) of freshwater and dredged sediments; Slightly polluted	Netherlands	NIPHEP, 1989
	FW	1,300	AETA	No Effect Concentration for Hyalella azteca 28-d test; Dry weight	United States	Ingersoll et al., 1996
	FW	2,500		Recommended Limit for classification (see reference) of freshwater and dredged sediments; Polluted	Netherlands	NIPHEP, 1989
	FW	3,200	AETA	Sediment Quality Value; Hyalella; Dry weight	Washington	Cubbage et al., 1997
Zinc total	FW	120	SLCA	Criterion; Lowest Effect Level	British Columbia	Nagpal et al., 1995
	FW	123		Working Sediment Quality Guideline; Interim; Dry weight	British Columbia	Nagpal et al., 1998
	FW	315		Working Guideline; Probable Effect Level; Dry weight	British Columbia	Nagpal et al., 1998
	FW	820	SLCA	Criterion; Severe Effect Level	British Columbia	Nagpal et al., 1995
Marine and estuarine						
Arsenic	SW	5.7	SQG	Sediment Quality Guideline; NRCC 85th percentile level in tidal streams; Dry weight	Texas	TNRCC, 1996
	EST	6.9	SQG	Sediment Quality Guideline; NRCC 85th percentile level in estuaries; Dry weight	Texas	TNRCC, 1996
	EST/SW	7.24	ERA	Interim Guideline; Threshold Effect Level; Dry weight	Nova Scotia	NSDOE, 1998
	EST/SW	7.24	ERA	Interim Guideline; Threshold Effect Level; Dry weight	Canada	Environment Canada, 1999
	EST/SW	7.24	ELA	Guideline; Threshold Effect Level; Dry weight	Florida	MacDonald et al., 1995
	SW	8	SQG	Sediment Quality Guideline; Lower chemical exceedence level; Dry weight	Hong Kong	HKGS, 1998
	SW	8	AETA	Apparent Effect Threshold; Benthic effects; Southern California; Dry weight	California	Becker et al., 1990
	EST/SW	8.2	ERA	Guideline; Effects range – low; Dry weight	United States	Long et al., 1995
	SW	8.2	EqPA	Criterion; Equilibrium Partitioning Based (Chronic); @ 1% OC; Dry weight	United States	Lyman et al., 1987
	SW	8.25	EqPA	Chronic Marine EqP Threshold; @ 1% OC; Dry weight	United States	Bolton et al., 1985
	SW	10	SQG	Sediment Quality Guideline; Low level; Dry weight	New England	NERBC, 1980
	SW	16	EqPA	Criterion; Equilibrium Partitioning Based (Acute); @ 1% OC; Dry weight	United States	Lyman et al., 1987

Table L.1. A summary of the available sediment quality criteria and guidelines for the protection of aquatic life (cont.)

Chemical name	Water type	Guideline (mg/kg)	Approach	Application	Jurisdiction	Reference
	SW	< 20	SQG	Sediment Quality Guideline; Class I (see reference); Dry weight	Norway	Knutzen et al., 1993
	SW	20		Proposed Objective; Burrard Inlet	Burrard Inlet, British Columbia	Nijman and Swain, 1989
	SW	20	SQG	Sediment Quality Guideline; High level; Dry weight	New England	NERBC, 1980
	SW	20	SQG	Sediment Quality Guideline; Class I (see reference); Dry weight	China	Tang et al., 1998
	SW	20	SQG	Sediment Quality Guideline; Screening level for sea disposal; Dry weight	Australia and New Zealand	ANZECC, 1998
	SW	20-80	SQG	Sediment Quality Guideline; Class II (see reference); Dry weight	Norway	Knutzen et al., 1993
	EST/SW	< 33	AETA	Criterion; For Wetlands creation cover, Levee restoration, Landfill daily cover; Dry weight	California	CDWR, 1995
	EST/SW	33	ERA	Effects range – low; Dry weight	United States	Long and Morgan, 1991
	EST/SW	33-85	AETA	Criterion; For Wetlands creation noncover; Dry weight	California	CDWR, 1995
	EST/SW	41.6	ERA	Probable Effect Level; Dry weight	Canada	Environment Canada, 1999
	EST/SW	41.6	ELA	Guideline; Probable Effect Level; Dry weight	Florida	MacDonald et al., 1995
	EST/SW	41.6	ERA	Probable Effect Level; Dry weight	Nova Scotia	NSDOE, 1998
	SW	42	SQG	Sediment Quality Guideline; Upper chemical exceedence level; Dry weight	Hong Kong	HKGS, 1998
	SW	57	AETA	Sediment Quality Standard; No Effect Level; Dry weight	Washington	WSDOE, 1990
	SW	57	AETA	Apparent Effect Threshold; Benthic infauna abundance; Dry weight; 1988 data	Puget Sound, Washington	Barrick et al., 1988a
	SW	≤ 58	SQTA	Guideline; For no, or minimal adverse biological effects; Dry weight	San Francisco Bay	DelValls and Chapman, 1998
	SW	≥ 64	SQTA	Guideline; Results in major adverse biological effects; Dry weight	San Francisco Bay	DelValls and Chapman, 1998
	SW	65	SQG	Sediment Quality Guideline; Class II (see reference); Dry weight	China	Tang et al., 1998
	EST/SW	70	ERA	Guideline; Effects range – median; Dry weight	United States	Long et al., 1995
	EST/SW	70	AETA	Guideline; Puget Sound Screening level for open water disposal; Dry weight	Puget Sound, Washington	Phillips et al., 1988

Table L.1. A summary of the available sediment quality criteria and guidelines for the protection of aquatic life (cont.)

Chemical name	Water type	Guideline (mg/kg)	Approach	Application	Jurisdiction	Reference
	SW	70	SQG	Sediment Quality Guideline; Maximum level for sea disposal; Dry weight	Australia and New Zealand	ANZECC, 1998
	SW	70	AETA	Apparent Effect Threshold; Benthic effects; Northern California; Dry weight	California	Becker et al., 1990
	SW	70	AETA	Apparent Effect Threshold; Benthic effects; All of California; Dry weight	California	Becker et al., 1990
	SW	70	AETA	Apparent Effect Threshold; Bivalve larvae abnormality; Dry weight	California	Becker et al., 1990
	SW	> 72	AETA	Apparent Effect Threshold; Amphipod mortality; All of California; Dry weight	California	Becker et al., 1990
	SW	> 72	AETA	Apparent Effect Threshold; Amphipod mortality; Northern California; Dry weight	California	Becker et al., 1990
	SW	80-400	SQG	Sediment Quality Guideline; Class III (see reference); Dry weight	Norway	Knutzen et al., 1993
	EST/SW	85	ERA	Effects range median; Dry weight	United States	Long and Morgan, 1991
	SW	85	AETA	Apparent Effect Threshold; Benthic infauna abundance; Dry weight; 1986 data	Puget Sound, Washington	Barrick et al., 1988a
	SW	93	AETA	Sediment Impact Zone Maximum Level; Minimum Cleanup Level; Minor Adverse Effect Level; Dry weight	Washington	WSDOE, 1990
	SW	93	AETA	Apparent Effect Threshold; Amphipod toxicity; Dry weight; 1988 data	Puget Sound, Washington	Barrick et al., 1988a
	SW	93	SQG	Sediment Quality Guideline; Class III (see reference); Dry weight	China	Tang et al., 1998
	SW	400-1,000	SQG	Sediment Quality Guideline; Class IV (see reference); Dry weight	Norway	Knutzen et al., 1993
	SW	700	AETA	Apparent Effect Threshold; Microtox bioassay; Dry weight; 1988 data	Puget Sound, Washington	Barrick et al., 1988a
	SW	700	AETA	Apparent Effect Threshold; Oyster larvae toxicity; Dry weight; 1988 data	Puget Sound, Washington	Barrick et al., 1988a
	EST/SW	> 1,000	AETA	Criterion; For Class I (see reference); Dry weight	California	CDWR, 1995
	SW	> 1,000	SQG	Sediment Quality Guideline; Class V (see reference); Dry weight	Norway	Knutzen et al., 1993
Arsenic total	SW	7.2		Working Sediment Quality Guideline; Interim; Dry weight	British Columbia	Nagpal et al., 1998
	SW	33	ERA	Criterion; Effects range low	British Columbia	Nagpal et al., 1995
	SW	42		Working Guideline; Probable Effect Level; Dry weight	British Columbia	Nagpal et al., 1998
	SW	85	ERA	Criterion; Effects range medium	British Columbia	Nagpal et al., 1995

Table L.1. A summary of the available sediment quality criteria and guidelines for the protection of aquatic life (cont.)

Chemical name	Water type	Guideline (mg/kg)	Approach	Application	Jurisdiction	Reference
Chromium	SW	6.25	EqPA	Chronic Marine EqP Threshold; @ 1% OC; Dry weight	United States	Bolton et al., 1985
	EST	29	SQG	Sediment Quality Guideline; NRCC 85th percentile level in estuaries; Dry weight	Texas	TNRCC, 1996
	SW	45	SQG	Sediment Quality Guideline; NRCC 85th percentile level in tidal streams; Dry weight	Texas	TNRCC, 1996
	SW	50	SQG	Sediment Quality Guideline; Class I (see reference); Dry weight	China	Tang et al., 1998
	EST/SW	52.3	ERA	Interim Guideline; Threshold Effect Level; Dry weight	Canada	Environment Canada, 1999
	EST/SW	52.3	ELA	Guideline; Threshold Effect Level; Dry weight	Florida	MacDonald et al., 1995
	EST/SW	52.3	ERA	Interim Guideline; Threshold Effect Level; Dry weight	Nova Scotia	NSDOE, 1998
	SW	60		Proposed Objective; Burrard Inlet	Burrard Inlet, British Columbia	Nijman and Swain, 1989
	SW	< 70	SQG	Sediment Quality Guideline; Class I (see reference); Dry weight	Norway	Knutzen et al., 1993
	SW	70-300	SQG	Sediment Quality Guideline; Class II (see reference); Dry weight	Norway	Knutzen et al., 1993
	EST/SW	80	ERA	Effects range – low; Dry weight	United States	Long and Morgan, 1991
	SW	80	SQG	Sediment Quality Guideline; Lower chemical exceedence level; Dry weight	Hong Kong	HKGS, 1998
	EST/SW	81	ERA	Guideline; Effects range – low; Dry weight	United States	Long et al., 1995
	SW	81	SQG	Sediment Quality Guideline; Screening level for sea disposal; Dry weight	Australia and New Zealand	ANZECC, 1998
	SW	100	SQG	Sediment Quality Guideline; Low level; Dry weight	New England	NERBC, 1980
	SW	≤ 101.2	SQTA	Guideline; For no, or minimal adverse biological effects; Dry weight	Spain	DelValls and Chapman, 1998
	SW	≤ 110	SQTA	Guideline; For no, or minimal adverse biological effects; Dry weight	San Francisco Bay	DelValls and Chapman, 1998
	SW	120	SQG	Sediment Quality Guideline; Class II (see reference); Dry weight	China	Tang et al., 1998
	SW	≥ 134	SQTA	Guideline; Results in major adverse biological effects; Dry weight	San Francisco Bay	DelValls and Chapman, 1998
	EST/SW	145	ERA	Effects range – median; Dry weight	United States	Long and Morgan, 1991
	EST/SW	160	ERA	Probable Effect Level; Dry weight	Canada	Environment Canada, 1999
	EST/SW	160	ELA	Guideline; Probable Effect Level; Dry weight	Florida	MacDonald et al., 1995

Table L.1. A summary of the available sediment quality criteria and guidelines for the protection of aquatic life (cont.)

Chemical name	Water type	Guideline (mg/kg)	Approach	Application	Jurisdiction	Reference
	EST/SW	160	ERA	Probable Effect Level; Dry weight	Nova Scotia	NSDOE, 1998
	SW	160	SQG	Sediment Quality Guideline; Upper chemical exceedence level; Dry weight	Hong Kong	HKGS, 1998
	SW	> 240	AETA	Apparent Effect Threshold; Bivalve larvae abnormality; Dry weight	California	Becker et al., 1990
	SW	> 240	AETA	Apparent Effect Threshold; Amphipod mortality; Northern California; Dry weight	California	Becker et al., 1990
	SW	> 240	AETA	Apparent Effect Threshold; Benthic effects; Northern California; Dry weight	California	Becker et al., 1990
	SW	260	AETA	Sediment Quality Standard; No Effect Level; Dry weight	Washington	WSDOE, 1990
	SW	270	AETA	Sediment Impact Zone Maximum Level; Minimum Cleanup Level; Minor Adverse Effect Level; Dry weight	Washington	WSDOE, 1990
	SW	270	SQG	Sediment Quality Guideline; Class III (see reference); Dry weight	China	Tang et al., 1998
	SW	$\geq$ 283.9	SQTA	Guideline; Results in major adverse biological effects; Dry weight	Spain	DelValls and Chapman, 1998
	SW	300	SQG	Sediment Quality Guideline; High level; Dry weight	New England	NERBC, 1980
	SW	300-1,500	SQG	Sediment Quality Guideline; Class III (see reference); Dry weight	Norway	Knutzen et al., 1993
	SW	310	AETA	Apparent Effect Threshold; Benthic effects; All of California; Dry weight	California	Becker et al., 1990
	SW	310	AETA	Apparent Effect Threshold; Benthic effects; Southern California; Dry weight	California	Becker et al., 1990
	FW/SW	370	SQG	Sediment Quality Guideline; High level; Dry weight	Australia and New Zealand	ANZECC, 1998
	SW	370	SQG	Sediment Quality Guideline; Maximum level for sea disposal; Dry weight	Australia and New Zealand	ANZECC, 1998
	SW	> 820	AETA	Apparent Effect Threshold; Amphipod mortality; Southern California; Dry weight	California	Becker et al., 1990
	SW	> 820	AETA	Apparent Effect Threshold; Amphipod mortality; All of California; Dry weight	California	Becker et al., 1990
	SW	1,500-5,000	SQG	Sediment Quality Guideline; Class IV (see reference); Dry weight	Norway	Knutzen et al., 1993
	SW	> 5,000	SQG	Sediment Quality Guideline; Class V (see reference); Dry weight	Norway	Knutzen et al., 1993

Table L.1. A summary of the available sediment quality criteria and guidelines for the protection of aquatic life (cont.)

Chemical name	Water type	Guideline (mg/kg)	Approach	Application	Jurisdiction	Reference
Chromium total	SW	27	AETA	Apparent Effect Threshold; Microtox bioassay; Dry weight; 1988 data	Puget Sound, Washington	Barrick et al., 1988b
	SW	52		Working Sediment Quality Guideline; Interim; Dry weight	British Columbia	Nagpal et al., 1998
	SW	80	ERA	Criterion; Effects range low	British Columbia	Nagpal et al., 1995
	SW	145	ERA	Criterion; Effects range median	British Columbia	Nagpal et al., 1995
	SW	160		Working Guideline; Probable Effect Level; Dry weight	British Columbia	Nagpal et al., 1998
	EST/SW	< 220	AETA	Criterion; For Wetlands creation cover, Levee restoration, Landfill daily cover; Dry weight	California	CDWR, 1995
	EST/SW	220-300	AETA	Criterion; For Wetlands creation noncover; Dry weight	California	CDWR, 1995
	SW	260	AETA	Apparent Effect Threshold; Benthic infauna abundance; Dry weight; 1988 data	Puget Sound, Washington	Barrick et al., 1988a
	SW	270	AETA	Apparent Effect Threshold; Amphipod toxicity; Dry weight; 1988 data	Puget Sound, Washington	Barrick et al., 1988a
	EST/SW	> 1,000	AETA	Criterion; For Class I (see reference); Dry weight	California	CDWR, 1995
	FW/SW	100	SBA	Sediment Quality Criterion; No Effect Threshold; Dry weight	St. Lawrence River, Canada	MENVIQ/EC, 1992
Copper	EST/SW	18.7	ELA	Guideline; Threshold Effect Level; Dry weight	Florida	MacDonald et al., 1995
	EST/SW	18.7	ERA	Interim Guideline; Threshold Effect Level; Dry weight	Canada	Environment Canada, 1999
	EST/SW	18.7	ERA	Interim Guideline; Threshold Effect Level; Dry weight	Nova Scotia	NSDOE, 1998
	EST	24	SQG	Sediment Quality Guideline; NRCC 85th percentile level in estuaries; Dry weight	Texas	TNRCC, 1996
	EST/SW	34	ERA	Guideline; Effects range – low; Dry weight	United States	Long et al., 1995
	SW	34	SQG	Sediment Quality Guideline; Screening level for sea disposal; Dry weight	Australia and New Zealand	ANZECC, 1998
	SW	34	EqPA	Chronic Marine EqP Threshold; @ 1% OC; Dry weight	United States	Bolton et al., 1985
	SW	34	EqPA	Criterion; Equilibrium Partitioning Based (Chronic); @ 1% OC; Dry weight	United States	Lyman et al., 1987
	SW	< 35	SQG	Sediment Quality Guideline; Class I (see reference); Dry weight	Norway	Knutzen et al., 1993
	SW	35	SQG	Sediment Quality Guideline; Class I (see reference); Dry weight	China	Tang et al., 1998
	SW	35-150	SQG	Sediment Quality Guideline; Class II (see reference); Dry weight	Norway	Knutzen et al., 1993

Table L.1. A summary of the available sediment quality criteria and guidelines for the protection of aquatic life (cont.)

Chemical name	Water type	Guideline (mg/kg)	Approach	Application	Jurisdiction	Reference
	SW	38.5	SQG	Sediment Quality Guideline; NRCC 85th percentile level in tidal streams; Dry weight	Texas	TNRCC, 1996
	SW	54	EqPA	Criterion; Equilibrium Partitioning Based (Acute); @ 1% OC; Dry weight	United States	Lyman et al., 1987
	SW	65	SQG	Sediment Quality Guideline; Lower chemical exceedence level; Dry weight	Hong Kong	HKGS, 1998
	SW	66	AETA	Apparent Effect Threshold; Bivalve larvae abnormality; Dry weight	California	Becker et al., 1990
	SW	66	AETA	Apparent Effect Threshold; Benthic effects; Northern California; Dry weight	California	Becker et al., 1990
	SW	≤ 68	SQTA	Guideline; For no, or minimal adverse biological effects; Dry weight	San Francisco Bay	DelValls and Chapman, 1998
	EST/SW	70	ERA	Effects range – low; Dry weight	United States	Long and Morgan, 1991
	EST/SW	81	AETA	Guideline; Puget Sound Screening level for open water disposal; Dry weight	Puget Sound, Washington	Phillips et al., 1988
	EST/SW	< 90	AETA	Criterion; For Wetlands creation cover, Levee restoration, Landfill daily cover; Dry weight	California	CDWR, 1995
	EST/SW	90-390	AETA	Criterion; For Wetlands creation noncover; Dry weight	California	CDWR, 1995
	SW	98	AETA	Apparent Effect Threshold; Amphipod mortality; Northern California; Dry weight	California	Becker et al., 1990
	SW	≥ 98	SQTA	Guideline; Results in major adverse biological effects; Dry weight	San Francisco Bay	DelValls and Chapman, 1998
	SW	100	SQG	Sediment Quality Guideline; Class II (see reference); Dry weight	China	Tang et al., 1998
	SW	100		Proposed Objective; Burrard Inlet	Burrard Inlet, British Columbia	Nijman and Swain, 1989
	EST/SW	108	ERA	Probable Effect Level; Dry weight	Canada	Environment Canada, 1999
	EST/SW	108	ELA	Guideline; Probable Effect Level; Dry weight	Florida	MacDonald et al., 1995
	EST/SW	108	ERA	Probable Effect Level; Dry weight	Nova Scotia	NSDOE, 1998
	SW	110	SQG	Sediment Quality Guideline; Upper chemical exceedence level; Dry weight	Hong Kong	HKGS, 1998

Table L.1. A summary of the available sediment quality criteria and guidelines for the protection of aquatic life (cont.)

Chemical name	Water type	Guideline (mg/kg)	Approach	Application	Jurisdiction	Reference
	SW	150-700	SQG	Sediment Quality Guideline; Class III (see reference); Dry weight	Norway	Knutzen et al., 1993
	SW	200	SQG	Sediment Quality Guideline; Class III (see reference); Dry weight	China	Tang et al., 1998
	SW	200	SQG	Sediment Quality Guideline; Low level; Dry weight	New England	NERBC, 1980
	EST/SW	270	ERA	Guideline; Effects range – median; Dry weight	United States	Long et al., 1995
	SW	270	SQG	Sediment Quality Guideline; Maximum level for sea disposal; Dry weight	Australia and New Zealand	ANZECC, 1998
	SW	310	AETA	Apparent Effect Threshold; Benthic effects; All of California; Dry weight	California	Becker et al., 1990
	SW	310	AETA	Apparent Effect Threshold; Benthic effects; Southern California; Dry weight	California	Becker et al., 1990
	SW	310	AETA	Apparent Effect Threshold; Benthic infauna abundance; Dry weight; 1986 data	Puget Sound, Washington	Barrick et al., 1988a
	EST/SW	390	ERA	Effects range – median; Dry weight	United States	Long and Morgan, 1991
	SW	390	AETA	Sediment Impact Zone Maximum Level; Minimum Cleanup Level; Minor Adverse Effect Level; Dry weight	Washington	WSDOE, 1990
	SW	390	AETA	Sediment Quality Standard; No Effect Level; Dry weight	Washington	WSDOE, 1990
	SW	390	AETA	Apparent Effect Threshold; Oyster larvae toxicity; Dry weight; 1986 data	Puget Sound, Washington	Barrick et al., 1988a
	SW	390	AETA	Apparent Effect Threshold; Oyster larvae toxicity; Dry weight; 1988 data	Puget Sound, Washington	Barrick et al., 1988a
	SW	390	AETA	Apparent Effect Threshold; Microtox bioassay; Dry weight; 1988 data	Puget Sound, Washington	Barrick et al., 1988a
	SW	390	AETA	Apparent Effect Threshold; Microtox bioassay; Dry weight; 1986 data	Puget Sound, Washington	Barrick et al., 1988a
	SW	400	SQG	Sediment Quality Guideline; High level; Dry weight	New England	NERBC, 1980
	SW	530	AETA	Apparent Effect Threshold; Benthic infauna abundance; Dry weight; 1988 data	Puget Sound, Washington	Barrick et al., 1988a
	SW	> 690	AETA	Apparent Effect Threshold; Amphipod mortality; All of California; Dry weight	California	Becker et al., 1990
	SW	> 690	AETA	Apparent Effect Threshold; Amphipod mortality; Southern California; Dry weight	California	Becker et al., 1990

Table L.1. A summary of the available sediment quality criteria and guidelines for the protection of aquatic life (cont.)

Chemical name	Water type	Guideline (mg/kg)	Approach	Application	Jurisdiction	Reference
	SW	700-1,500	SQG	Sediment Quality Guideline; Class IV (see reference); Dry weight	Norway	Knutzen et al., 1993
	SW	810	AETA	Apparent Effect Threshold; Amphipod toxicity; Dry weight; 1986 data	Puget Sound, Washington	Barrick et al., 1988a
	SW	1,300	AETA	Apparent Effect Threshold; Amphipod toxicity; Dry weight; 1988 data	Puget Sound, Washington	Barrick et al., 1988a
	SW	> 1,500	SQG	Sediment Quality Guideline; Class V (see reference); Dry weight	Norway	Knutzen et al., 1993
	EST/SW	> 5,000	AETA	Criterion; For Class I (see reference); Dry weight	California	CDWR, 1995
Copper total	SW	19		Working Sediment Quality Guideline; Interim; Dry weight	British Columbia	Nagpal et al., 1998
	SW	70	ERA	Criterion; Effects range low	British Columbia	Nagpal et al., 1995
	SW	108		Working Guideline; Probable Effect Level; Dry weight	British Columbia	Nagpal et al., 1998
	SW	390	ERA	Criterion; Effects range median	British Columbia	Nagpal et al., 1995
Zinc	EST	110	SQG	Sediment Quality Guideline; NRCC 85th percentile level in estuaries; Dry weight	Texas	TNRCC, 1996
	EST/SW	120	ERA	Effects range – low; Dry weight	United States	Long and Morgan, 1991
	EST/SW	124	ERA	Interim Guideline; Threshold Effect Level; Dry weight	Nova Scotia	NSDOE, 1998
	EST/SW	124	ELA	Guideline; Threshold Effect Level; Dry weight	Florida	MacDonald et al., 1995
	EST/SW	124	ERA	Interim Guideline; Threshold Effect Level; Dry weight	Canada	Environment Canada, 1999
	SW	< 150	SQG	Sediment Quality Guideline; Class I (see reference); Dry weight	Norway	Knutzen et al., 1993
	EST/SW	150	ERA	Guideline; Effects range – low; Dry weight	United States	Long et al., 1995
	SW	150	SQG	Sediment Quality Guideline; Class I (see reference); Dry weight	China	Tang et al., 1998
	SW	150	AETA	Apparent Effect Threshold; Bivalve larvae abnormality; Dry weight	California	Becker et al., 1990
	SW	150	AETA	Apparent Effect Threshold; Benthic effects; Northern California; Dry weight	California	Becker et al., 1990
	SW	150		Proposed Objective; Burrard Inlet	Burrard Inlet, British Columbia	Nijman and Swain, 1989
	SW	150-700	SQG	Sediment Quality Guideline; Class II (see reference); Dry weight	Norway	Knutzen et al., 1993
	SW	$\leq$ 156	SQTA	Guideline; For no, or minimal adverse biological effects; Dry weight	San Francisco Bay	DelValls and Chapman, 1998
	EST/SW	< 160	AETA	Criterion; For Wetlands creation cover, Levee restoration, Landfill daily cover; Dry weight	California	CDWR, 1995

Table L.1. A summary of the available sediment quality criteria and guidelines for the protection of aquatic life (cont.)

Chemical name	Water type	Guideline (mg/kg)	Approach	Application	Jurisdiction	Reference
	EST/SW	160	AETA	Guideline; Puget Sound Screening level for open water disposal; Dry weight	Puget Sound, Washington	Phillips et al., 1988
	EST/SW	160-270	AETA	Criterion; For Wetlands creation noncover; Dry weight	California	CDWR, 1995
	SW	190	EqPA	Criterion; Equilibrium Partitioning Based (Chronic); @ 1% OC; Dry weight	United States	Lyman et al., 1987
	SW	190	EqPA	Chronic Marine EqP Threshold; @ 1% OC; Dry weight	United States	Bolton et al., 1985
	SW	191	SQG	Sediment Quality Guideline; NRCC 85th percentile level in tidal streams; Dry weight	Texas	TNRCC, 1996
	SW	200	SQG	Sediment Quality Guideline; Lower chemical exceedence level; Dry weight	Hong Kong	HKGS, 1998
	SW	200	SQG	Sediment Quality Guideline; Low level; Dry weight	New England	NERBC, 1980
	SW	200	SQG	Sediment Quality Guideline; Screening level for sea disposal; Dry weight	Australia and New Zealand	ANZECC, 1998
	SW	≥ 225	SQTA	Guideline; Results in major adverse biological effects; Dry weight	San Francisco Bay	DelValls and Chapman, 1998
	SW	230	AETA	Apparent Effect Threshold; Amphipod mortality; Northern California; Dry weight	California	Becker et al., 1990
	SW	260	AETA	Apparent Effect Threshold; Benthic infauna abundance; Dry weight; 1986 data	Puget Sound, Washington	Barrick et al., 1988a
	EST/SW	270	ERA	Effects range – median; Dry weight	United States	Long and Morgan, 1991
	SW	270	SQG	Sediment Quality Guideline; Upper chemical exceedence level; Dry weight	Hong Kong	HKGS, 1998
	EST/SW	271	ERA	Probable Effect Level; Dry weight	Canada	Environment Canada, 1999
	EST/SW	271	ERA	Probable Effect Level; Dry weight	Nova Scotia	NSDOE, 1998
	EST/SW	271	ELA	Guideline; Probable Effect Level; Dry weight	Florida	MacDonald et al., 1995
	SW	340	AETA	Apparent Effect Threshold; Benthic effects; All of California; Dry weight	California	Becker et al., 1990
	SW	340	AETA	Apparent Effect Threshold; Benthic effects; Southern California; Dry weight	California	Becker et al., 1990

Table L.1. A summary of the available sediment quality criteria and guidelines for the protection of aquatic life (cont.)

Chemical name	Water type	Guideline (mg/kg)	Approach	Application	Jurisdiction	Reference
	SW	350	SQG	Sediment Quality Guideline; Class II (see reference); Dry weight	China	Tang et al., 1998
	SW	400	SQG	Sediment Quality Guideline; High level; Dry weight	New England	NERBC, 1980
	EST/SW	410	ERA	Guideline; Effects range – median; Dry weight	United States	Long et al., 1995
	SW	410	SQG	Sediment Quality Guideline; Maximum level for sea disposal; Dry weight	Australia and New Zealand	ANZECC, 1998
	SW	410	AETA	Sediment Quality Standard; No Effect Level; Dry weight	Washington	WSDOE, 1990
	SW	410	AETA	Apparent Effect Threshold; Benthic infauna abundance; Dry weight; 1988 data	Puget Sound, Washington	Barrick et al., 1988a
	SW	560	EqPA	Criterion; Equilibrium Partitioning Based (Acute); @ 1% OC; Dry weight	United States	Lyman et al., 1987
	SW	600	SQG	Sediment Quality Guideline; Class III (see reference); Dry weight	China	Tang et al., 1998
	SW	700-3,000	SQG	Sediment Quality Guideline; Class III (see reference); Dry weight	Norway	Knutzen et al., 1993
	SW	870	AETA	Apparent Effect Threshold; Amphipod toxicity; Dry weight; 1986 data	Puget Sound, Washington	Barrick et al., 1988a
	SW	> 870	AETA	Apparent Effect Threshold; Amphipod mortality; All of California; Dry weight	California	Becker et al., 1990
	SW	> 870	AETA	Apparent Effect Threshold; Amphipod mortality; Southern California; Dry weight	California	Becker et al., 1990
	SW	960	AETA	Sediment Impact Zone Maximum Level; Minimum Cleanup Level; Minor Adverse Effect Level; Dry weight	Washington	WSDOE, 1990
	SW	960	AETA	Apparent Effect Threshold; Amphipod toxicity; Dry weight; 1988 data	Puget Sound, Washington	Barrick et al., 1988a
	SW	1,600	AETA	Apparent Effect Threshold; Microtox bioassay; Dry weight; 1988 data	Puget Sound, Washington	Barrick et al., 1988a
	SW	1,600	AETA	Apparent Effect Threshold; Oyster larvae toxicity; Dry weight; 1986 data	Puget Sound, Washington	Barrick et al., 1988a
	SW	1,600	AETA	Apparent Effect Threshold; Oyster larvae toxicity; Dry weight; 1988 data	Puget Sound, Washington	Barrick et al., 1988a
	SW	1,600	AETA	Apparent Effect Threshold; Microtox bioassay; Dry weight; 1986 data	Puget Sound, Washington	Barrick et al., 1988a
	SW	3,000-10,000	SQG	Sediment Quality Guideline; Class IV (see reference); Dry weight	Norway	Knutzen et al., 1993

Table L.1. A summary of the available sediment quality criteria and guidelines for the protection of aquatic life (cont.)

Chemical name	Water type	Guideline (mg/kg)	Approach	Application	Jurisdiction	Reference
	EST/SW	> 10,000	AETA	Criterion; For Class I (see reference); Dry weight	California	CDWR, 1995
	SW	>10,000	SQG	Sediment Quality Guideline; Class V (see reference); Dry weight	Norway	Knutzen et al., 1993
Zinc total	SW	120	ERA	Criterion; Effects range low	British Columbia	Nagpal et al., 1995
	SW	124		Working Sediment Quality Guideline; Interim; Dry weight	British Columbia	Nagpal et al., 1998
	SW	270	ERA	Criterion; Effects range median	British Columbia	Nagpal et al., 1995
	SW	271		Working Guideline; Probable Effect Level; Dry weight	British Columbia	Nagpal et al., 1998

STRATUS CONSULTING

1881 Ninth Street, Suite 201 **Boulder, Colorado 80302** **phone 303.381.8000** **fax 303.381.8200** (headquarters)

1920 L Street, N.W., Suite 420 **Washington, D.C. 20036** **phone 202.466.3731** **fax 202.466.3732**

www.stratusconsulting.com