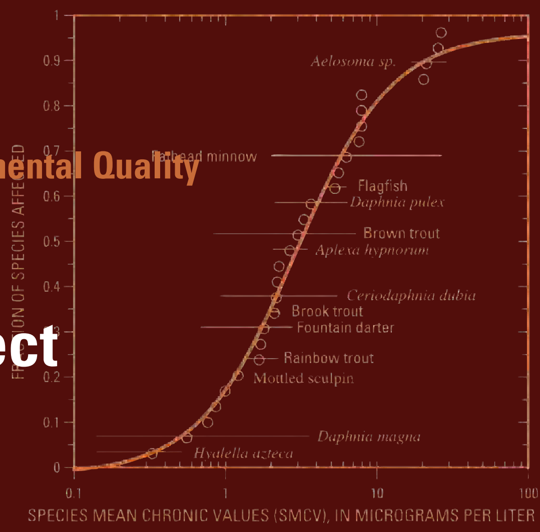
Prepared in cooperation with the Idaho Department of Environmental Quality

Cadmium Risks to Freshwater Life: Derivation and Validation of Low-Effect Criteria Values using Laboratory and Field Studies



Scientific Investigations Report 2006–5245

U.S. Department of the Interior
U.S. Geological Survey

Cover: The montage illustrates an analytical approach and photographs of species that were important in this report as follows:

Graphic: A species-sensitivity distribution (SSD) of freshwater species to chronic cadmium exposures is illustrated. The SSD concept and analyses were fundamental to this report. The open circles and horizontal lines represent the means and ranges of available chronic effects thresholds to cadmium for different species, respectively. Selected species are labeled.

Photographs:

Left: The amphipod *Hyalella azteca*; photograph by Scott Bauer, U.S. Department of Agriculture, Agricultural Research Service.

Top right: A westslope cutthroat trout, *Oncorhynchus clarki lewisi*; photograph by Ernest R. Keeley, Idaho State University.

Middle right: A mottled sculpin, *Cottus bairdi*, from captive broodstock that was used for toxicity testing; photograph by Doug Hardesty, U.S. Geological Survey, Columbia Ecological Research Center, Columbia, Missouri.

Bottom right: A mottled sculpin in a natural setting; photograph by Ernest R. Keeley, Idaho State University.

Cadmium Risks to Freshwater Life: Derivation and Validation of Low-Effect Criteria Values using Laboratory and Field Studies

By Christopher A. Mebane

Prepared in cooperation with the Idaho Department of Environmental Quality

Scientific Investigations Report 2006–5245

U.S. Department of the Interior
U.S. Geological Survey

U.S. Department of the Interior
DIRK KEMPTHORNE, Secretary

U.S. Geological Survey
Mark D. Myers Director

U.S. Geological Survey, Reston, Virginia: 2006

For product and ordering information:
World Wide Web: http://www.usgs.gov/pubprod
Telephone: 1-888-ASK-USGS

For more information on the USGS—the Federal source for science about the Earth, its natural and living resources, natural hazards, and the environment:
World Wide Web: http://www.usgs.gov
Telephone: 1-888-ASK-USGS

Suggested citation:
Mebane, C.A., 2006, Cadmium risks to freshwater life: Derivation and validation of low-effect criteria values using laboratory and field studies: U.S. Geological Survey Scientific Investigations Report 2006-5245, 130 p.

Acknowledgments

Many people generously shared their time and insights to provide data, explanations, or helpful criticisms. Reviewers included Harry Ohlendorf, CH2M Hill, Sacramento, Calif.; Steven Canton, Chadwick Ecological Consultants, Littleton, Colo.; Charles Stephan, U.S. Environmental Protection Agency (EPA) Environmental Research Laboratory, Duluth, Minn.; Don Essig and Joanna Sandow, Idaho Department of Environmental Quality, Boise, Idaho; Christopher Ingersoll, U.S. Geological Survey, Columbia Ecological Research Center, Columbia, Mo.; and Daren Carlisle, U.S. Geological Survey National Water Quality Assessment, Reston, Va.

Many investigators generously took time to answer questions, provide additional details of their work, share data, or suggest other information sources. These included John Besser, Nile Kemble, and Stacy James, U.S. Geological Survey, Columbia Ecological Research Center, Columbia, Mo.; Stephen Brinkman, Colorado Division of Wildlife, Fort Collins, Colo.; Uwe Borgmann, National Water Research Institute, Burlington, Ontario, Canada; Victor Castillo III, Southwest Texas State University, San Marcos, Tex.; Michaël Coeurdassier, University of Franche-Comte, Besançon, France; Valery Forbes, Roskilde University, Roskilde, Denmark; Clyde Goulden, National Academy of Science Philadelphia; Duane Gossiaux, National Oceanic and Atmospheric Administration, Ann Arbor, Mich.; Lydia Hollis, Golder Associates, Calgary; David Mount, U.S. Environmental Protection Agency, Environmental Research Laboratory, Duluth, Minn.; David Nimick, U.S. Geological Survey, Helena, Mont.; Vince Palace and Richard McNicol, Freshwater Research Institute, Winnipeg; David Riddell, University of New Brunswick, Fredericton, New Brunswick, Canada; Andrew Wigginton, University of Kentucky, Lexington, Ky.; and Chris Wood, McMaster University, Waterloo, Ontario, Canada. Russell Erickson, U.S. Environmental Protection Agency, Environmental Research Laboratory, Duluth, Minn., gave statistical advice; Anne Kuhn and Glenn Thursby, U.S. Environmental Protection Agency, Environmental Research Laboratory, Narragansett, R.I., and Webb Van Winkle, Oak Ridge National Laboratory (retired), Boise, Idaho provided advice on population modeling. The Idaho Department of Environmental Quality's Surface Water Program provided principal funding for this project. Don E. Essig was the project manager for the Idaho Department of Environmental Quality.

This page left intentionally blank

Contents

Contents—Continued

Figures

Tables

Conversion Factors

[Emphasizing conversions that were used to compile the tables in this report using comparable units of concentration. Good sources for additional characteristics of water include Hem (1992) and Hounslow (1995). Atomic weights are from Weast (1990)]

Multiply	By	To obtain
microgram per kilogram (µg/kg)	0.001	milligram per kilogram
millequivalents calcium	20.04	milligrams per liter calcium
milliequivalents magnesium	12.15	milligrams per liter magnesium
milligram per kilogram (mg/kg)	1,000	microgram per kilogram
millimoles calcium (mM)	40.078	milligrams per liter calcium
millimoles magnesium (mM)	24.305	milligrams per liter magnesium
micromoles cadmium (µM)	112.411	microgram per liter
nanomoles cadmium (nM)	0.112411	micrograms per liter
milligrams per liter (mg/L)	1,000	micrograms per liter
nanograms per liter (ng/L)	0.001	micrograms per liter
water hardness as milliequivalents (meq)	50.05	hardness as calcium carbonate ($CaCO_3$) in milligrams per liter
water hardness as millimoles (mM)	100.09	Hardness as calcium carbonate ($CaCO_3$) in milligrams per liter
gram (g)	0.03527	ounce, avoirdupois
square meter (m^2)	0.0002471	acre

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:

$$°F=(1.8\times°C)+32$$

Water hardness as milligrams per liter (mg/L) calcium carbonate may be derived from concentrations of calcium (Ca) and magnesium (Mg) as follows:

Hardness (as $CaCO_3$) in mg/L) = 2.497 x (Ca in mg/L) + 4.118 x (Mg in mg/L).

Concentrations of chemical constituents in water are given either in milligrams per liter (mg/L) or micrograms per liter (µg/L).

Cadmium Risks to Freshwater Life: Derivation and Validation of Low-Effect Criteria Values using Laboratory and Field Studies

By Christopher A. Mebane

Abstract

In 2001, the U.S. Environmental Protection Agency (EPA) released updated aquatic life criteria for cadmium. Since then, additional data on the effects of cadmium to aquatic life have become available from studies supported by the EPA, Idaho Department of Environmental Quality (IDEQ), and the U.S. Geological Survey, among other sources. Updated data on the effects of cadmium to aquatic life were compiled and reviewed and low-effect concentrations were estimated. Low-effect values were calculated using EPA's guidelines for deriving numerical national water-quality criteria for the protection of aquatic organisms and their uses. Data on the short-term (acute) effects of cadmium on North American freshwater species that were suitable for criteria derivation were located for 69 species representing 58 genera and 33 families. For longer-term (chronic) effects of cadmium on North American freshwater species, suitable data were located for 28 species representing 21 genera and 17 families. Both the acute and chronic toxicity of cadmium were dependent on the hardness of the test water. Hardness-toxicity regressions were developed for both acute and chronic datasets so that effects data from different tests could be adjusted to a common water hardness. Hardness-adjusted effects values were pooled to obtain species and genus mean acute and chronic values, which then were ranked by their sensitivity to cadmium. The four most sensitive genera to acute exposures were, in order of increasing cadmium resistance, *Oncorhynchus* (Pacific trout and salmon), *Salvelinus* ("char" trout), *Salmo* (Atlantic trout and salmon), and *Cottus* (sculpin). The four most sensitive genera to chronic exposures were *Hyalella* (amphipod), *Gammarus* (amphipod), *Cottus*, and *Salvelinus*. Using the updated datasets, hardness dependent criteria equations were calculated for acute and chronic exposures to cadmium. At a hardness of 50 mg/L as calcium carbonate, the criterion maximum concentration (CMC, or "acute" criterion) was calculated as 0.75 µg/L cadmium using the hardness-dependent equation CMC = $e^{(0.83675 \times \ln(\text{hardness}) - 3.5602)}$ where the "ln hardness" is the natural logarithm of the water hardness. Likewise, the criterion continuous concentration (CCC, or "chronic" criterion) was calculated as 0.38 µg/L cadmium using the hardness-dependent equation CCC = $(e^{(0.6247 \times \ln(\text{hardness}) - 3.344)}) \times (1.101672 - ((\ln \text{hardness}) \times 0.041838)))$.

Using data that were independent of those used to derive the criteria, the criteria concentrations were evaluated to estimate whether adverse effects were expected to the biological integrity of natural waters or to selected species listed as threatened or endangered. One species was identified that would not be fully protected by the derived CCC, the amphipod *Hyalella azteca*. Exposure to CCC conditions likely would lead to population decreases in *Hyalella azteca*, the food web consequences of which probably would be slight if macroinvertebrate communities were otherwise diverse. Some data also suggested adverse behavioral changes are possible in fish following long-term exposures to low levels of cadmium, particularly in char (genus *Salvelinus*). Although ambiguous, these data indicate a need to periodically review the literature on behavioral changes in fish following metals exposure as more information becomes available. Most data reviewed indicated that criteria conditions were unlikely to contribute to overt adverse effects to either biological integrity or listed species. If elevated cadmium concentrations that approach the chronic criterion values occur in ambient waters, careful biological monitoring of invertebrate and fish assemblages would be prudent to validate the prediction that the assemblages would not be adversely affected by cadmium at criterion concentrations.

Introduction

In 2001, the U.S. Environmental Protection Agency (EPA) published an update of aquatic life criteria for cadmium. Since the publication of the U.S. Environmental Protection Agency's (2001) update, additional data on the effects of cadmium on aquatic life have been generated, including data from studies that focused on Idaho species or conditions. These include studies conducted or sponsored by EPA, Idaho Department of Environmental Quality (IDEQ), U.S. Geological Survey (USGS), and an Idaho discharger, the Thompson Creek Mining Company (Mebane, 2003; Chadwick Ecological Consultants, Inc., 2004a; Besser and others, 2006). In cooperation with the IDEQ, the USGS compiled and reviewed updated data on the effects of cadmium to aquatic life and estimated low-effect values. The main purposes of this report are (1) to compile updated datasets on the effects of cadmium to aquatic organisms, and to synthesize the compilation using EPA's procedures for deriving numerical national criteria for the protection of aquatic life (Stephan and others, 1985); and (2) to evaluate whether the calculated criteria would likely protect aquatic communities under realistic field conditions. The second purpose emphasizes species and conditions that are relevant to Idaho

Section 304(a) of the Clean Water Act (CWA, 33 U.S. Code §1314(a)) requires that EPA publish, and from time to time revise, water-quality criteria that accurately reflect the latest scientific knowledge on the fate and effects of pollutants in water bodies. Section 304(a) of the Clean Water Act listed numerous specific areas of scientific knowledge to be addressed by water-quality criteria for protecting aquatic life and human uses of waters. Topics specified by the statute pertaining to aquatic life criteria (as opposed to human health criteria) include (1) identifiable effects to plankton, fish, shellfish, wildlife, and plant life; (2) concentration and dispersal of pollutants through biological, physical, and chemical processes; and (3) the effects of pollutants on biological community diversity, productivity, and stability for varying types of receiving waters. Federal water-quality standards regulations further provide that States adopt numeric water-quality criteria based on either EPA's criteria, EPA's criteria modified to reflect site-specific conditions, or other scientifically defensible methods (40 CFR 131.11).

As a "scientifically defensible method of analysis," this analysis largely relies on EPA's 1985 guidelines for developing numerical National aquatic-life criteria. EPA's guidelines ("the guidelines") were developed to provide an objective, internally consistent, appropriate, and feasible way of deriving National criteria (Stephan and others, 1985). Although the EPA distinguishes between regulatory "water-quality standards" and non-regulatory "aquatic-life criteria" (U.S. Environmental Protection Agency, 1994), the words "criterion" and "criteria" often imply regulatory limits in other contexts. As used here, an acute or a chronic criterion is a concentration and exposure duration corresponding to a specified risk probability and does not imply a regulatory limit. Synonyms to "criteria" in this context might include low-effect concentrations, benchmarks, guidelines, targets, thresholds, or risk values.

Cadmium in Ambient Waters

Cadmium, like other heavy metals, generally is rare in unpolluted natural waters. The term "heavy metals" refers to metallic elements with an atomic number greater than about 20, the atomic number of calcium. Cadmium often co-occurs with copper and zinc. Cadmium, copper, and zinc concentrations in natural waters apparently are roughly proportional to their relative abundance in rocks. In granite, typical cadmium concentrations are about 150 to 400 times less than copper or zinc concentrations, respectively (about 0.13, 20, and 50 mg/kg, respectively for cadmium, copper, and zinc). This rank order between the three metals' concentrations, and the approximate two orders of magnitude difference between cadmium and copper or zinc concentrations also was reported for basalt, shale, and limestone (Drever, 1995). In natural waters, typical cadmium concentrations are about 10–50 times less than copper or zinc concentrations (Hem, 1992; Stephan and others, 1994a; Drever, 1995).

Concentrations of trace metals are often reported as total or dissolved concentrations. "Total" or "total recoverable" concentrations usually refer to values measured from raw, unfiltered wholewater samples. "Dissolved" concentrations usually refer to values measured from water samples that have been physically filtered to separate particulate and aqueous fractions. For filtered trace-metal samples, the standard procedure used by the EPA and USGS is to filter wholewater samples through a 0.45-micrometer (µm) filter (U.S. Environmental Protection Agency, 1994; Wilde and others, 2004). In this report, when filtered (dissolved) values are available, they are emphasized because of the belief that dissolved metal more closely approximates the bioavailable fraction of metal in the water column than does total metal (Prothro, 1993). As a practical matter, with cadmium this distinction is less important than for some other trace metals such as copper or lead. Because cadmium is highly soluble in water, differences between unfiltered (total concentrations) and 0.45-µm filtered samples are usually small, with dissolved cadmium concentrations averaging about 90 to 95 percent those of unfiltered "total" samples (Stephan, 1995; Clark, 2002).

Cadmium concentration data collected from natural waters may be biased high in some older publications. As of 1971, the median natural background concentration of cadmium in waters of the United States was estimated at about 1 µg/L (Hem, 1992). In contrast, Stephan and others (1994b) estimated that typical background dissolved cadmium concentrations in freshwaters of the United States ranged from 0.002 to 0.08 µg/L. The difference in these estimates does not indicate that over two decades ambient cadmium concentrations decreased by one to two orders of magnitude, but rather reduced field and laboratory sample contamination through the use of "clean" techniques (Stephan and others, 1994a). Even in locations in North America that because of unusually elevated cadmium concentrations were the subject of several investigations, concentrations in streams and lakes still seldom reach more than about 15 µg/L. These locations include the vicinities of Rouyn-Noranda, Quebec and Sudbury, Ontario, Canada, and Coeur d'Alene, Idaho (Clark, 2002; Couture and Kumar, 2003; this report). In Idaho, data collected using "clean" field and laboratory techniques indicate that background concentrations of dissolved cadmium range from <0.02 to 0.1 µg/L (see section "Criteria in Variable Environments"), consistent with Stephan and others (1994a) estimate.

Cadmium concentrations can become elevated in waters that are influenced by sources such as mining, minerals processing, and combustion of fossil fuel. Cadmium is present in zinc ore minerals such as sphalerite and is recovered from some copper ores during smelting and refining. Cadmium has a tendency to enter the atmosphere through vaporization at high temperatures; therefore, cadmium may be released to the environment in metallurgical processes and fossil fuel combustion. Cadmium is used for electroplating, for pigments in paint, printing ink, and plastics, as a stabilizer for PVC plastics, and in electrical batteries. Many of these uses tend to make the element available to water that comes into contact with wastes (Hem, 1992). Cadmium also may be elevated in wastewater discharges due to corrosion of galvanized pipes in public water-supply systems (R. Finch, City of Boise, Idaho, oral commun., 2005).

Although rare in surface waters, cadmium is highly toxic to some aquatic life. In comparative acute toxicity testing of all 63 atomically stable heavy metals in the periodic table, cadmium clearly was the most toxic metal (Borgmann and others, 2005). Recent EPA recommendations for regulatory criteria to protect aquatic life from cadmium toxicity are not much higher than the upper range estimates of background cadmium concentrations. U.S. Environmental Protection Agency (2001) recommended that the highest 4-day average dissolved cadmium concentration not exceed 0.08 to 0.25 µg/L at water hardnesses of 20 and 100 mg/L as calcium carbonate, respectively.

Methods

This analysis involved two distinct parts. First, datasets of acute (short-term) and chronic (long-term) responses of aquatic organisms to cadmium were compiled and synthesized following EPA guidelines (Stephan and others, 1985). Second, the resulting criteria, field survey results, and ecosystem experiments were compared to evaluate the protectiveness of criteria in realistic settings. Criteria also were compared with estimated thresholds of minimal effects to especially vulnerable species listed as threatened or endangered under the Endangered Species Act.

Methods for Criteria Derivation

This analysis followed the general approach described in EPA's National guidelines for deriving numerical water-quality criteria (Stephan and others, 1985). Major steps in the approach and deviations from the guidelines are briefly described here, with further details described in the results.

Overview of Criteria Derivation Steps

1. *Data compilation and review*—The guidelines call for collecting all available, relevant data on the toxicity to, and bioaccumulation by, aquatic animals and plants, Food and Drug Administration Action levels, and feeding studies or field studies of wildlife that regularly consume aquatic organisms. Literature was searched for relevant data through electronic databases, reviewing contents of selected key journals, and contacts with other investigators to locate unpublished data. Four primary electronic databases were searched. The Web of Science®—Science Citation Index, the Aquatic Sciences and Fisheries Abstracts, and Water Resources Abstracts (Cambridge Scientific Abstracts, 2005a, 2005b; Thomson Scientific, 2005) were searched using the keywords "cadmium" and "toxicity." EPA's Ecotox database was searched using as search criteria cadmium and any survival, growth, reproduction, or behavioral endpoints (U.S. Environmental Protection Agency, 2002a). Most searches were from January 1, 2000, through April 2005, based on the premise that the bibliography of EPA's 2001 review was sufficiently comprehensive for data published in up to and including 1999. Additionally, because not all authors used the term "toxicity" in their keywords or titles, the following journals were searched for the same period using "cadmium" as a search term in titles, keywords, or abstracts: Aquatic Toxicology, Canadian Journal of Fisheries and Aquatic Sciences, and Environmental Toxicology and Chemistry. With more than 2,000 search results, at least the titles were read to evaluate if the papers might contain useful information. Additionally, a relevant report was obtained and reviewed for additional data (Chadwick Ecological Consultants, Inc., 2004b). EPA's 2001 review concluded that plants were less sensitive to cadmium than aquatic animals. EPA's review did not include dietary effects to wildlife (U.S. Environmental Protection Agency, 2001). Neither plant toxicity or dietary effects of cadmium to wildlife were reviewed in this report.

Data were not used if test methods were questionable or insufficiently described. Although Stephan and others (1985) provided guidelines for screening questionable data, other judgments of data acceptability were postponed until the overall dataset was compiled and analyzed. Guidelines for screening data include whether the tests included a control treatment, mortality in the control treatments was too high, organisms were stressed or previously exposed to substantial concentrations of test materials or other pollutants, use of deionized water or other inappropriate dilution water, whether the test species does not have reproducing populations in the wild in North America, or whether the organisms were exposed to chemicals in a mixture (Stephan and others, 1985).

2. *Develop hardness-toxicity regressions* —Because of repeated observations of a linear relation between the natural logarithms of hardness and acute toxicity values for metals (Stephan and others, 1985; Meyer, 1999), no effort was made to examine other possible relations, such as two-variable models using pH and hardness. Data available over a range of hardness for a species were useful for establishing hardness-toxicity relations if they included a broad range of hardness, if agreement among the species was reasonably good, and if hardness-toxicity slopes between different species were reasonably consistent. The slopes from each species' hardness-toxicity relation were pooled through an analysis of covariance (Zar, 1984).

3. *Determine genus mean acute and chronic values (GMAVs and GMCVs) for all test species at a standard hardness level and rank order the GMAVs and GMCVs by sensitivity*—Acute values were based on short-term exposures (48 to 96 hours) that killed 50 percent of the organisms in the test (LC_{50}). Using the hardness-toxicity relations, each value was adjusted to a common hardness value. If after hardness-normalization, values for a species and values that appear questionable compared with other values within the genus probably should not be used. Stephan and others (1985) gave a factor of 10 difference as an example in which some or all values probably should be rejected. No fixed factor was used for this analysis, rather data were examined species-by-species to judge whether values were so dissimilar to be questionable. For example, for a species with many values (for example rainbow trout with 37 values), a single extreme value would not change the geometric mean for the species much, and if no individual value was obviously anomalous, all data were included. In contrast, if only two data points were available, and differed by a factor of about 3, or if method or result details were limited or questionable, one or more values might not be used.

4. *Calculate the final acute value (FAV) and final chronic value (FCV)* – The FAV and FCV are acute and chronic effects concentrations corresponding to the 5th percentile genus in the respective datasets. More details and assumptions for this step are described later in the "Methods" section.

5. *Derive the hardness-dependent criterion maximum concentration (CMC or acute criterion) and the criterion continuous concentration (CCC or chronic criterion) equations using the hardness-toxicity slopes* – The CMC is equal to the FAV divided by two. This step extrapolates from a concentration that would likely be extremely harmful to sensitive species in short-term exposures (kill 50 percent of the population) to a concentration expected to kill few if any individuals. This assumption was recently supported by results of the acute toxicity testing of 20 species with 5 chemicals representing a broad range of toxic modes of action. In those data, multiplying the LC_{50} by a factor of 0.56 resulted in a low- or no-acute effect concentration (Dwyer and others, 2005b). No analogous adjustment was made to the FCV because it is derived using effects less severe than killing one-half the population of sensitive species.

Extrapolating Small Toxicity Test Datasets to Aquatic Communities

Aquatic-life criteria are intended to apply to a diversity of freshwater ecosystems and the species that inhabit them. Conceptually, it would be desirable to develop criteria for substances that address the diversity of ecosystems by conducting a series of field experiments on a wide variety of unpolluted water body types (for example, lakes, rivers, and streams that are warm or cold, large or small). However, such realism is infeasible, and instead of testing diverse communities or assemblages of organisms, criteria are usually derived from the results of single species toxicity tests on a variety of test organisms (Stephan and other, 1985). The problem of extrapolating toxicity values from small datasets to estimate values needed to protect diverse aquatic-life communities is illustrated by comparing the diversity of aquatic life in North America and Idaho to the diversity of organisms that have been tested with cadmium. At least 1,000 times more aquatic animals are native to North America than have been tested for their short-term (acute) sensitivity to cadmium (table 1). This disparity is greater for long-term (chronic) data because chronic tests are much more costly than acute tests, and because chronic tests can only be conducted with species that can be cultured or at least maintained in laboratories.

Table 1. Comparison of the diversity of selected aquatic assemblages in North America to the diversity of aquatic species with cadmium toxicity data available.

[Number of species in acute and chronic datasets are from this report. **Symbols:** ~, approximately; >, greater than]

Assemblage	Number of native species		Number of species in available datasets	
	North America	Idaho	Acute	Chronic
Zooplankton	[1] ~480	Unknown	5	4
Benthic macro-invertebrates	[1] >9,000	[2]>1,100	37	5
Fish	[3]~950	[4] 40	23	17
Other organisms	Unknown	Unknown	4	2

[1]Data from Thorp and Covich (2001).

[2]Data from Grafe and others (2002).

[3]Data from Matthews (1998).

[4]Data from Zaroban and others (1999).

In the United States, the approach generally has been to develop species-sensitivity distributions (SSDs) of acute toxicity. To account for untested and potentially sensitive taxa, a hypothetical LC_{50} for the 95th percentile most sensitive species is calculated from the SSD for a substance. This hypothetical LC_{50}, referred to as the “final acute value” by Stephan and others, 1985), then is divided by an overall acute-to-chronic ratio to estimate a chronic criterion that would not result in unacceptable adverse effects. Although chronic criteria could be derived directly from chronic data, because much fewer chronic data are available than acute data, in practice acute-to-chronic ratios have been more commonly used to estimate the chronic toxicity of untested species and derive chronic criteria in the United States (Stephan and others, 1985; Mount and others, 2003). For example, EPA developed aquatic-life criteria for about 25 “priority pollutants” and several other “nonpriority” pollutants or characteristics including ammonia and dissolved oxygen (U.S. Environmental Protection Agency, 2002b). Of these, only the chronic criteria for four pollutants (ammonia, cadmium, selenium, and saltwater dissolved oxygen) were developed directly with chronic effects data without using an acute-to-chronic extrapolation.

Extrapolating single-species toxicity values to derive criteria to protect ecosystems requires making some fundamental assumptions. These include:

1. The shape of the statistical distribution of the sensitivities of untested organisms;
2. Test data can be considered a random sample of the responses of all species, most of which are untested;
3. Acute and chronic responses in laboratory reasonably approximate those for the same species in the wild; and
4. The overall distribution of single species responses approximates the distribution that would be found in community responses.

A fundamental assumption for deriving aquatic-life criteria is that responses of a variety of individual species to a substance measured through single-species toxicity testing can be extrapolated to responses at the aquatic community level (Stephan and others, 1985; Erickson and Stephan, 1988). Species sensitivities are further assumed constant and are defined and predicted through models based on statistical distributions and thus are commonly referred to as species-sensitivity-distributions (SSDs). SSDs are statistical distributions estimated from toxicity data for a dataset of species responses and are considered to be a sample of an entire population (in the statistical sense) of species responses.

The idea of a SSD approach followed the observation that in multiple aquatic toxicity datasets of LC_{50} values, the species sensitivity was distributed in a rather constant way for most chemicals, in a distribution that resembles a lognormal one. Therefore, no one tested species is representative of any other species, but is one estimate of the general species sensitivity, that is, one point along the distribution (Posthuma and others, 2002). A further implicit assumption of the SSD approach to setting environmental criteria or in ecological risk assessment is that risks cannot be completely eliminated but should be reduced to an acceptable low risk. Thus, SSDs are used to calculate the concentration at which a specified proportion of species will be affected, referred to as the hazardous concentration (HC) for the p (percentile) of species (HCp) (Posthuma and others, 2002). The HC used in water-quality criteria derivation is the HC_5, where the HC_5 in aquatic-life derivation is derived from the 5th percentile of the genus mean acute or chronic values (GMAVs or GMCVs). The choice of the 5th percentile as the HC_5 value was simply based on the judgment that the 1st or 10th percentile would result in criteria that seemed too low or too high when compared with the datasets from which they were calculated. The 5th percentile was an easily recognizable number that was midway between the 1st and 10th percentile (Stephan and others, 1985; Stephan, 2002). The HC_5 level is the most commonly used value in the practice of using SSD to set environmental criteria or ecological risk assessment; however, the choice of this level probably is rooted in convention, for no clear ecological or toxicological reasons support the choice (Posthuma and others, 2002).

The description of the use of the 5th percentile of a SSD to derive water-quality criteria implies that 95 percent of the species in an ecosystem would be protected, and 5 percent could be sacrificed for each chemical criterion. If important species had toxicity values less than the 5th percentile, then criteria should be set lower than the 5th percentile to protect these species (Stephan and others, 1985; Posthuma and others, 2002). In their description of "important" species, Stephan and others (1985) considered only recreational or commercial value as reasons for considering a species important; however, over time descriptions of important or critical species in related contexts were broadened. Subsequent descriptions of important or critical species in the context of ecological risk assessment or site-specific aquatic-life criteria included four general categories of valued species:

1. Keystone species of great ecological value because their loss would indirectly affect many other species or ecosystem function;
2. Species that have been given special conservation status through treaties, laws, or policies as threatened, endangered, or vulnerable;
3. Recreationally valued species; or
4. Commercially important species (U.S. Environmental Protection Agency, 1994; Stephan and others, 1994b; Posthuma and others, 2002).

Erickson and Stephan (1988) developed a statistical procedure for estimating the 5th percentile of the SSD that was intended to minimize statistical bias in most datasets. Their procedure used extrapolation or interpolation to estimate the 5th percentile of a statistical population of genus mean values in which the available GMAVs were assumed to have been randomly obtained. The available GMAVs were ranked from low to high and the cumulative probability for each was calculated as $P=R/(N+1)$, where R = rank and N = number of GMAVs in the dataset. The calculation used the log-triangular distribution and the four GMAVs whose P values were closest to 0.05. The calculation procedure is not dependent upon the taxonomic level of the dataset, for example, species, genus, or family (Erickson and Stephan, 1988). Although previous versions of EPA's criteria derivation guidelines used species or family, in the 1985 version, criteria were determined at the genus level. On average, species within a genus are toxicologically more similar than species in different genera. The use of genus mean values is intended to prevent datasets from becoming biased by an overabundance of species in one or few genera.

Erickson and Stephan (1988) recommended a triangular-shaped statistical distribution of toxicity values. Figure 1 shows the concept of a triangularly distributed "true" population of sensitivities for North American species, overlain on the available samples of acute and chronic sensitivities to cadmium. The triangular-shaped distribution may look odd compared to the more familiar illustrations of bell-shaped, normal-distributions with a curved hump in the center and the tails of the data gradually tapering towards, but

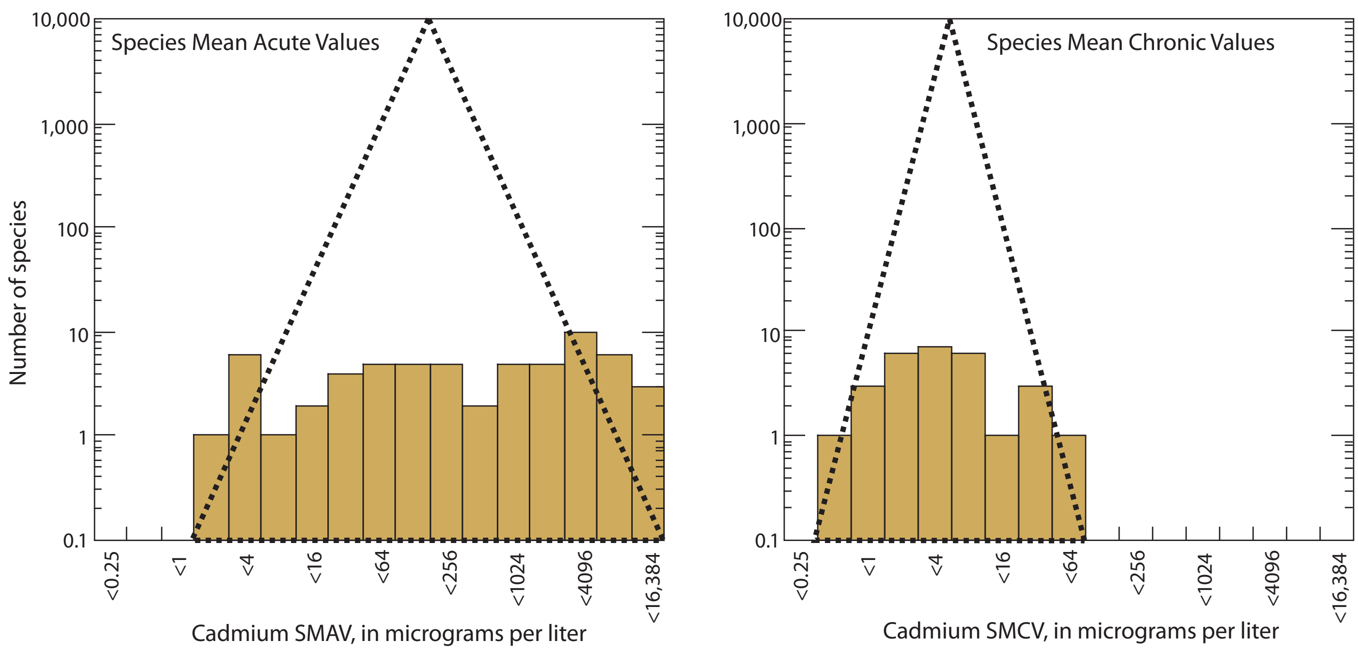


Figure 1. Comparison of the assumed distributions of species mean acute and chronic cadmium values (triangles) for all North American species and the distributions of measured values (bars).

never meeting zero. However, the triangular distribution better fits the toxicological assumption that SSDs do have definite lower and upper bounds (that is, no species is sensitive to infinitesimal concentrations and no species is resistant to any concentration of a substance). Further, in aquatic-life criteria derivation, the primary interest is the more sensitive lower corner of the distribution (5th percentile), so the unrealistic peak at the center of the distribution is unimportant.

An important feature of procedures that use the 5th percentile of SSDs for deriving criteria values is that if the dataset contains less than 20 genus mean values, the estimate of the 5th percentile will be determined by extrapolation, and if greater than 20 values are used, the 5th percentile will be determined by interpolation. This is significant because if the dataset is small, the extrapolated estimate will be lower than the lowest value in the dataset. As the size of the dataset increases, the 5th percentile estimate usually increases, and uncertainties in its estimate owing to extrapolation and bias are reduced (Erickson and Stephan, 1988; Newman and others, 2000).

The statistical procedure for deriving criteria values assumes that tested species are a random sample from the entire population of North American species (Erickson and Stephan, 1988). If this were the case, because insects and other invertebrates make up more than 90 percent of the aquatic animals in North American freshwaters, about 90 percent of the toxicity data for cadmium would be for invertebrates. As shown in the section “Diversity of Data,” such is not the case. Species are not selected at random for testing, but are selected based on factors such as cost, whether they are considered as surrogates for untested valued species, sensitivity to pollutants, availability of standard methods and test organisms, repeatability, happenstance, and regulatory requirements. Erickson and Stephan (1988), recognizing that species selection was more haphazard than random, tested various simulations with extreme deviations from random sampling and concluded that questions about the propriety of applying methods based upon random sampling to datasets where sampling was not strictly random probably was not of great importance.

The assumptions and practice of using the “5th percentile of a SSD” approach to setting environmental quality criteria and assessing ecological risk have been the subjects of spirited debate in the ecotoxicology literature (Forbes and Forbes, 1993a; Smith and Cairns, 1993; Calow and others, 1997; Power and McCarty, 1997; Aldenberg and Jaworska, 2000; Newman and others, 2000; Forbes and Calow, 2002; Posthuma and others, 2002; Selk and others, 2002; Suter and others, 2002b; Fisher and Burton, 2003; Brix and others, 2005; Maltby and others, 2005). The root of the debates are arguments that fundamental assumptions of the approach may not be met or are untested. Some concerns include:

1. Whether haphazard collections of data from single-species laboratory toxicity tests can be considered representative of any natural ecosystems;

2. Whether small datasets can be significantly biased toward more or less sensitive species than would be expected in natural ecosystems;

3. Whether any species loss from a community due to a toxin is acceptable. Supporting arguments are that the 5th percentile of a SSD is simply a pragmatic “statistical cutoff” to estimate a predicted low- or no-effect concentration to protect ecosystems, rather than a real sacrifice of species. Further, accepting some species loss, especially among “lower” organisms, is acceptable because ecosystems have enough functional redundancy to absorb species loss. The counterpoint to this redundant species hypothesis combines the “rivet popper” hypothesis (community integrity is reduced by each loss of a species) with the argument that a conservative stance is best when faced with uncertainty in ecosystem functions;

4. Whether the 5th percentile of the SSD is the appropriate level of protection or just a familiar number;

5. Reducing community integrity to a simple proportion of species could discount keystone or dominant species if they were in the lower 5th percentile of sensitivity;

6. The approach depends on comparable data which results in a bias toward mortality data, which are most abundant, and a bias against data on abnormal behavior or other sublethal data that may be as important for maintaining biological integrity;

7. The few species for which multiple tests results are available sometimes show high variability in sensitivity. Thus, apparent differences between species’ ranks on a SSD may be unreliable, especially for species with only single or few datapoints. and,

8. Statistical uncertainties.

Because of these concerns, this report goes beyond the minimum steps to derive criteria values following Stephan and others (1985) guidelines and, as the available data allow, evaluated these concerns in the context of cadmium effects to relevant aquatic ecosystems. Evaluations included predictions of the consequences of not protecting specific species that have cadmium sensitivities in the lower 5th percentile of the SSD, and comparisons of the criteria derived through single-species testing in laboratory conditions to model ecosystem or field studies.

Minimum Species-Diversity Required for Criteria Derivation

The statistical desirability of selecting species at random to derive a statistical sample to base criteria upon is countered by the toxicological desirability of targeting for inclusion of specific groups that were considered important and were expected to be sensitive to many substances. Thus, Stephan and others (1985) recommended that the minimum data requirements for deriving aquatic life criteria include results from at least eight different families representing important groups of taxa (table 2). The minimum species-diversity guidelines were focused on acute toxicity data for developing a final acute value, and included procedures for extrapolating a chronic criterion from acute results based on acute-to-chronic ratios (ACRs). However, if chronic data met the minimum eight-family rule, as is the case for cadmium, a chronic criterion should be developed using directly from the chronic data using the same procedures that Erickson and Stephan (1988) developed for acute data.

Important Elements and Assumptions of Criteria Derivation

Stephan and others' (1985, p. 18) guidelines for deriving aquatic-life criteria provide an objective and rigorous analytical framework, but much of their guidance is necessarily qualitative rather than quantitative. They caution,

Table 2. Comparison of the minimum taxonomic diversity needed to derive aquatic-life criteria and the diversity represented in the dataset of chronic effects of cadmium on aquatic animals.

[**Minimum species diversity** from Stephan and others (1985)]

Minimum species diversity needed to derive aquatic life criteria	Family	Species represented
A representative of:		
1. The family Salmonidae in the class Osteichthyes	Salmonidae	Atlantic salmon, *Salmo salar*
		Brook trout, *Salvelinus fontinalis*
		Brown trout, *Salmo trutta*
		Bull trout, *Salvelinus confluentus*
		Chinook salmon, *Oncorhynchus tshawytscha*
		Coho salmon, *Oncorhynchus kisutch*
		Lake trout, *Salvelinus namaycush*
		Rainbow trout, *Oncorhynchus mykiss*
2. A second family in the class Osteichthyes (preferably including important warm water species)	Catostomidae	White sucker, *Catostomus commersoni*
	Centrarchidae	*Lepomis macrochirus*, Bluegill
	Centrarchidae	Smallmouth bass, *Micropterus dolomieui*
	Cichlidae	Blue tilapia, *Oreochromis aurea*
	Cottidae	Mottled sculpin, *Cottus bairdi*
	Cyprinidae	Fathead minnow, *Pimephales promelas*
	Cyprinodontidae	American flagfish, *Jordanella floridae*
	Esocidae	Northern pike, *Esox lucius*
	Percidae	Fountain darter, *Etheostoma fonticola*
3. A third family in the phylum Chordata (may be in the class Osteichthyes or an amphibian)	Bufonidae	American toad, Bufo americanus
	Ambystoma	Northwestern salamander, *Ambystoma gracile*
4. A planktonic crustacean (cladoceran, copepod, etc.)	Daphniidae	Cladoceran, *Ceriodaphnia dubia*
		Cladoceran, *Ceriodaphnia reticulata*
		Cladoceran, *Daphnia magna*
		Cladoceran, *Daphnia pulex*
5. A benthic crustacean (amphipod, crayfish)	Gammaridae	Amphipod, *Gammarus fasciatus*
	Hyalellidae	Amphipod, *Hyalella azteca*
6. An insect	Chironomidae	Midge, *Chironomus tentan*s
7. A family in a phylum other than Arthropoda or Chordata (Annelida, Mollusca, etc.)	Aeolosomatidae	Polychaete worm, *Aelosoma headley*i
8. A family in any order of insect or any phylum not already represented	Physidae	Mollusc, *Aplexa hypnorum*

> *"...much judgment will usually be required to derive a water-quality criterion for aquatic organisms and their uses"* and that *"All necessary decisions should be based on a thorough knowledge of aquatic toxicology and an understanding of these Guidelines and should be consistent with spirit of these Guidelines, i.e., to make best use of the available data to derive the most appropriate criteria."*

This analysis strives to adhere to these principles. Several considerations from the guidelines that were important in this analysis, deviations from the guidelines, and assumptions are described below:

Focus on aquatic animals—U.S. Environmental Protection Agency (2001) compared the relative sensitivities of aquatic plants and animals to cadmium toxicity and concluded that criteria that adequately protect aquatic animals and their uses would be sufficient to protect aquatic plants as well. Therefore, no further investigation was done regarding plant toxicity. Only one of the studies compiled by U.S. Environmental Protection Agency (2001) reported cadmium effects at a concentration (2 µg/L) in the sensitivity range of sensitive animals, with all other plant values considerably higher. Also, the 2 µg/L value was from a test that did not meet data acceptability guidelines: cadmium concentrations were not measured, water hardness and the duration of the test were not reported.

Tissue residue values and bioaccumulation or bioconcentration—EPA's criteria derivation guidelines include a procedure for deriving a "final residue value" to prevent tissue concentrations in important aquatic species from exceeding applicable U.S. Food and Drug Administration (FDA) action levels and to protect wildlife, including fishes and birds, that consume aquatic organisms from demonstrated unacceptable effects. The final residue value is defined as a concentration in water and is calculated by dividing a maximum permissible tissue concentration by appropriate bioconcentration or bioaccumulation factors. For wildlife, a maximum permissible tissue concentration could be established by defining a maximum acceptable dietary intake based on observations of survival, growth, or reproduction in a chronic feeding study or a long-term field study (Stephan and others, 1985). No regulatory action levels for cadmium and human health applicable to the consumption of aquatic organisms were located on the FDA's websites, therefore, no basis exists for developing a residue-based criterion for cadmium based on exceeding FDA action levels. For wildlife, literature on effects of dietary cadmium and effects associated with tissue residues from field studies was evaluated.

Direct calculation of a final chronic value is preferable to extrapolations using acute-to-chronic ratios—In EPA's aquatic-life criteria documents, chronic criteria are most commonly based on the final acute value that was derived for acute criteria, and then using an acute-to-chronic ratio, were extrapolated to chronic criteria values. This extrapolation is often necessary because chronic data are scarcer than acute data. Chronic testing requires more time and resources than acute testing and chronic test methods have been developed for fewer species than have acute test methods. However, if sufficient chronic values are available, derivation of a chronic criterion directly from chronic values is considered preferable to extrapolating a chronic criterion from acute values. Stephan and others (1985) advise that if chronic values are available for species in at least eight specified families, the final chronic value should be calculated using the same statistical procedure as that used to develop the final acute value. Because chronic values were available for diverse enough species to meet the eight family minimum data requirements to use the FAV procedure with chronic data (table 2), updated ACRs and a final ACR (overall ACR for cadmium) are reported here, but were not used in the chronic criterion calculations.

Updated ACRs may have utility should the cadmium criteria developed following the National procedures be modified to reflect site-specific conditions. Water-quality criteria for aquatic life developed following the National guidelines may be under- or over-protective at a specific site if the species at the site are more or less sensitive than those included in the dataset, or if physical and (or) chemical characteristics of the site alter the biological availability and (or) toxicity of the chemical. EPA established three procedures for the development of site-specific water-quality criteria. Each procedure uses a final ACR for a substance (U.S. Environmental Protection Agency, 1994):

1. The recalculation procedure is intended to take into account relevant differences between the sensitivities of the aquatic organisms in the nationally based dataset and the sensitivities of organisms that occur at the site. Organisms are added or deleted from the National dataset and criteria recalculated. If the eight-family diversity rule is not met for chronic values, chronic criteria would be calculated using a final ACR.

2. The water-effect ratio (WER) procedure is intended to take into account relevant differences between the toxicities of the chemical in laboratory dilution water and in site water. Usually paired acute toxicity tests using "standard" test organisms such as the fathead minnow are conducted in site-water and a "typical" laboratory dilution water, providing a water-effect ratio of site and laboratory water LC_{50s}. The site-specific criteria are derived by multiplying the Statewide acute criterion by the WER, and dividing that result by the final ACR provides a chronic criterion.

3. The resident species procedure is intended to take into account both differences simultaneously. In this procedure, the acute criterion is derived for a site by testing at least eight species in site water and calculating an acute criterion using National guidelines. The chronic criterion may be derived using the final ACR for the chemical, or a site-specific ACR may be developed.

Data Sources and Inclusion Requirements

Because toxicity testing can be done using various methods, guidelines for data comparability and inclusion are needed. Stephan and others (1985) offer several specific guidelines for data acceptability for criteria derivation. Based on the data reviewed herein, additional decisions were made regarding which results were suitable for pooling in the criteria derivation dataset. The term “acceptable data” as used in this context simply refers to whether testing methods were sufficiently similar to one another, that differences in results can be reasonably attributed to differences in species sensitivity, characteristics of test water, or biological variability rather than testing artifacts. All data were presumed acceptable for the purposes for which they were generated.

General guidelines include:

Exclusion of unwritten or otherwise questionable data – Data should be available in a written form, such as a publication, manuscript, letter, or memorandum (electronic mail was not widely used in 1985, but is considered an acceptable written form here). Confidential or other information not available for distribution should not be used. Questionable data, whether published or unpublished, should not be used. For example, data from tests that did not contain a control treatment, tests in which too many control organisms died or were stressed, or tests using mixtures of chemicals should not be used (Stephan and others, 1985).

Resident in North America —Stephan and others (1985) recommend rejecting data for species with no reproducing wild populations in North America. For this report, this data-criterion was modified to rejecting families that are not known to have reproducing wild populations in North America. This modification is based on the premise that (1) taxa within a genus are toxicologically much more similar than taxa in different families, and (2) families may be more widely distributed than species. For example, aquatic insect assemblages in cool, temperate streams in North America and New Zealand share few if any native animal species, but share many families that have shown similar sensitivities to metals (Hickey and Golding, 2002). This residency provision also is problematic because introduced species are steadily becoming established, and arguably, invasive nuisance species need not be protected or considered in criteria. However, following the concept that each tested species is not representative of any other species but is one estimate of the general species sensitivity, no exclusions were made for species that can be considered invasive nuisances.

Life stage sensitivity—Because a species can only be considered protected from acute toxicity if all life stages are protected, Stephan and others (1985) recommended that if the available data indicate that some life stages are at least a factor of two more resistant than other life stages, the data for the more resistant life stages should not be used to calculate species mean acute values. Smaller, juvenile life stages of fish and early invertebrate instars are commonly expected to be more vulnerable to metals toxicity than larger, older life stages of the same species. In acute testing methods, fish are recommended to be post-larval or older and actively feeding, daphnids less than 24 hours old; amphipods, mayflies, and stoneflies in an early instar; and midges in the second or third instar (American Society for Testing and Materials, 1997).

Static versus flow-through exposure systems—For some highly volatile, hydrolysable, or degradable materials, it is probably appropriate to use only results of flow-through tests (tests in which the test solutions are constantly being replenished) in which concentrations of the test solutions were measured often enough using acceptable analytical methods.

Tests with fed organisms—In most instances, acute test data were only used if the animals were not fed during the tests. Stephan and others (1985) advise that,

> *“...results of acute tests during which the test organisms were fed should not be used, unless data indicate that the food did not affect the toxicity of the test material.”*

The reason for this guidance is that fecal matter and uneaten food will decrease the dissolved-oxygen concentration and the biological activity of some test materials. These problems are most severe with the static technique, but sometimes are important with the renewal and flow-through techniques (American Society for Testing and Materials, 1997).

This was primarily an issue for very small organisms that do not have the energy reserves to survive the customary 96-hour duration of an acute toxicity test without feeding. Because of this, Stephan and others (1985) recommended that only the results of 48-hour toxicity tests be used to determine mean acute values for cladocerans and midges, instead of 96 hours used for other organisms. Although this recommendation avoids the potential problems of introducing organic material into test vessels, it introduces a potential bias toward more resistant mean acute values resulting from the shorter test durations. Suedel and others (1997) reported that LC_{50} values from 48-hour toxicity tests with cadmium and the cladocerans *Ceriodaphnia dubia* and *Daphnia magna* and the midge *Chironomus tentans* were 2 to 4 times higher than 96-hour LC_{50s}. Because comparisons among mean acute values for different taxa assume that values are unbiased, uncertainty is added when pooling or comparing values that were determined from both 48- and 96-hour tests, such as species-sensitivity rankings and acute to chronic ratio comparisons.

The data reviewed did not consistently indicate that feeding would appreciably affect the toxicity of cadmium in acute tests. Lewis and Weber (1985) reported no effect of feeding on the sensitivity of *Daphnia pulex* to cadmium, but *Daphnia magna* were more resistant. However, feeding did not appear to make *Daphnia magna* more resistant in tests by Nebeker and others (1986a). Severe starvation of the amphipod *Hyalella azteca* prior to acute toxicity tests with cadmium did not result in lower LC_{50s}. Starved amphipods exposed to low cadmium concentrations actually survived longer than control organisms (McNulty and others, 1999). Therefore, tests in which organisms were fed were evaluated for inclusion in this report on a case-by-case basis.

Feeding and nutritional status of test organisms is important in longer exposures as well. For example, fewer mortalities, larger growth, and higher reproductive rates were observed with *Hyalella azteca* tested in nutrient rich effluents than with *Hyalella azteca* tested in nutrient poor reconstituted laboratory waters (Stanley and others, 2005). In contrast, tests with daphnids reared with different food rations indicated that cadmium exposure significantly decreased survival, growth, and reproduction and this decrease was more pronounced with *increasing* food concentration. Therefore, the sensitivity of daphnids to stress increased with increasing food ration. This increased sensitivity is likely the result of a change in life history from emphasizing survival at low food supply to stressing reproduction at high food supply (Smolders and others, 2005).

Concordance with other data for taxa —Values that appear to be questionable in comparison with other acute and chronic data available for the same species and for other species within the same genus probably should not be used. For example, if after adjustment for hardness, acute values for a species or genus differ by more than a factor of 10, rejection of some or all values probably is appropriate (Stephan and others, 1985).

Methods for Evaluation of Risks to Biological Integrity and Vulnerable Species under Criteria Conditions

Following derivation of criteria values, the values were evaluated in the context of whether waters with cadmium concentrations near criteria conditions would be expected to maintain biological integrity. This effort included:

1. Evaluating the criteria concentrations that were derived from toxicity tests done in laboratories under constant conditions in the context of fluctuating cadmium concentrations in field conditions;
2. Critically examining the assumption that the most sensitive 5 percent of taxa from a species-sensitivity distribution need not be explicitly protected in criteria derivation by considering the role that a more-sensitive "unprotected" species has in aquatic food webs and through population modeling; and
3. Comparing apparent effects concentrations from field surveys or ecosystem studies to corresponding criteria concentrations.

Derived criteria values were additionally evaluated to predict if they would harm or kill species listed under the Endangered Species Act as threatened or endangered species. Estimates of no-observed-adverse-effect-concentrations (NOECs) were made for several threatened or endangered species and compared to corresponding chronic criterion concentrations. The use of NOECs is more conservative than the method used to derive most "chronic values" for the criteria derivation. For criterion derivation, most chronic values were obtained by taking the geometric mean of the NOEC and the lowest-observed-adverse-effect concentration (LOEC) from a test. Because the chronic value lies between a no- (or presumably at least a small effect) concentration and a concentration that did cause adverse effects, by design chronic values could allow some amount of adverse effects. NOECs were selected as the test parameter used to make effects determinations of chronic criteria to listed species in EPA's methods manual for conducting biological evaluations of water quality criteria (U.S. Environmental Protection Agency, 2003b).

Results

Results are presented in three major interrelated parts:

1. Data compilations, data reduction, and criteria derivation following the steps in Stephan and others' (1985) guidelines. The steps consisted of:
   a. Compiling and evaluating datasets of acute and chronic cadmium values (tables 15 and 16, at back of report);
   b. Compiling a dataset of other data that were not directly comparable to the acute and chronic datasets, but were still pertinent (table 17, at back of report);
   c. Determining hardness-toxicity relations for acute and chronic effects of cadmium to various species (tables 3-5);
   d. Calculation of acute to chronic toxicity ratios (table 6);
   e. Ranking the sensitivities of the hardness-adjusted acute-chronic values for each genus (table 7);
   f. Calculation of the 5th percentile of the ranked acute and chronic genus mean values (final acute value and final chronic value, respectively (table 8);
   g. Derivation of hardness-dependent acute and chronic criteria equations (table 9).

2. An interpretive synthesis of the aquatic toxicology of cadmium in the context of the derived criteria, including a review of some effects data that could not be directly used in the criteria derivation. This synthesis includes observations on the importance of alternative testing methods on the acceptability of data. These include factors such as alternative exposure designs (flow-through, renewal, or static), whether organisms were or were not fed, life stage sensitivity including the influence of the life stage when tests began on the acclimation of later life stages to cadmium, duration of chronic tests, effects associated with bioaccumulation and dietary exposures to cadmium (table 10), and behavioral effects of cadmium.

3. These analyses were followed by “validation” evaluations of the suitability of the derived cadmium criteria using different types of data than were used to develop them. These included considerations for applying the criteria to variable environments (fig. 3), population modeling (fig. 4), ecosystem experiments and field surveys (figs. 5, 6, and 7; table 13), and comparisons of the sensitivities of threatened or endangered species to comparable criteria concentrations (table 14).

Criteria Derivation Results

Data Compilation and Evaluation

Stephan and others (1985) provided general guidelines on data acceptability for use in criteria calculations. Following the review of a large number of relevant studies, more specific considerations about which data were useful were made as follows.

The guideline not to use studies “if too many organisms in the control treatments died” requires judgments of how many deaths are too many. In some instances, specific criteria have been published. For example, in acute testing of fishes, macroinvertebrates and amphibians with exposures of 48–96 hours, control survival should be at least 90 percent. In early life stage testing of fishes, tests should be considered unacceptable if following thinning of the embryos, control survival is less than 70 percent (American Society for Testing and Materials, 1997, 1998). However, in some tests no clear threshold was obvious, such as with chronic tests using less commonly tested species. In these cases, it was assumed that if a concentration-effect was observed with cadmium and if controls and low treatment concentrations were statistically distinguished from higher treatments by analysis of variance (ANOVA), then the observed responses likely were reliable.

Some tests reported concentrations of cadmium or other metals (for example, copper, lead, or zinc) at background concentrations that approached or exceeded contemporary chronic aquatic-life criteria (for example, Pickering and Gast, 1972; Chapman, 1978b; Rombough and Garside, 1982). Because these tests results showed a concentration—response to the cadmium treatments and little control mortality, the presence of other metals probably were toxicologically insignificant or background metals concentrations possibly were biased high, artifacts of “pre-clean” field and laboratory procedures (Stephan and others, 1994a). These tests, and others with no reported background metals concentrations, but that were done at the same laboratories were not considered to be disqualified as mixture tests or as having pre-exposed the test organisms to high concentrations of the test substance or other contaminants according to guidelines by Stephan and others (1985).

Static, Renewal, and Flow-Through Exposures

Exposures of test organisms to test solutions are usually done through variations on three techniques. In “static” exposures, test solutions and organisms are placed in chambers and kept there for the duration of the test. The “renewal” technique is like the static technique except that test organisms are periodically exposed to fresh test solution of the same composition, usually once every 24 or 48 hours, by replacing nearly all the test solution. In the “flow-through” technique, test solution flows through the test chamber on a once-through basis throughout the test (American Society for Testing and Materials, 1997). The term “flow-through test” is sometimes misunderstood to be a test with flowing water that mimics a lotic environment. Rather, the term refers to the once-through, continuous delivery of test solutions (or frequent delivery in designs using a metering system that cycles every few minutes). Flows on the order of about 5-volume replacements per 24 hours are insufficient to cause appreciable flow velocities.

Stephan and others (1985, p. 22) advise that for “some highly volatile, hydrolysable, or degradable materials it is probably appropriate to use only results of flow-through tests in which the concentrations of the material were measured often enough using acceptable analytical methods.” Because at environmentally realistic concentrations, cadmium is highly soluble and is not highly volatile, hydrolysable, or degradable, this aspect of the guidelines would not appear to be relevant in this analysis. However, following this broad statement of principle, in guidelines for calculating species mean acute values (SMAVs) preference is given to flow-through tests when available, but if data from flow-through tests are not available for a species, then renewal tests where only the initial concentrations were measured, followed by data from static exposures in which the actual (as opposed to intended) concentrations were never measured. Renewal tests that measured test concentrations at the beginning and ending of the test were not mentioned (Stephan and others, 1985, p. 29-30, 36). American Society for Testing and Materials (1997) similarly gives a preference for renewal or flow-through tests over static tests,

> *"...because the pH and concentrations of dissolved oxygen and test material are maintained at desired levels and degradation and metabolic products are removed, tests using renewal and flow-through methods are preferable and may last longer than 96 h; test organisms may be fed during renewal and flow-through tests. Although renewal tests might be more cost-effective, flow-through tests are generally preferable."*

In the 1985 guidelines, the rationale for the general preference for flow-through exposures was not detailed, but probably was based on assumptions that static exposures will result in LC_{50s} that are biased higher (apparently less toxic) than comparable flow-through tests (for example, Pickering and Gast, 1972). Additionally, flow-through tests are assumed to have more stable exposure chemistries and will result in more precise LC_{50} estimates. When testing with cadmium, the assumption of test bias was not supported by recent information.

Test bias—Pickering's and Gast's (1972) study of acute and chronic responses of fathead minnows to cadmium may have been one basis for the recommendation that flow-through bioassays be used preferentially over static bioassays. The flow-through LC_{50s} were lower than the static LC_{50s} (~4,500–11,000 µg/L for flow-through tests versus about 30,000 µg/L for static tests). The fish used in the static tests were described as "immature" weighing about 2 g (2,000 mg). The size of the fish used in Pickering and Gast's (1972) flow-through acute tests was not reported, but probably was similar. In contrast, 8- to 9-day old fathead minnow fry usually weigh about 1 mg or less (U.S. Environmental Protection Agency, 2002c). Using newly hatched fry weighing about 1/1,000th of the fish used in the 1960s by Pickering and Gast (1972), more recent cadmium LC_{50s} for fathead minnows at similar hardnesses tend to be around 50 µg/L with no obvious bias for test exposure (table 16). The use of large fish probably contributed to the exorbitantly high LC_{50s} obtained, and at very high concentrations of about 30,000 µg/L may have exceeded solubility limits resulting in undetected (if not measured) losses from solution during the test. This indicated that fish actually experienced lower concentrations over the course of the test than measurements at the beginning of the test would indicate, resulting in LC_{50} estimates that were biased high.

More recent testing with copper toxicity to fathead minnows has shown that for some test combinations, flow-through tests tended to produce lower LC_{50s} than static or renewal tests (Santore and others, 2001). However in contrast to earlier recommendations favoring flow-through testing, Santore and others (2001) suggested that flow-through tests were biased low because copper complexation with organic carbon, which reduces acute toxicity, is not instantaneous and typical flow-through exposure systems allowed insufficient hydraulic residence time for complete copper-organic carbon complexation to occur. (Similar findings were made in tests with cadmium and carbonate complexation; Davies and Brinkman [1994b]).

Consequently, it could be argued that flow-through tests with low hydraulic residence times could be less relevant to field conditions and should not be used if renewal or static tests are available. However in this report, flow-through and renewal tests were used with equal priority in regressions and to estimate species mean acute values. This also is the approach used in U.S. Environmental Protection Agency's (2003a) update of aquatic-life criteria for copper. Because hydraulic residence time and contact time in field settings would be variable, rather than assuming that one type of exposure technique best represents field conditions, if static, renewal, and flow-through tests otherwise met data acceptability guidelines, all were used.

The decision in this analysis not to give singular importance to the exposure technique is further supported by examples of cadmium acute toxicity. For rainbow trout, a well-tested and sensitive species, the two most sensitive values (<1 µg/L when adjusted to a hardness of 50 mg/L) were obtained using flow-through and renewal testing (table 15). However, the two least sensitive acute rainbow trout values also were obtained with flow-through exposures (>9 µg/L when adjusted to a hardness of 50 mg/L). For fathead minnows, in contrast with flow-through values of >4,500 µg/L at a hardness of 200 mg/L obtained by Pickering and Gast (1972), static values of about 1,000 times lower were obtained in tests in even harder water (280 mg/L), but using newly hatched fish that were only 24-48 hours old (table 15). Phipps and Holcombe (1985) obtained an LC_{50} of 1,500 µg/L in flow-through testing using Lake Superior water with a hardness of 44 mg/L; this value is about 100 times higher than acute values obtained through static testing with young fry in Lake Superior water at similar hardness (table 15). Pickering and Gast (1972) and Phipps and Holcombe (1985) both used about the same size fish, 0.6 g. Similar results were reported with brook trout. One each flow-through and static acute tests with brook trout were located, both conducted in waters of similar hardness (41–47 mg/L). The LC_{50} of the static test that used fry was less than 1.5 µg/L where the LC_{50} of the flow-through test using yearlings was greater than 5,000 µg/L (table 17). These examples show that factors other than test type (flow-through or static), especially the life stage of exposures, can have more influence on acute toxicity of fish to cadmium. Static, renewal, and flow-through test values were all used unless inspection of individual test results suggested bias.

Tests with No Analytical Confirmation of Exposure Concentrations

In aquatic toxicity testing, treatment concentrations often are described as "nominal" for intended target test concentrations and "measured" for concentrations measured through laboratory analysis of the chemical in water. Stephan and others (1985) recommended using the results of tests with measured concentrations in preference to those for which the intended nominal concentrations were not confirmed.

However, they also recommended that

> *"...for a species for which no such result is available, the species mean acute value should be calculated as the geometric mean of all available acute values, i.e., results of flow-through tests in which the concentrations were not measured and results of static and renewal tests based on initial concentrations (nominal concentrations are acceptable for most materials if measured concentrations are not available) of the test material."*

The limited acceptance of some unmeasured test values probably was partly influenced by the practical consideration that minimum species diversity requirements might not be met with available data for some chemicals if all unmeasured results were excluded. Sufficient data were available to derive criteria using only test values based on measured concentrations for the cadmium dataset compiled here. Further, for the acute and chronic datasets, some of the lowest reported results were for species for which the only test values located were from unmeasured exposures. These species (striped bass, *Morone saxtilis* and the cladoceran *Moina macrocopa*) also were the only representatives of their genus. In the calculations of the final acute and chronic criteria values, the acute and chronic datasets were censored so that only the four most sensitive genus values were used. Thus, in preliminary calculations the final acute and chronic values were influenced by questionable values from single test results, because the intended exposure concentrations were not measured. It therefore seemed appropriate to exclude from the final acute and chronic value calculations any influential values that were questionable because they were obtained from unmeasured exposures. However, instead of selectively excluding only unmeasured values that appeared sensitive, it was more straightforward to simply exclude all test values that were obtained without analytical confirmations of exposure concentrations. Several noteworthy tests that used unmeasured exposures are listed in table 17, such as tests with unusually low effects values or tests with threatened or endangered fish or molluscs, or closely related surrogates.

Life-Cycle Versus Shorter-Term Chronic Data

For this analysis, data from long-term "chronic" exposures were used quantitatively (for example, in hardness-toxicity regressions and in species mean value calculations) if they were obtained using life-cycle methods or from shorter term methods for estimating chronic toxicity if the results obtained with the shorter-term methods were more sensitive, or similar in sensitivity to those results obtained from life cycle testing. This is a modification from Stephan and others (1985) recommendations to use life-cycle or partial life-cycle results in preference to shorter-term methods when data from both are available. This modification has the practical advantage of increasing the amount of available data, such as allowing the development of hardness—chronic toxicity relations in lieu of applying hardness—acute toxicity relations to chronic data. For fish species, life-cycle data are too limited to develop such relations. More importantly, in the 20 years since Stephan and others (1985) made their recommendations, more data have become available that indicate that, at least for fish, the life stage at which long-term exposures are initiated (for example, embryo or newly hatched) can influence the test sensitivity at least as much as the test duration.

Shorter-term methods that test apparently sensitive life stages were favorably compared with longer-term life cycle or partial life cycle tests, based on similar sensitivities of responses (McKim, 1977, 1985; Norberg-King, 1989). However, Suter and others (1987) cautioned that early-life stage testing sometimes can underestimate the toxicity of substances as compared in life-cycle tests. Regardless, numerous tests have shown that reduced sensitivity of either life cycle or early-life stage tests may occur if the tests are initiated during an insensitive life stage (for example, embryos or adults). Initiating metal exposure at the resistant embryo stage may allow acclimation, which may mute later responses of otherwise sensitive life stages (Sinley and others, 1974; Spehar, 1976; Chapman, 1978a, 1982, 1994; Spehar and others, 1978b; Brinkman and Hansen, 2004a, 2004b).

Brinkman and Hansen (2004a) conducted three pairs of chronic tests with brown trout in which exposures began at the embryo stage (early-life stage test) or at the swim-up fry stage (juvenile growth and survival test). Tests initiated at the swim-up fry stage and with a 30-day duration were 2–3 times more sensitive than paired tests initiated at the embryo stage and maintained for 55 days. In contrast, Norberg-King (1989) compared the effects of zinc exposures to newly hatched fathead minnows that either were pre-exposed to zinc as embryos or not and found little difference from the pre-exposures. Results of comparative testing with copper vary. Chapman (1994) found that the most sensitive test for rainbow trout was exposure beginning at swim-up with no prior exposure to provide acclimation. The least sensitive test was to begin exposure either prior to hatch at the eyed stage or at hatch. Initiating exposures at these stages provided some opportunity for acclimation prior to reaching the sensitive swim-up stage. However, Besser and others (2005) observed little difference in results of copper toxicity tests with rainbow trout as swim-up fry and maintaining exposures for 30 days or by initiating tests as embryos in classic early-life stage tests and maintaining exposures for 60 days. In summary, the available literature with metals indicate that for shorter-term (shorter than life cycle) methods for estimating chronic toxicity, juvenile growth and survival tests initiated as fry often, but not always, provide more sensitive data than early-life stage initiated as embryos. Thus in this report, chronic values from each type of test were used without giving any priority to one over the other. Of the data reviewed, no life-cycle test results with cadmium and fish were appreciably more sensitive than early-life stage test results with the same species. Therefore, juvenile growth and survival, early-life stage tests, and life-cycle tests all were used together to estimate mean chronic values for fish species.

Limiting estimates of chronic values to life-cycle test data would have the practical disadvantage of limiting the chronic dataset for fish to a small, static dataset that mostly was generated 20–40 years before this report. This is because for most fish species, life cycle tests of about 9 months to more than 3 years are extremely labor intensive, costly, and have been infrequently published. For example, no studies of cadmium and fish with life-cycle toxicity tests were reviewed that were conducted within the last 10 years, two within the last 20 years, and about 6 more within the last 40 years (Pickering and Gast, 1972; Benoit and others, 1976; Brown and others, 1994; U.S. Environmental Protection Agency, 2001).

The fathead minnow is the standard fish species used to estimate chronic effects of effluents in regulatory toxicity tests (U.S. Environmental Protection Agency, 2002c). Results of two life-cycle tests with cadmium conducted by Pickering and Gast (1972) in the mid 1960s were about 2–5 times higher than results of recent 7-day short-term chronic tests that were conducted at higher hardness levels (table 17). The life-cycle tests were initiated with much larger (about 600 mg) fish than those used in the contemporary short-term tests, which commonly weigh <1 mg at the end of the test (for example, Castillo and Longley, 2001; U.S. Environmental Protection Agency, 2002c).

Fewer comparative data on exposure duration and responses to cadmium were located for invertebrates than for fish. Winner (1988) found that a 7-day exposure was insufficient to elicit sensitive responses from *Daphnia magna* in comparisons with 7-day *Ceriodaphnia* exposures or 21-day *Daphnia magna* exposures. With the amphipod *Hyalella azteca*, Ingersoll and Kemble (2001) obtained LC_{25} values of 2.7 to 2.1 µg/L from 10- to 28-day exposures, respectively, with little additional decrease from 28 to 42 days. In contrast, in regression analyses of responses of *Hyalella azteca* to cadmium over a hardness range of 17–280 mg/L, most variability in responses was accounted for by hardness, despite exposure durations varying from 14 to 42 days.

An issue related to test exposure duration was whether exposures that were interrupted shortly before planned test termination could still provide useful data. Exposure durations in early-life stage tests with salmonids should be at least 60 days (American Society for Testing and Materials, 1998). However, Brinkman and Hansen (2004a) reported results from three tests with brown trout that were exposed for only 55 days and C.A. Mebane, D.P. Hennessy, and F.S. Dillon (unpub. data., 2004) obtained results from a rainbow trout test that exposed fish for only 53 days. Similarly, Chadwick Ecological Consultants, Inc. (2004a) exposed *Daphnia pulex* to cadmium for only 18 days instead of the 21 day protocol duration. For these situations, the data were included in the analyses following a case-by-case review that compared the hardness-adjusted results to those obtained from other tests with the same species, and where data on responses over time were reported, whether or not mortality rates had leveled off.

Statistical Interpretation of Chronic Tests

Chronic data particularly were evaluated carefully because previous investigators used a variety of terms and definitions to report results, and because no consistent guideline on what magnitude over effect is considered an "unacceptable" effect. Stephan and others (1985) advise that a chronic value may be obtained by calculating the geometric mean of the lower and upper chronic limits from a chronic test or by analyzing chronic data using regression analysis. Lower and upper chronic limits often were based on statistical hypothesis testing and correspond to the no- and lowest-observed-effect concentrations. In aquatic toxicology literature, typically, the no-observed-effect concentration (NOEC) is the highest test concentration that results in responses that are not statistically different from the control responses. The lowest-observed-effect concentration (LOEC) is the first treatment that is greater than the NOEC. Between these "chronic limits," is a hypothetical maximum acceptable toxicant concentration (MATC), which is an assumed threshold for toxic effects. The point estimate of a MATC is the geometric mean of the NOEC and the LOEC (McKim, 1985).

Stephan and others (1985) used precise language to describe how chronic values may be obtained with a subtle but important difference from the usual NOEC and LOEC definitions. The term "not statistically different" was replaced with "did not cause an unacceptable amount of adverse effect[1]" (The full definition of chronic limits is given in the glossary.) This language recognized important limitations inherent in statistical hypothesis testing for "significance." Chronic responses often are considered significant if statistical hypothesis testing indicates less than a 5 percent likelihood (a significance level of 0.05) that apparent differences in test responses are due simply to chance. However, in hypothesis testing, biologically important adverse effects may not be statistically significant due to high variability, small sample sizes, or other statistical vagaries. Biologically trivial effects might be statistically significant if variation is low or sample

[1] The complete definitions of chronic limits were, "A chronic value may be obtained by calculating the geometric mean of the lower and upper chronic limits from a chronic test or by analyzing chronic data using regression analysis. A lower chronic limit is the highest tested concentration (a) in an acceptable chronic test, (b) which did not cause an unacceptable amount of adverse effect on any specified biological measurements, and (c) below which no tested concentration caused an unacceptable effect. An upper chronic limit is the lowest tested concentration (a) in an acceptable chronic test, (b) which did cause an unacceptable amount of adverse effect on any specified biological measurements, and (c) above which all tested concentration also caused such an effect." (Stephan and others 1985, p. 39)

sizes are large. Because chronic toxicity tests commonly use about two to five sample replicates, the problem of important adverse effects not being statistically significant probably is the more likely of the two outcomes in most tests. Cases of the statistical MATCs and NOECs corresponding to fairly high levels of adverse effect have been documented (Suter and others, 1987; Crane and Newman, 2000). Stephan and others (1985) were aware of these limitations and advised that

> *"...the amount of effect that is considered unacceptable is often based on a statistical hypothesis test, but might also be defined in terms of a specified percent reduction from the controls. A small percent reduction (e.g., 3 percent) might be considered acceptable even if it is statistically significantly different from the control, whereas a large percent reduction (e.g., 30 percent) might be considered unacceptable even if it is not statistically significant)."*

However, beyond this example, Stephan and others (1985) made no further recommendations for specifying what percent reductions from the controls would be considered acceptable or unacceptable. Few recommendations from the refereed literature were helpful to define what level of effect *x* for the effects concentration adversely affecting *p* percentage of a test population (EC*p*) is "biologically acceptable" for biological communities. Published views referring to a level of effect *p* for the EC*p* considered biologically acceptable ranged from 5 to 25 percent, and were based on professional judgments (Bruce and Versteeg, 1992; de Bruin and Hof, 1997; van der Hoeven and others, 1997).

For this report, chronic values were obtained from reported chronic tests as follows. Generally, chronic values were calculated as MATCs, the geometric mean of the NOECs and LOECs. If the percentage of reductions in survival, growth, reproduction, etc., associated with the NOECs and LOECs were readily apparent in the source, reductions much greater than about 10 percent in tests with fish could be "unacceptable" and reductions much greater than about 20 percent in tests with invertebrates could be "unacceptable." If MATCs were greatly different from these EC*p* percent reductions, then a test statistic was used that was closer to the respective EC_{10} or EC_{20} value. For some influential tests, EC*x* values were calculated from the original data if sufficient data were presented. In many instances, insufficient data were presented to calculate EC*p* values.

For fish, this choice of 10 percent as the estimate of the upper bound of an "acceptable effect" was supported by modeled extrapolations of early-life stage mortality rates to wild fish populations. In several scenarios when reductions occurred in fish populations that were reasonably stable, habitats were intact, and if environmental conditions were not otherwise severe, reductions of about 20 percent (EC_{20}) in growth or first year survival likely would be sustainable. However, in more vulnerable populations, or in populations subject to other stressors, reductions of 10 percent (EC_{10}) or less for growth or mortality endpoints would be a better estimate of an acceptable low-effects threshold (Barnthouse and others, 1990; Boreman, 1997; Paul and others, 2003; Spromberg and Meador, 2005). For invertebrates, the assumption that endpoint reductions much greater than 20 percent could have "unacceptable" adverse effects on populations follows modeled extrapolations of mortality rates. For example, Kuhn and others (2002) predicted amphipod population growth rates will begin to decline dramatically when acute mortality (measured in 10-day tests) starts to increase above 20 percent. Concentrations causing invertebrate acute mortality rates on the order of 50 percent have resulted in extinctions in longer term mesocosm tests (Stark, 2005). These examples suggest that reductions much more than about 20 percent would not be sustainable in at least some invertebrate populations. Although these selected examples are not intended to represent the rich literature on extrapolating effects from the individual to population, they do give some support to the assumptions that reductions on the order of 10 or 20 percent for fish and invertebrates, respectively, approximate a divide between effects that are likely acceptable and unacceptable.

The assumption that many invertebrate populations could tolerate greater percentage of reductions than fish populations is supported by shorter recovery times from disturbances for invertebrates compared to fish. For a given toxicological sensitivity to a contaminant at a given level of effect, species that are less able to recover from disturbance are more vulnerable and are at increased risk of extinction. Aquatic insect populations conceptually would be less vulnerable to contaminant-induced mortality than more long-lived fish populations. Aquatic insects commonly have shorter life cycles than fish, high dispersal abilities, and generally high reproductive potential. Long-lived large, invertebrates such as some molluscs are exceptions to these generalizations. Empirical and predicted comparisons of recoveries from disturbances for many fish and invertebrate populations or assemblages found that invertebrates usually recover from disturbances faster than fish (Niemi and others, 1990; Detenbeck and others, 1992; Barnthouse, 2004).

Of the data reviewed for this report, EC*p* values were calculated and compared with statistical no- and lowest-observed effect concentrations for 19 tests with invertebrates and 15 tests with fish. These data showed reasonably good agreement between the MATC values and the EC_{20} and EC_{10} values for invertebrates and fish, respectively. Of the invertebrate values, the average MATCs and NOECs were greater than corresponding EC_{20} concentrations (28 and 16 percent greater, respectively). With fish the average NOEC values were 11 percent lower than the corresponding EC_{10} values, and average MATCs were 16 percent greater than EC_{10} values. This comparison suggests that for cadmium, the typical expected adverse effect level associated with a MATC probably is about 20–30 percent for invertebrates, and about 10–15 percent for fish.

Diversity of Data

About 279 acute values were located in datasets, representing 69 species, 58 genera, and 33 families. About 93 chronic values were located in datasets, representing 28 species, 21 genera, and 17 families. Although these data are limited in diversity compared to the diversity of aquatic life in North America (>10,000 animal species, table 2), these datasets are more diverse than those available for many other chemicals. For example, Forbes and Calow (2002) reviewed published chronic SSD-based datasets consisting of 5–25 species, and 12–64 species for acute datasets. The minimum diversity guidelines for the chronic dataset are exceeded for fish and planktonic crustaceans, but are only just met for other major taxonomic groups such as insects and molluscs (table 2). If in fact, the chronic dataset represented a random sample from all fish and invertebrate species available, then about 50 percent of the test values would be for insects, 90 percent for all invertebrates, and less than 10 percent for fish. Instead, only 4 percent of the chronic values were for insects, 32 percent for all invertebrates, and 60 percent were for fish. Among the fish, the richest data were for salmonids, which probably is because of their social importance and their sensitivity. The larger acute dataset meets or exceeds the minimum diversity guidelines for all groups (not shown).

Some variations in taxonomic nomenclature and spelling of common and scientific names of organisms were found in the literature review. For this report, the Integrated Taxonomic Information System (2002) was used as the taxonomic authority for scientific and common names of taxa other than fishes. For fishes, Nelson and others (2004) was used except for *Oncorhynchus mykiss*. The common name for *Oncorhynchus mykiss* used in this report is “rainbow trout” regardless if the original study referred to specific forms such as redband trout, golden trout, or the sea-run steelhead.

Acute Toxicity of Cadmium to Freshwater Animals

Acute values for Daphnids, for example, *Daphnia* sp. and *Ceriodaphnia* sp., were based on 48-hour LC_{50}s, all other acute values were 96-hour LC_{50}s. Using pooled hardness-acute toxicity relations, acute values were adjusted to a hardness of 50 mg/L to make the responses more comparable. Estimated species mean acute values (SMAVs) were calculated as the geometric means of the hardness adjusted individual test values for a species and the estimated genus mean acute values (GMAVs) were calculated from the geometric means of the SMAVs (table 15).

The extreme variability of *Daphnia magna* acute values, which even after normalization vary by more than 100 times, is particularly noteworthy. Even among replicated tests in the same experiment acute *Daphnia* values varied by more than a factor of 8 (Nebeker and others, 1986a). In cases where the available data for a species vary by greater than 10 times, Stephan and others (1985) recommend not using some or all values in species mean or genus mean values. However, all data were used based on the rationale that this variability, which is primarily genetic, in part reflects that occurring in field populations. Different genotypes or clones of this species have been shown to have high variability in sensitivity. This variability is assumed to be both less than but at least partly reflect that found in field populations (Baird and others, 1990, 1991; Baird, 1992; Forbes and Depledge, 1992, 1993; Forbes and Forbes, 1993b; Barata and others, 1998; Goulden, 1999; Barata and others, 2000; Barata and others, 2002a). Barata and others (2002a) concluded that laboratory selection favors individuals with high fitness or reproductive performance under optimal laboratory conditions resulting in laboratory populations with similar or lower tolerance to toxic stress than their original field populations. Given that populations can exhibit high levels of genetic variability in tolerance to toxic stress, minimizing genetic diversity in toxicity tests will increase the uncertainty attendant in extrapolating from the lab to the field.

Overall, available data indicate that salmonids, sculpins, a darter, and the amphipod Hyalella are distinctly more acutely sensitive to cadmium than other taxa. When rank-ordered by genus mean acute values, the four most sensitive genera were *Oncorhynchus, Salvelinus, Salmo*, and *Cottus* (table 7).

Chronic Toxicity of Cadmium to Freshwater Animals

The most consistently sensitive species was the amphipod *Hyalella azteca* with seven hardness-adjusted values ranging from 0.14 to 0.51 µg/L. The second most sensitive genus value also was for an amphipod, *Gammarus fasciatus*, followed by sculpin, *Cottus*, and the trout *Salvelinus* (table 7).

As with acute data, *Daphnia magna* values were noteworthy for their variability, varying by over a factor of 20. As with the acute data, this variability probably at least partly reflects that found in field populations. Therefore, no values were discarded solely based on variability. Methodological or test condition differences likely contributed to the variability in the results. For example, the three lowest chronic values for *Daphnia magna* (0.13–0.18 µg/L, hardness adjusted) were from a study by Chapman and others (1980). Winner (1986) suggested these results may have been biased low because hardness was adjusted by diluting test-water with deionized water and deionized waters may contain chemicals leached from the deionized resin, which was very toxic to Daphnids. Diet may have played some role because the animals used by Chapman and others (1980) were fed a trout-chow diet and *Daphnia magna* fed a trout-chow diet were much more sensitive to copper than those fed a vitamin enriched algal diet (Winner, 1986).

Presumably, other Daphnids or other species also might have similar genetic variability, but no similar research to that reported for *Daphnia magna* was reviewed. Thus, a chronic value obtained with *Ceriodaphnia dubia* by Jop and others (1995), that was 31 times higher than the lowest hardness adjusted value for this species was not used, even though the test appeared otherwise acceptable. The remaining seven chronic values for the genus *Ceriodaphnia* were within 7 times of each other. Within the genus *Salvelinus*, the value for lake trout was greater than 10 times the lowest value for the genus, which was obtained with bull trout. The lake trout value was not used in the GMCV calculations. The remaining five values for *Salvelinus* were all within 3 times of each other (table 16). In the case of fathead minnow, *Pimephales promelas*, data, the highest hardness-adjusted chronic value, generated from a life-cycle test, was 12 times higher than the lowest value, which was generated from a 7-day growth and survival test. In this case, these life-cycle tests were retained because of the preference for life-cycle test data in estimating species mean chronic values (Stephan and others, 1985, p. 38), despite the risk that these results possibly were biased high.

Few, if any, of the chronic datasets reviewed were ideal. Results published as journal articles were sometimes highly summarized, omitting details such as what magnitude of responses were considered no- and lowest-observed effect concentrations (for example, Eaton, 1978). In what were likely tradeoffs between replication, numbers of treatments, and test durations, some tests were unreplicated or only used two replicates, which were insufficient for statistical testing for differences between treatments using ANOVA (Benoit and others, 1976; Sauter and others, 1976; Davies and others, 1993).

Other Data on the Effects of Cadmium to Freshwater Animals.

“Other data” that for various reasons were not directly used to develop criteria, but were still pertinent were evaluated and compared to the acute and chronic criteria values (table 17). Stephan and others (1985) considered test endpoints relating to mortality, growth, or reproduction to be biologically important and they specifically called for using at least these endpoints in criteria derivation. Although nothing was noted in their guidelines to preclude including other biological responses in criteria derivation, in practice criteria documents have focused on these endpoints. However, other pertinent data are available on adverse effects of cadmium to aquatic organisms. Numerous reports were reviewed that evaluated effects of cadmium on freshwater animals including acclimation, biochemical effects, swimming performance, predator-prey interactions, bioaccumulation and elimination, and factors affecting toxicity. Such data could, however, affect a criterion if test concentrations were measured, the endpoint was biologically important, and if the data were obtained with a biologically important species (Stephan and others, 1985).

Tests compiled in table 17 were considered pertinent for estimating risks or response thresholds for freshwater organisms to cadmium, but for some reasons it did not seem appropriate to pool with the acute and chronic data listed in tables 15 and 16. Reasons for treating these data separately included:

1. Tests with durations or test statistics that could not be directly compared with other data (for example acute exposures longer than 96-hours, incipient lethal levels reported in lieu of LC_{50s});

2. Effects were observed in potentially important biological endpoints, such as swimming impairment or reduced prey capture efficiency in salmonids, but response thresholds could not be estimated from the concentrations tested. In these cases unbounded LOECs were obtained, that is, effects were observed at the lowest treatment tested. Often, it did not seem appropriate to interpolate between the unbounded LOEC and the near-zero control NOEC concentration;

3. Tests of biochemical or physiological change that were not accompanied by organism level effects such as reduced survival or growth;

4. Tests of acclimation influences on subsequent toxicity or tests that otherwise appeared to have substantial pre-exposure to cadmium;

5. Tests that used unusual dilution waters, for which the general assumption that hardness is correlated with calcium concentrations, alkalinity, and pH would not hold;

6. Tests that used insensitive life stages; some values obtained using more sensitive life stages are shown here for contrast but also were included in table 15;

7. Data in a publication which were identical to data reviewed in another source were assumed to be the same data described in more than one publication and were used only once in tables 15 or 16;

8. Tests with questionable results, such as tests with unmeasured exposure concentrations, or that reported insufficient information to evaluate.

Some of the “other” tests summarized in table 17 indicated adverse effect concentrations lower than calculated criteria values. In unmeasured exposures, an estimated chronic value of 0.28 µg/L was obtained for the cladoceran *Moina* *macrocopa,* which was lower than the calculated chronic criterion of 0.52 µg/L at a hardness of 82 mg/L (table 17). This value is slightly higher than the lowest values used in the chronic dataset for a cladoceran (*Daphnia magna*, table 16). The foraging success of lake trout, *Salvelinus namaycush* on fingerling rainbow trout, *Oncorhynchus mykiss,* was reduced after both species were exposed to 0.5 µg/L cadmium for 8–9 months. This reduced foraging success suggests a low response threshold for cadmium-caused behavioral changes (table 17). Whether that response would be expected in more natural conditions is unclear because a population of lake trout that was exposed to cadmium through a whole-lake experiment did not show reduced size or condition factor compared to baseline or reference conditions (this report "Cadmium Whole-Lake Ecosystem Experiment" section). Another test reporting adverse effects below the calculated chronic criterion involved reduced hatching success of rainbow trout embryos after their parents were exposed to 0.2 µg/L of cadmium for 18 months (W.J. Birge and J.A. Black, unpub. data, as cited by Birge and others, 1981). However, because the information reported for this test was very limited, the results were difficult to interpret and thus the significance of this test is unclear (table 17).

An effort was made to locate data on effects of cadmium on snails because several snail species are considered endangered in Idaho and because of concerns that pulmonate (air breathing) aquatic snails are sensitive to copper and lead, and were fairly sensitive to zinc (Grosell and Brix, 2004; Grosell and others, 2006). Several studies of the responses of the common pond snail, *Lymnaea* spp., to cadmium were located, including the two chronic studies listed in table 17. The LOECs were 20–40 times higher than chronic criterion values for the test hardnesses, but these data are ambiguous because adverse effects were observed at the lowest concentrations tested.

Hardness–Toxicity Relations

Because previous investigations have found that a log-log relation fits data for acute toxicity of metals and hardness well (Stephan and others, 1985; Meyer, 1999), logarithms were used for evaluating hardness-toxicity relations for cadmium (fig. 2; tables 3 and 4).

Acute values obtained over a broad enough range of hardness values to be useful to evaluate apparent hardness-toxicity relations were available for 11 species over hardnesses ranging from 5 to 290 mg/L (fig. 2). The slopes of hardness-acute toxicity relations ranged from 0.4 to 1.5, with this relation explaining from 32 to 99 percent of the variability in species acute values When the slopes were pooled using species as a grouping variable (Zar, 1984), the overall hardness-acute toxicity slope for all species was 0.8368 (table 3).

Chronic values obtained over a broad enough range of hardness values to be useful to evaluate apparent hardness–toxicity relations were available for seven species. Data were available for hardnesses ranging from 17 to 280 mg/L (fig. 2). In contrast to the acute slopes, the slopes of hardness–chronic toxicity relations were remarkably similar between the species, and when plotted were nearly parallel (fig. 2). Individual regression slopes ranged only from 0.50 to 0.77, and for 5 of the 7 species, this simple log-linear least-squares relation explained greater than 90 percent of the variability in the data. When the slopes were pooled using species as a grouping variable (Zar, 1984), an overall hardness–chronic toxicity slope for all species of 0.6247 was obtained (table 4).

It is noteworthy that this hardness-toxicity relation accounted for greater than 90 percent of the variability in chronic test results because the data with brook trout, brown trout, rainbow trout, and *Hyalella* resulted from varying "chronic" exposure durations. Durations of the chronic tests ranged from 60 to 1,100 days for brook trout, 55 to 665 days for brown trout, 53 to 665 days for rainbow trout, and 14 to 42 days for *Hyalella azteca* (table 16).

In several cases, the data used in the regressions were censored to minimize the influence on confounding variables such as sensitivity differences from different life stages and possible differences between animal strains or test methods. Data were restricted more for inclusion in the hardness–toxicity analyses than for the overall calculations of species mean acute or chronic data. This difference was because to evaluate hardness–toxicity relations all variability from other factors is undesirable, whereas to evaluate species sensitivity rankings, obtaining the most representative data for sensitive life stages of a species probably is more important than minimizing variability so that some variability from different stocks, cultures, or other unknown factors is tolerable. Specific test values that were used in the hardness–toxicity regressions are indicated in the notes on tables 15 and 16.

For *Daphnia magna*, only data from Chapman and others (1980) were used in both the acute and chronic hardness regressions. This was the only study reviewed that explicitly tested organisms from the same culture, using the same testing facility and investigators, and conducted tests in waters with multiple (≥ 3) hardnesses. This censoring resulted in a chronic slope estimate for this species of 0.77 with a regression coefficient (R^2) of 0.96, whereas if all otherwise acceptable chronic *Daphnia magna* data were used, no hardness-toxicity relation would result (slope of 0.2, R^2 of 0.007, P=0.8 (tables 4 and 5). Similarly, to reduce variability that may have resulted from different cultures or laboratory waters, only chronic *Daphnia pulex* data generated at Miami University were used.

The overall slopes obtained for acute and chronic toxicity–hardness relations of 0.84 and 0.62 are lower than the about 1.0 slope for acute toxicity that Meyer (1999) predicted for transition metals. A possible reason for these

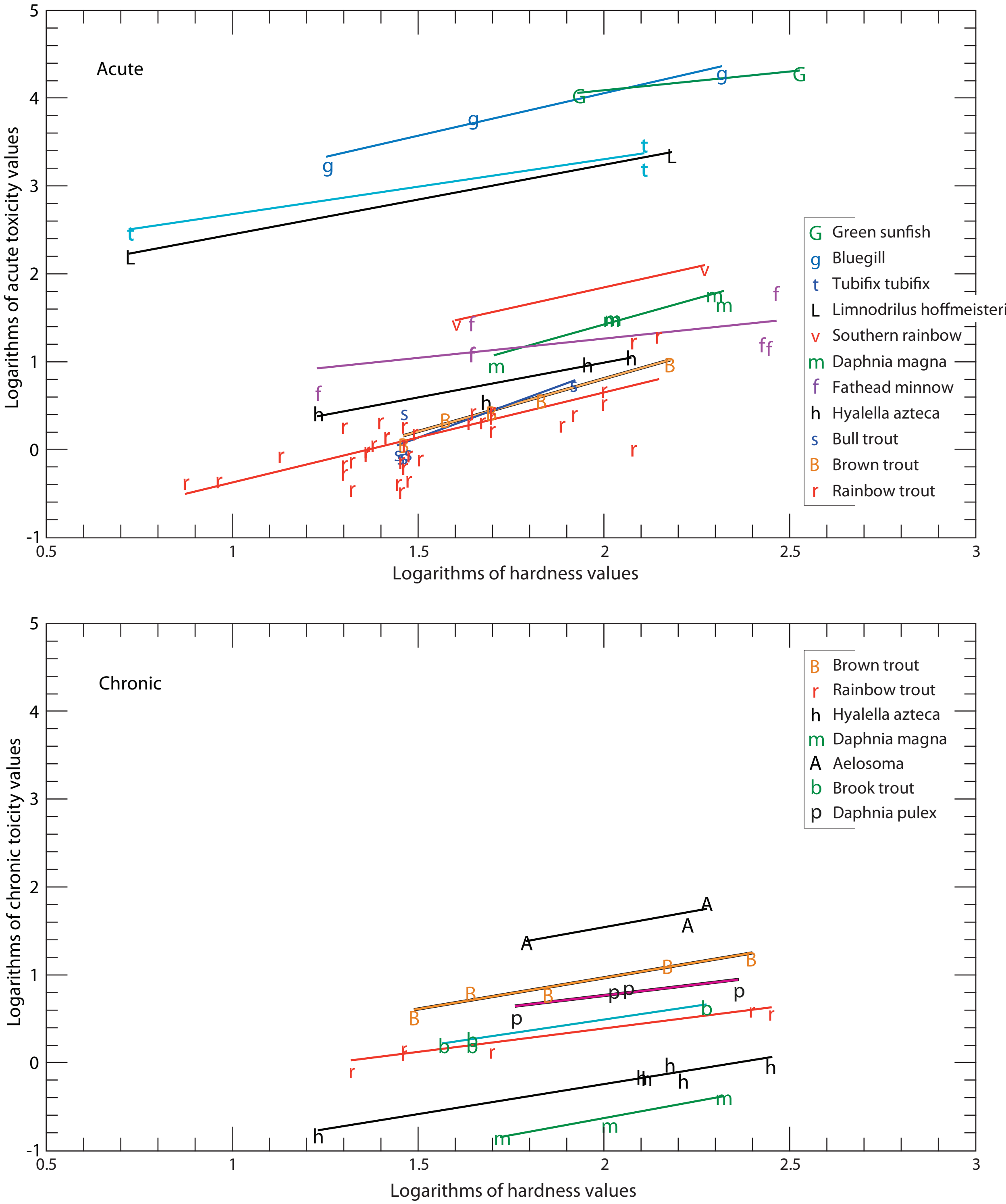


Figure 2. Hardness-toxicity relations for acute and chronic data. Data are from studies in tables 15 and 16 with an "h" in the code/notes column.

Table 3. Acute toxicity values versus hardness for individual and pooled regressions.

[**n:** number of observations. **Slope:** Slope of the relations between the natural logarithm (ln) of hardness values and ln toxicity values. $\boldsymbol{R^2}$: Coefficient of determination or fraction of the variance explained by the regression (Helsel and Hirsch, 2002). ***P*:** probability that the apparent relations between toxicity and hardness actually resulted from chance alone and that the actual slope of the linear regression is zero. Underlining emphasizes *P*-values that are significant at 0.05 or less. mg/L, milligram per liter; –, missing or inapplicable value; <, less than]

Species	n	Slope	R^2	*P* (significance)	95-percent confidence limit
Daphnia magna[1]	5	1.182	0.91	<u>0.01</u>	0.52, 1.84
Worm, *Limnodrilus hoffmeisteri*	2	.788	–	–	None
Worm, *Tubifix tubifix*	3	.6238	.93	.17	-1.56, 2.81
Southern rainbow (mussel), *Villosa vibex*	2	.9286	–	–	None
Amphipod, *Hyalella azteca*[2]	4	.7866	.89	.06	-0.07, 1.64
Fathead minnow[3]	6	.6222	.316	<u><.01</u>	0.46, 0.78
Brown trout	5	1.193	.973	<u><.01</u>	0.83, 1.55
Bull trout	6	1.548	.696	<u>.04</u>	1.29, 2.96
Rainbow trout	37	1.024	.564	<u><.01</u>	0.72, 1.33
Green sunfish	2	.4220	–	–	None
Bluegill	4	.9111	.99	<u><.01</u>	0.627, 1.19
Pooled slope for all species	76	0.8368	0.966	<0.01	0.637, 1.00

[1] Data from Chapman and others (1980).

[2] Using the mean of six values that were all generated by Collyard and others (1994) at a hardness of 90 mg/L as one value.

[3] Using only the results of acceptable tests with fathead minnow fry as shown in table 16.

Table 4. Chronic toxicity values versus hardness for individual and pooled regressions.

[**n**: sample size. **Slope**: slope of the relations between the natural logarithm (ln) of hardness values and ln toxicity values. $\boldsymbol{R^2}$: coefficient of determination or fraction of the variance explained by the regression (Helsel and Hirsch, 2002). ***P***, probability that the apparent relations between toxicity and hardness actually resulted from chance alone and that the actual slope of the linear regression is zero. Underlining emphasizes *P*-values that are significant at 0.05 or less. <, less than]

Species	n	Slope	R^2	*P* (significance)	95-percent confidence limits
Aelosoma headleyi	3	0.7429	0.7855	0.30	-4.19, 5.67
Daphnia magna[1]	3	.7712	.962	.12	-1.67, 2.71
Daphnia pulex[2]	6	.5039	.617	.21	0.348, 0.713
Hyalella azteca[3]	6	.6853	.93	<.01	0.42, 0.94
Brook trout	4	.6187	.980	.01	0.348, 0.889
Brown trout[4]	5	.6987	.926	<.01	0.335, 1.06
Rainbow trout[4]	6	.5300	.942	<.01	0.348, 0.712
Pooled slope for all species	31	0.6247	**0.991**	<0.001	0.533, 0.716

[1]Using only data from Chapman and others, (1980).

[2]Miami University data only.

[3]Excluding Borgmann and others (1991) value as anomalous.

[4]Early life stage and life cycle tests only.

Table 5. Other data on chronic toxicity values versus hardness not included in individual and pooled regressions.

[**n**: sample size. **Slope**: slope of the relations between the natural logarithm (ln) of hardness values and ln toxicity values. $\boldsymbol{R^2}$**:** coefficient of determination or fraction of the variance explained by the regression (Helsel and Hirsch, 2002) ***P***: probability that the apparent relations between toxicity and hardness actually resulted from chance alone and that the actual slope of the linear regression is zero. Underlining emphasizes *P*-values that are significant at 0.05 or less]

Species	n	Slope	R^2	*P* (significance)	95-percent confidence limits
Ceriodaphnia dubia[1]	7	0.441	0.37	0.14	-0.02, 1.10
Daphnia magna, all data	8	.206	.007	.84	-2.3, 2.7
Hyalella azteca, all data[2]	7	.657	.66	<u>.03</u>	0.11, 1.19
Fathead minnow, all data[3]	11	.627	.45	<u>.02</u>	0.15, 1.15

[1]Not included in covariance analysis, because all but one data point were derived over a narrow range of hardness values. Jop and others (1995) data point was excluded because anomalous (≈10 times higher than other values obtained at higher hardnesses).

[2]Calculated including Borgmann and others (1991) outlying data point. The Hyalella toxicity-hardness relation used in table 4 to estimate a species toxicity–hardness relation excluded this data point, based on the otherwise high degree of agreement of data within this species.

[3]Not included in covariance analysis, because several data points were derived over a narrow range of hardness values. Also suspect that pooling disparate methods contributes to variability of data.

lower slopes is that with cadmium the mitigating influence of increasing hardness on acute cadmium toxicity is offset by the corresponding aggravating influence of increasing pH on acute cadmium toxicity.

Extrapolating the calculated hardness–toxicity relations much beyond the tested range of 5 to 290 mg/L as calcium carbonate would be questionable. Working with *Daphnia magna* and zinc, which has some ecotoxicological similarities to cadmium, Heijerick and others (2002) found that at hardnesses greater than 325 mg/L as calcium carbonate, no linearity, and even a decrease in 48-hour EC_{50s} was observed. At the low end of the range, further extrapolation of the relations probably is moot because few ambient waters have hardness much less than 5 mg/L. At least in the case of salmonids, most reported cadmium toxicity during chronic tests was actually acute mortality that either occurred in the first 4–5 days in tests initiated with fry, or in the first 4–5 days after swim-up occurred in tests initiated at the more resistant egg stage (McDonald and Wood, 1993; Hollis and others, 1999, 2000b; Hansen and others, 2002b; Brinkman and Hansen, 2004a; C.A. Mebane, D.P. Hennessy, and F.S. Dillon, unpub. data, 2004). As a result, acute to chronic ratios with salmonids tend to be close to 1.0 (table 6). Therefore, at least for salmonids, it follows that the tested hardness–chronic toxicity relations probably would be relevant over the range of hardness–acute toxicity relations.

Newman (1991) presented arguments why a statistical bias encountered with the backtransformed, least-squares regression models used to develop metals criteria can compromise the accuracy of associated predictions. In datasets selected from metals criteria documents available at the time, Newman found the bias was as high as 57 percent and suggested that hardness-based metals criteria equations would benefit from additional terms to correct for this bias. Using his method of estimating this bias, the hardness adjustments for acute values were estimated to be low biased by 17 percent, and the hardness adjustments for chronic values were estimated to be low biased by less than 2 percent. Because the potential bias of these hardness-adjusted values seemed low, the potential benefit of improving model precision gained through adding additional terms to the cadmium criteria equations seemed offset by a more complex criteria equation that would have a different form than other metals criteria.

Acute to Chronic Ratios

Acute to chronic ratios (ACRs) were calculated species-by-species to provide a tool for estimating chronic effects thresholds from acute data. Species mean ACRs were estimated for 17 species using 37 pairs of acute and chronic tests (table 6). Only pairs of tests that were conducted using the same dilution water source were used. Some test pairs were excluded because the acute tests were conducted with resistant, older fish and the resulting ACRs calculated from these tests would be biased high.

When species mean ACRs seem to increase as the SMAVs increase, Stephan and others (1985) recommended that the final ACR be calculated with species that have SMAVs

Table 6. Cadmium acute-chronic ratios (ACRs), ranked in order of closeness of the species mean acute values (SMAV) to the final acute value (FAV).

[Underlined values were used to calculate the final ACR. SMACR, species mean acute-chronic ratio]

Species	Common name	Hardness	Acute value	Chronic value	ACR	SMACR	SMAV	SMAV/ FAV	Reference
Lepomis macrochirus	Bluegill	207	21,100	49.8	423.69	423.69	6,053.63	4,032.06	U.S. Environmental Protection Agency, 2001
Jordanella floridae	American flagfish	44	2,500	5.76	434.03	434.03	2,943.02	1,960.22	U.S. Environmental Protection Agency, 2001
Ambystoma gracile	Northwestern salamander	45	468.4	23.89	19.60	19.60	468.40	311.98	Nebeker and others, 1994, 1995
Ephemerella[2]	Mayfly	44	238	1.5	158.67	158.67	238	158.52	Spehar and others, 1978a
Physa integra[2]	Snail	44	238	5	47.60	47.60	238	158.52	Spehar and others, 1978a
Aplexa hypnorum[1]	Snail	45.3	93	3.26	28.53		101.01	67.28	Holcombe and others, 1984
Aplexa hypnorum	Snail	45.3	93	1.94	47.87	36.95	101.01	67.28	Holcombe and others, 1984
Daphnia pulex	Water flea	58	92	7.07	13.01	13.01	73.65	49.06	Ingersoll and Winner, 1982
Ceriodaphnia dubia	Zooplankton	44	27.3	2.2	12.41		28.35	18.88	Spehar and Fiandt, 1986
Ceriodaphnia dubia	Zooplankton	17	16.9	2	8.45	10.24	28.35	18.88	Suedel and others, 1997
Pimephales promelas	Fathead minnow	44	13.2	10	1.32		16.49	10.98	Spehar and Fiandt, 1986
Pimephales promelas	Fathead minnow	201	5,995	63.80	93.97		16.49	10.98	Pickering and Gast, 1972
Pimephales promelas[4]	Fathead minnow	201	5,995	39.23	152.82		16.49	10.98	Pickering and Gast, 1972
Pimephales promelas	Fathead minnow	17	4.8	1.41	3.39	<u>2.12</u>	16.49	10.98	Suedel and others, 1997
Daphnia magna	Water flea	53	9.9	.1523	65.00		14.08	9.38	Chapman and others, 1980
Daphnia magna	Water flea	103	33	.2117	155.91		14.08	9.38	Chapman and others, 1980
Daphnia magna	Water flea	209	49	.4371	112.09	<u>104.34</u>	14.08	9.38	Chapman and others, 1980
Hyalella azteca	Amphipod	17	2.8	.16	17.50	<u>17.50</u>	5.01	3.34	Suedel and others, 1997
Etheostoma fonticola	Fountain darter	266	8.68	7.9599	1.09		3.32	2.21	Castillo and Longley, 2001
Etheostoma fonticola	Fountain darter	257	14.23	7.953	1.79		3.32	2.21	Castillo and Longley, 2001
Etheostoma fonticola	Fountain darter	268	15.7	1.9799	7.93		3.32	2.21	Castillo and Longley, 2001
Etheostoma fonticola	Fountain darter	280	13.27	4.6615	2.85		3.32	2.21	Castillo and Longley, 2001
Etheostoma fonticola	Fountain darter	278	13.27	6.8367	1.94	<u>2.43</u>	3.32	2.21	Castillo and Longley, 2001
Oncorhynchus tshawytscha	Chinook salmon	25	1.41	1.12	1.26	<u>1.26</u>	2.67	1.78	Chapman, 1975, 1982
Cottus bairdi	Mottled sculpin	100	5.2	1.91	2.73	<u>2.73</u>	2.61	1.70	Besser and others, 2006
Salmo trutta	Brown trout	37.6	2.37	.70	3.39		2.61	1.74	Davies and Brinkman, 1994c
Salmo trutta[5]	Brown trout	29	1.23	1.892	.65		2.61	1.74	Brinkman and Hansen, 2004a
Salmo trutta[5]	Brown trout	68	3.9	1.83	2.13		2.61	1.74	Brinkman and Hansen, 2004a
Salmo trutta[5]	Brown trout	151	10.1	4.81	2.10		2.61	1.74	Brinkman and Hansen, 2004a
Salmo trutta[6]	Brown trout	29	1.23	3.52	.35		2.61	1.74	Brinkman and Hansen, 2004a
Salmo trutta[6]	Brown trout	68	3.9	6.36	.61		2.61	1.74	Brinkman and Hansen, 2004a
Salmo trutta[6]	Brown trout	151	10.1	13.56	.75	<u>1.07</u>	2.61	1.74	Brinkman and Hansen, 2004a

Table 6. Cadmium acute-chronic ratios (ACRs), ranked in order of closeness to species mean acute values (SMAV) and final acute value (FAV).—Continued

Species	Common name	Hardness	Acute value	Chronic value	ACR	SMACR	SMAV	SMAV/ FAV	Reference
Salvelinus confluentus	Bull trout	30	0.95	0.55	1.73	1.73	2.13	1.42	Hansen and others, 2002a; 2002b
Oncorhynchus mykiss	Rainbow trout	21	.84	.88	.95		2.04	1.36	C.A. Mebane, D.P. Hennessy, and F.S. Dillon, unpub. data, 2004
Oncorhynchus mykiss	Rainbow trout	29	.89	1.58	.56		2.04	1.36	C.A. Mebane, D.P. Hennessy, and F.S. Dillon, unpub. data, 2004
Oncorhynchus mykiss[3]	Rainbow trout	50	2.85	1.47	1.94		2.04	1.36	Davies and others, 1993
Oncorhynchus mykiss[3]	Rainbow trout	29	2.09	1.39	1.51	1.12	2.04	1.36	Davies and Brinkman, 1994b
Aggregate (FAV to FCV ratio)[7]	N/A	50	1.50	.4075	3.68				
Final ACR[8]					**3.47**				

[1] Acute tests were conducted with adult snails. With some species, adults are more acutely resistant to cadmium than juveniles. Thus, ACR estimates using adults might be high biased.

[2] Qualitative estimate, acute value is "greater than," chronic value is estimated from authors graphs.

[3] Using average of two acute tests conducted concurrent to chronic test using same dilution water.

[4] Acute tests conducted with adult fish which were much more resistant to cadmium than newly hatched fry. ACR estimates considered invalid.

[5] ACR calculated using juvenile growth and survival test as denominator.

[6] ACR calculated using early life stage test as denominator.

[7] Calculated as an estimate of overall ACR values, to provide a separate comparison of species-by-species ACRs. In cases where the species mean ACRs seem to increase as the SMAVs increase, Stephan and others (1985) recommends that the final ACR be a calculated species which SMAVs are close to the FAV.

[8] Geometric mean of the species mean acute-chronic ratios (SMACRs) from species for which their SMAVs were within about an order of magnitude of the FAV (actually 11 times difference, using underlined SMACR values).

close to the FAV. Species mean ACRs clearly increased as SMAVs increased ($R^2 = 0.79$, $P < 0.01$). However, selecting a cutoff for which SMAVs are suitably close to the FAV was less clear. SMAVs that were within about one order of magnitude of the FAV were considered "close" to the FAV. This resulted in a final ACR value that was close to the value obtained from simply using an acute-criterion to chronic-criterion ratio (about 3.5). Although the final ACR estimates were not used in the criteria derivation, they are presented because they might be useful for other purposes, such as site-specific modifications of criteria.

Hardness-Based Cadmium Criteria Equations

With the development of hardness-toxicity regressions, sufficient information is available to calculate hardness-dependent criterion maximum concentration (CMC or "acute criterion") and the criterion continuous concentration (CCC or "chronic criterion") equations. The criteria derivation process involved five major steps: (1) compile and review data; (2) develop hardness-toxicity regressions; (3) determine genus mean acute and chronic values (GMAVs and GMCVs) and rank in order by sensitivity; (4) calculate final acute and chronic values (FAVs and FCVs); and (5) derive the hardness-dependent CMC and CCC equations using the hardness-toxicity slopes.

The results of step 1 were shown in tables 15 and 16, step 2 in tables 3 and 4, and ranked genus mean acute and chronic values (step 3) are shown in table 7. Of these ranked values, only the lowest four with cumulative probabilities closest to 0.05 were used with equation 1 to calculate the FAV (Stephan and others, 1985; Erickson and Stephan, 1988).

Solving equation 1, the FAV was calculated as 2.451 µg/L cadmium (table 8). This calculated FAV is higher than the species mean acute values (SMAVs) for cutthroat trout, rainbow trout, and bull trout (table 7). Since these species are the Idaho state fish, a popular game fish, and a threatened species, respectively, these three species are considered "important" species as described in the "Methods" section. The calculated FAV is therefore lowered to the lowest SMAV of these important species, 1.5 µg/L (Stephan and others, 1985, p. 26).

Table 7. Ranked genus mean acute and chronic values.

[Ranked in order of least resistant to most resistant. **Abbreviations:** GMAV, genus mean acute value; GMCV; genus mean chronic value; SMAV, species mean acute value; SMCV, species mean chronic value. **Symbols:** µg/L, microgram per liter; –, missing or inapplicable values]

Acute ranking						Chronic ranking					
GMAV rank	GMAV (µg/L)	SMAV (µg/L)	Family	Species	Common name	GMCV rank	GMCV (µg/L)	SMCV (µg/L)	Family	Species	Common name
1	2.02	1.50	Salmonidae	*Oncorhynchus clarki*	Cutthroat trout	1	0.33	0.33	Hyalellidae	*Hyalella azteca*	Amphipod
1	2.02	2.07	Salmonidae	*Oncorhynchus mykiss*	Rainbow trout	2	1.00	1.00	Gammaridae	*Gammarus fasciatus*	Amphipod
1	2.02	2.67	Salmonidae	*Oncorhynchus tshawytscha*	Chinook salmon	3	1.22	1.22	Cottidae	*Cottus bairdi*	Mottled sculpin
2	2.13	2.13	Salmonidae	*Salvelinus confluentus*	Bull trout	4	1.25	2.01	Salmonidae	*Salvelinus fontinalis*	Brook trout
3	2.61	2.61	Salmonidae	*Salmo trutta*	Brown trout	4	1.25	.77	Salmonidae	*Salvelinus confluentus*	Bull trout
4	2.61	2.56	Cottidae	*Cottus bairdi*	Mottled sculpin	5	1.41	.54	Daphniidae	*Daphnia magna*	Cladoceran
4	2.61	2.67	Cottidae	*Cottus confusus*	Shorthead sculpin	5	1.41	3.70	Daphniidae	*Daphnia pulex*	Cladoceran
5	3.32	3.32	Percidae	*Etheostoma fonticola*	Fountain darter	6	1.62	.87	Salmonidae	*Salmo salar*	Atlantic salmon
6	5.39	5.39	Hyalellidae	*Hyalella azteca*	Amphipod	6	1.62	3.03	Salmonidae	*Salmo trutta*	Brown trout
7	8.29	8.29	Salmonidae	*Prosopium williamsoni*	Mountain whitefish	7	1.81	1.81	Percidae	*Etheostoma fonticola*	Fountain darter
8	16.5	16.5	Cyprinidae	*Pimephales promelas*	Fathead minnow	8	1.87	2.28	Salmonidae	*Oncorhynchus kisutch*	Coho salmon
9	22.7	22.7	Pontoporeiidae	*Diporeia sp.*	Amphipod	8	1.87	1.68	Salmonidae	*Oncorhynchus mykiss*	Rainbow trout
10	26.2	26.2	Daphniidae	*Simocephalus serrulatus*	Cladoceran	8	1.87	1.72	Salmonidae	*Oncorhynchus tshawytscha*	Chinook salmon
11	28.6	12.2	Daphniidae	*Daphnia ambigua*	Cladoceran	9	2.21	2.21	Bufonidae	*Bufo americanus*	American toad
11	28.6	22.3	Daphniidae	*Daphnia magna*	Cladoceran	10	2.68	2.68	Physidae	*Aplexa hypnorum*	Snail
11	28.6	87.1	Daphniidae	*Daphnia pulex*	Cladoceran	11	3.32	3.32	Chironomidae	*Chironomus tentans*	Midge
12	32.4	45.8	Unionidae	*Lampsilis straminea claibornensis*	Southern fatmucket (mussel)	12	3.36	2.04	Daphniidae	*Ceriodaphnia dubia*	Cladoceran
12	32.4	23.0	Unionidae	*Lampsilis teres*	Yellow sandshell (mussel)	12	3.36	5.63	Daphniidae	*Ceriodaphnia reticulata*	Cladoceran
13	32.9	32.9	Daphniidae	*Ceriodaphnia dubia*	Cladoceran	13	5.32	5.32	Cyprinodon-tidae	*Jordanella floridae*	American flagfish
14	36.9	36.9	Unionidae	*Actinonaias pectorosa*	Pheasant shell (mussel)	14	6.33	6.33	Cyprinidae	*Pimephales promelas*	Fathead minnow
15	38.8	38.8	Unionidae	*Villosa vibex*	Southern rainbow (mussel)	15	7.69	7.69	Catostomidae	*Catostomus commersoni*	White sucker
16	45.7	45.7	Unionidae	*Utterbackia imbecilis*	Paper pondshell (mussel)	16	7.97	7.97	Esocidae	*Esox lucius*	Northern pike
17	47.8	66.8	Cambaridae	*Orconectes juvenilis*	Crayfish						
17	47.8	34.2	Cambarinae	*Orconectes placidus*	Placid crayfish						

Table 7. Ranked genus mean acute and chronic values. —Continued

Acute ranking						Chronic ranking					
GMAV rank	**GMAV (µg/L)**	**SMAV (µg/L)**	**Family**	**Species**	**Common name**	**GMCV rank**	**GMCV (µg/L)**	**SMCV (µg/L)**	**Family**	**Species**	**Common name**
18	59.2	59.2	Asellidae	*Lirceus alabamae*	Isopod	17	8.00	8.00	Centrarchidae	*Micropterus dolomieui*	Smallmouth bass
19	76.7	76.7	Gammaridae	*Gammarus pseudolimnaeus*	Amphipod						
20	81.7	96.3	Hydridae	*Hydra oligactis*	Hydra	18	20.5	20.5	Centrarchidae	*Lepomis macrochirus*	Bluegill
20	81.7	63.2	Hydridae	*Hydra viridissima*	Hydra	19	23.3	23.3	Aeolosomatidae	*Aelosoma headleyi*	Polychaete worm
20	81.7	89.6	Hydridae	*Hydra vulgaris*	Hydra	20	25.5	25.5	Ambystoma	*Ambystoma gracile*	Northwestern salamander
21	101	101	Physidae	*Aplexa hypnorum*	Physid snail						
22	103	103	Heptageniidae	*Rhithrogena sp.*	Mayfly	21	26.7	26.7	Cichlidae	*Oreochromis aurea*	Blue tilapia
23	129	129	Physidae	*Physa gyrina*	Physid snail						
24	151	151	Planorbidae	*Gyraulus sp.*	Gyro snail						
25	179	179	Lumbriculidae	*Lumbriculus variegatus*	Oligochaete worm						
26	193	193	Lymnaeidae	*Lymnaea stagnalis*	Pulmonate pond snail, swamp lymnaea						
27	226	226	Hydrobiidae	*Potamopyrgus antipodarum*	New Zealand mud snail						
28	226	226	Glossiphoniidae	*Glossiphonia complanata*	Leech						
29	306	306	Baetidae	*Baetis tricaudatus*	Mayfly						
30	354	354	Salmonidae	*Coregonus clupeaformis*	Lake whitefish						
31	512	512	Ambystomatidae	*Ambystoma gracile*	Northwestern salamander						
32	616	616	Asellidae	*Asellus sp.*	Isopod						
33	716	716	Arctopsychidae	*Arctopsyche sp.*	Caddisfly						
34	722	722	Cambaridae	*Procambarus clarkii*	Red swamp crayfish						
35	832	832	Cyprinidae	*Carassius auratus*	Goldfish						
36	1,026	1,026	Tubificidae	*Limnodrilus hoffmeisteri*	Tubificid worm						
37	1,332	1,332	Tubificidae	*Tubifex tubifex*	Tubificid worm						
38	1,346	1,346	Aeolosomatidae	*Aelosoma headleyi*	Polychaete worm						
39	1,570	1,570	Tubificidae	*Branchiura sowerbyi*	Tubificid worm						
40	1,950	1,950	Cyprinidae	*Ptychocheilus oregonensis*	Northern pikeminnow						
41	2,093	2,093	Tubificidae	*Quistadrilus multisetosus*	Tubificid worm						
42	2,485	2,485	Tubificidae	*Varichaetadrilus pacificus*	Tubificid worm						
43	2,289	2,289	Tubificidae	*Spirosperma sp. (“ferox”)*	Tubificid worm						

Table 7. Ranked genus mean acute and chronic values. —Continued

Acute ranking						Chronic ranking					
GMAV rank	**GMAV (µg/L)**	**SMAV (µg/L)**	**Family**	**Species**	**Common name**	**GMCV rank**	**GMCV (µg/L)**	**SMCV (µg/L)**	**Family**	**Species**	**Common name**
43	2,595	2,943	Tubificidae	*Spirosperma nikolskyi*	Tubificid worm						
44	2,610	2,610	Catostomidae	*Catostomous commersoni*	White sucker						
45	2,617	2,617	Poeciliidae	*Poecilia reticulata*	Guppy						
46	2,782	2,782	Cyprinodontidae	*Jordanella floridae*	American flagfish						
47	3,597	3,597	Lumbriculidae	*Stylodrilus heringlianus*	Oligochaete worm						
48	3,913	3,913	Percidae	*Perca flavescens*	Yellow perch						
49	4,120	4,120	Tubificidae	*Rhyacodrilus montana*	Tubificid worm						
50	4,226	4,226	Cyprinidae	*Cyprinella lutrensis*	Red shiner						
51	4,986	4,986	Ictaluridae	*Ictalurus punctatus*	Channel catfish						
52	4,995	4,995	Poeciliidae	*Gambusia affinis*	Mosquitofish						
53	5,791	5,540	Centrarchidae	*Lepomis cyanellus*	Green sunfish						
53	5,791	6,054	Centrarchidae	*Lepomis macrochirus*	Bluegill						
54	7,866	7,866	Perlodidae	*Perlodidae*	Stonefly						
55	11,742	6,988	Chironomidae	Chironomus riparius	Midge						
55	11,742	19,730	Chironomidae	Chironomus tentans	Midge						
56	12,328	12,328	Gasterosteidae	*Gasterosteus aculeatus*	Threespine stickleback						
57	15,540	15,540	Dendrocoelidae	*Dendrocoelum lacteum*	Planarian						
Acute values:						Chronic data:					
Number of species:			69			Number of species:		28			
Number of genera:			57			Number of genera:		21			
Number of families:			33			Number of families:		17			

Table 8. Acute and chronic criteria calculations.

[See equation 1. **Abbreviations:** GMAV, genus mean acute value; SQR, (square root); n, number; FAV, final acute value; FCV, final chronic value. µg/L, microgram per liter]

Solution to the FAV equation using a column and row spreadsheet matrix format
(Syntax shown is for the Excel® spreadsheet program (Microsoft Corporation, Redmond, Wash.)

	A	B	C	D	E	F	G
1	**Genus**	**Rank**	**GMAV**	**Cumulative Probability (*P*=*R*/(n+1)**	**ln GMAV**	**(ln GMAV)^2**	**SQRT *P***
2	*Cottus*	4	2.610	=B2/(B$6+1)	=LN(C2)	=POWER(E2,2)	=SQRT(F2)
3	*Salmo*	3	2.610	=B3/(B$6+1)	=LN(C3)	=POWER(E3,2)	=SQRT(F3)
4	*Salvelinus*	2	2.126	=B4/(B$6+1)	=LN(C4)	=POWER(E4,2)	=SQRT(F4)
5	*Oncorhynchus*	1	2.019	=B5/(B$6+1)	=LN(C5)	=POWER(E5,2)	=SQRT(F5)
6	n=	55					
7	S=	=SQRT(C7)	=(F7-E7)/ (D7-G7)	=SUM(D2:D5)	=POWER(SUM (E2 E5),2)/4	=SUM(F2:F5)	=(POWER(SUM (G2:G5),2))/4
8	L=	=(SUM(E2:E5) -C8)/4	=(B7*SUM (G2:G5))				
9	A=	=(B7*SQRT (0.05))+B8					
10	FAV=	=EXP(B9)					
11	CMC=	B10/2					

Final acute value (FAV) equation results

Rank	Genera	GMAV (µg/L)	Cumulative probability (*P*=*R*/(*n*+1)	ln GMAV	(ln GMAV)^2	SQRT P
4	*Cottus* (sculpin)	2.610	0.07	0.9595	0.9206	0.2626
3	*Salmo* (Atlantic trout and salmon)	2.610	.06	.9593	.9203	.2357
2	*Salvelinus* (char)	2.126	.04	.7542	.5688	.1925
1	*Oncorhynchus* (Pacific trout and salmon)	2.019	.02	.7003	.4935	.1361
Sum:			0.19	3.3908	2.934	0.8365

Number of GMAVs = 57; S = 2.374; L = 0.3654; A = 0.8964.
FAV (calculated) = 2.451 µg/L.
Calculated FAV lowered to protect cutthroat trout, bull trout and rainbow trout = 1.50 µg/L.
(Lowered to the lowest SMAV for these three “important” species which have SMAVs lower than the FAV (Stephan and others, 1985, p. 26).
CMC = FAV/2 = 0.75 µg/L

Final chronic value (FCV)

Rank	Genera	GMAV (µg/L)	Cumulative probability (*P*=*R*/(*n*+1)	ln GMAV	(ln GMAV)^2	SQRT P
4	*Salvelinus* (char)	1.281	0.18	0.2476	0.06120	0.4264
3	*Cottus* (sculpin)	1.219	.14	.1978	.03911	.3692
2	*Gammarus* (amphipod)	1.004	.09	3.501E-3	1.226E-05	.3015
1	*Hyalella* (amphipod)	.326	.05	-1.124	1.263	.2132
Sum:			0.4545	-0.6711	1.364	1.310

Number of GMCVs = 21; S = 7.033; L = -2.473; A = 0.900.
FCV = 0.4065 µg/L.

The FCV was calculated similarly using equation 2 as shown in table 8. The species mean chronic value (SMCV) for one species, *Hyalella azteca*, was lower than the FCV (tables 7 and 8). Unlike the FAV derivation, the FCV was not lowered to explicitly protect *Hyalella azteca* because *Hyalella azteca* was not obviously an important or critical species as defined in the "Methods" section. Whether the toxicity of cadmium to this species at FCV concentrations would likely result in indirect effects to other species or alter benthic assemblages such that biological integrity could be compromised are considered in more detail in the section "Risks of the 5th Percentile Species-Sensitivity Distribution (SSD)-Based Chronic Criterion to a More Sensitive Species."

The CMC is the FAV/2 (table 8). The hardness-based CMC equation has two components, the slope and the criterion maximum intercept (CMI). The slope is the same as that derived in step 2 (table 3). The intercept was calculated as a function of the natural log (ln) of the hardness-specified CMC and the slope. The intercept equation is:

$$\ln(\text{CMI}) = \ln(\text{hardness-specified CMC}) - (\text{slope} \times \ln(\text{specified hardness}))$$

Once the intercept is determined, the CMC equation is:

$$\text{CMC at hardness } X = e((\text{slope} \times \ln(X)) + \ln(\text{CMI}))$$

Using the CMC and hardness slope from tables 3 and 8, the hardness-based CMC equation is:

$$\text{CMC } (\mu\text{g/L dissolved cadmium}) = e^{(0.83675 \times \ln(\text{hardness}) - 3.5602)}$$

where the (ln hardness) is the natural logarithm of the water hardness.

The CCC hardness-dependent equation was similarly derived. Two differences between the CMC and CCC derivation were that first, unlike the FAV, the FCV was not divided by two because the chronic values used to develop the FCV reflect less severe effects than 50 percent mortality. Second, because of the belief that dissolved metals (the fraction of metals that pass a 0.45-µm filter) better predict toxicity than total metals concentrations from unfiltered samples, and because most chronic values used to derive the CCC were derived with unfiltered samples, an estimated total-to-dissolved cadmium conversion factor is used with the CCC. The chronic conversion factor (CF) used was

$$\text{CF} = 1.101672 - ((\ln \text{hardness}) \times 0.041838)),$$

which at a hardness of 50 mg/L equals 0.94 (Stephan, 1995). No conversion for the CMC was needed since the FAV was based tests that reported dissolved values. The CCC hardness-dependent equation is

$$\text{CCC } (\mu\text{g/L dissolved cadmium}) = (e^{(0.6247 \times \ln(\text{hardness}) - 3.344)}) \times (1.101672 - ((\ln \text{hardness}) \times 0.041838)).$$

These acute and chronic cadmium criteria values are compared to previous cadmium criteria derivations in table 9.

$$\begin{aligned} &P = R/(n+1) \\ &S = \sqrt{\frac{\sum((\ln GMAV)^2 - ((\sum(\ln GMAV))^2/4)}{\sum(P) - ((\sum(\sqrt{P}))^2/4)}} \\ &L = \left(\sum(\ln GMAV) - S\left(\sum\left(\sqrt{P}\right)\right)\right)/4 \\ &A = S(\sqrt{0.05}) + L \\ &FAV = e^A \end{aligned} \quad , \qquad (1)$$

where

- n is number of genera with Genus Mean Acute Values (*GMAV*s),
- R is relative sensitivity rank of each *GMAV*,
- P is cumulative probability for each *GMAV*.

$$\begin{aligned} &P = R/(n+1) \\ &S = \sqrt{\frac{\sum((\ln GMCV)^2 - ((\sum(\ln GMCV))^2/4)}{\sum(P) - ((\sum(\sqrt{P}))^2/4)}} \\ &L = \left(\sum(\ln GMCV) - S\left(\sum\left(\sqrt{P}\right)\right)\right)/4 \\ &A = S(\sqrt{0.05}) + L \\ &FCV = e^A \end{aligned} \quad , \qquad (2)$$

where

- n is number of genera with Genus Mean Chronic Values (GMCVs),
- R is relative sensitivity rank or each GMCV, and
- P is cumulative probability for each GMCV.

Table 9. Comparison of acute and chronic cadmium criteria values derived in this report to previously reported cadmium criteria.

[Criteria values all calculated for a hardness of 50 milligrams per liter; cadmium values are for total (unfiltered) concentrations. **Abbreviations:** SSD, species sensitivity distribution; Cd, cadmium; GMAV, genus mean acute value; FCV, final chronic value. **Symbols:** µg/L, microgram per liter]

Source[1]	Number of test values used	Number of genera used to calculate SSD 5th percentile value (n)	Criterion value calculated from 5th percentile of SSD	Final criterion value (µg/L total Cd)
Acute criteria derivation				
This report[1]	279	57	1.2	0.75
U.S. Environmental Protection Agency, 2001[2]	226	55	1.4	1.0
U.S. Environmental Protection Agency, 1984[3]	133	44	4.4	1.8
Chronic criteria derivation				
This report	93	21	0.40	0.40
U.S. Environmental Protection Agency, 2001	34	16	.17	.17
U.S. Environmental Protection Agency, 1984[4]	25	44	.66	.66

[1]The final acute criterion value was lowered to protect cutthroat trout, rainbow trout and bull trout, important species with species mean acute values lower than the 5th percentile of the SSD of GMAVs.

[2]The final acute criterion value was lowered to protect rainbow trout and bull trout.

[3]The final acute criterion value was lowered to protect rainbow trout.

[4]The actual number of genera in dataset was 13, not 44. However, because the 13 mean genus chronic values contained values for 5 of the 6 genera most acutely sensitive to cadmium, it was judged more appropriate to calculate the FCV using 44 genera instead of 13 genera. The effect of this judgment was to raise the FCV from 0.04 µg/L to 0.66 µg/L cadmium (U.S. Environmental Protection Agency, 1984, p. 10, 46).

Aquatic Toxicology of Cadmium—Synthesis

Mechanisms and Factors Affecting Aquatic Toxicology of Cadmium

Waterborne cadmium can cause severe, acute toxicological and physiological effects to aquatic organisms. At acute waterborne concentrations, cadmium severely disrupts calcium homeostasis, which ultimately leads to death. At the gill surface, cadmium competes with calcium for high affinity calcium-binding sites and once it enters the chloride cell irreversibly blocks calcium uptake. The cumulative effect of these two processes causes acute hypocalcaemia in freshwater fish (Wood and others, 1997). However, these effects can be altered by the calcium content of the water. Among various water-quality parameters that often influence the uptake and toxicity of metals, calcium seems to be of primary importance, with pH of secondary importance. Other water-quality parameters that influence the toxicity of other metals (such as copper) probably are less important in modifying cadmium toxicity (dissolved organic matter (DOM), magnesium, sodium, and alkalinity). This protective action of calcium has been attributed to changes in gill permeability and (or) competition between cadmium and calcium for gill-binding sites. Toxicity is reduced because calcium out-competes cadmium for binding sites on gills. Protection from cadmium uptake and acute toxicity in freshwater fish is related to calcium concentration in water rather than magnesium or alkalinity concentrations indicating that calcium is the primary cation responsible for the protective action of hard water (Carroll and others, 1979; Davies and others, 1993). In many tests, water hardness (defined as the sum of calcium and magnesium concentrations, expressed as $CaCO_3$ equivalents) is often reported in lieu of calcium concentrations. Because calcium is the principle cation of hardness in most natural waters, hardness is often a reliable surrogate measure of calcium (Hem, 1992). Exceptions to this generalization include some synthetic laboratory test waters or atypical natural waters, where the presumption that hardness is a good surrogate for the protective effects of calcium would not hold (Welsh and others, 2000).

Moreover, the bioavailability and toxicity of cadmium are less affected by DOM than copper because the binding affinity of cadmium to the gill is stronger than the binding affinity of cadmium to DOM (Winner, 1984; Playle and others, 1993; Playle, 2004). DOM from some natural sources made cadmium more toxic to fish, whereas copper and lead were less toxic in the presence of DOM (Schwartz and others, 2004). Calcium is therefore a more important modifier of cadmium toxicity in fish than water DOM, which plays an important role in modifying the toxicity of some metals, such as copper and lead. Water pH also modifies cadmium toxicity, with increasing cadmium toxicity with increasing pH (Campbell and Stokes, 1985; Cusimano and others, 1986; Schubauer-Berigan and others, 1993; Hansen and others, 2002a). Because in natural and synthetic waters where pH is not artificially manipulated water hardness is usually correlated with pH, water hardness probably is a good surrogate for the specific factors affecting cadmium bioavailability and toxicity. For example, using EPA's recipes for preparing synthetic freshwaters, very soft water with a hardness of about 10–13 mg/L will have pH values ranging from about 6.4 to 6.8, and very hard water with a hardness of about 280–320 mg/L will have pH values ranging from about 8.0 to 8.4 (U.S. Environmental Protection Agency, 2002c).

Toxicity of Cadmium in Chemical Mixtures

Cadmium probably seldom occurs as a single contaminant of concern in releases to ambient waters, but commonly occurs with zinc and copper. Evaluating interactive effects of even binary mixtures is a complex and difficult topic in ecological risk assessment. The joint toxicity of chemicals is often described as synergism, antagonism, or additivity. The toxicity of a particular chemical when combined with another can remain unchanged (additive responses), increase (synergism, or more than additive responses), or decrease (antagonism, or less than additive responses) (Sorensen, 1991; Hertzberg and MacDonell, 2002; Norwood and others, 2003). Although a detailed treatment of the joint toxicity of cadmium with other chemicals is beyond the scope of this report, the following results are relevant.

Although cadmium, copper, and zinc have some toxicological similarities, the toxicity of cadmium to fish in the presence of zinc or copper seems to result in only slightly increased toxicity over that of cadmium alone. In acute tests of cadmium plus zinc mixtures with bull trout and rainbow trout, cadmium and zinc LC_{50s} were similar when tested singly or as a mixture (Hansen and others, 2002a). Similarly in chronic toxicity tests of cadmium plus zinc with the American flagfish, no- and lowest-observed-effect concentrations of cadmium and zinc were similar when tested singly or as mixtures (Spehar and others, 1978b). The acute LC_{50s} obtained with Chinook salmon from cadmium singly or with two and three metal mixtures (cadmium + copper, and cadmium + copper + zinc) were slightly lower with mixtures than with cadmium singly in some combinations of ratios. However, the differences were not statistically significant and the three metals could have been exhibiting independent action (Finlayson and Verrue, 1982).

Contradictory patterns were observed with invertebrates in toxicity testing and field experiments. In 96-hour tests with the paper pondshell mussel *Utterbackia imbecilis,* Keller and Zam (1991) found that cadmium was slightly less toxic in combination with copper, indicating that copper and cadmium were slightly antagonistic to each other. Shaw and others (2006) tested the joint toxicity of cadmium and zinc mixtures to four species of zooplankton. Mixture toxicity was governed primarily by zinc. The presence of zinc at concentrations that caused low toxicity (LC_{15}) tended to reduce cadmium toxicity. However, when the exposure was reversed to test a range of zinc concentrations in the presence of cadmium at its LC_{15}, no interaction was indicated between the metals (no increase or decrease in zinc toxicity). Attar and Maly (1982) also noticed that cadmium and zinc mixtures were less toxic than expected. In contrast, in stream microcosm and field experiments, Clements (2004) observed that macroinvertebrate responses to a mixture of copper, cadmium, and zinc were greater than to cadmium plus zinc mixtures or zinc alone.

In urban or industrial settings, metals and polynuclear aromatic hydrocarbons often co-occur in runoff. Gust (2006) investigated the joint toxicity of cadmium and phenanthrene to the amphipod *Hyalella azteca* in various waterborne and sediment exposures. The mixtures could cause synergistic or independent toxicity in *Hyalella* depending on the endpoint investigated and the experimental protocol employed. Adding sublethal phenanthrene concentrations to sediment decreased cadmium LC_{50s}, that is, increased the risk of cadmium lethality to *Hyalella*. Yet adding sublethal phenanthrene concentrations to lower sublethal cadmium concentrations had no effect on sublethal cadmium effects thresholds to *Hyalella*. In acute waterborne exposures, the presence of sublethal phenanthrene concentrations decreased the toxicity of cadmium.

Generally, these examples indicated that the toxicity of cadmium is little changed in the presence of other metals. In a more comprehensive review of metals mixture toxicity studies, Norwood and others (2003) reported a tendency toward "less than additive" results in metals mixture testing. However, in a minority of published studies of the joint toxicity of cadmium in binary mixtures (15 of 53 or 28 percent), investigators reported "greater than additive effects" (synergism). This suggests that in situ assessments of toxicity may be prudent if joint toxicity of metals mixtures are of concern, such as benthic community monitoring or laboratory testing of field-collected water.

Relative Sensitivity of Chronic Test Endpoints

Some generalizations about the relative sensitivity of different chronic endpoints in different species are supported by the data reviewed. In chronic tests with salmonids, the principle adverse effect was most often acute mortality. In several chronic tests with cadmium and salmonids, the onset of increased mortalities began shortly after the newly hatched fish reached swim-up stage, with mortalities then ceasing or greatly diminishing by about 4–5 days after their onset. The fish that survived this acute phase usually survived in the test concentrations for several more weeks until the tests ended with subtle if any apparent adverse effects (Hollis and others, 1999, 2000b; Hansen and others, 2002b; C.A. Mebane, D.P. Hennessy, and F.S. Dillon, unpub. data, 2004). Similarly, in 3 pairs of 4- and 30-day exposures of brown trout to cadmium, Brinkman and Hansen (2004a) obtained similar LC_{50s} from the 4- or 30-day tests (table 17). In contrast to the similarity in acute and chronic sensitivities of salmonids, 28-day exposures of newly hatched mottled sculpin produced more sensitive results than did 4-day exposures (tables 15 and 16). Growth of juveniles seldom was the most sensitive endpoint with cadmium exposures in tests reviewed with fish, and in few tests with invertebrates. Reproduction often was the most sensitive endpoint in tests with invertebrates.

Bioaccumulation and Effects of Dietary Exposures to Cadmium

Most analyses presented in this report focus on relating observed effects in organisms to concentrations of cadmium in water to estimate thresholds of acceptable risk for those organisms. However, this approach is less direct than relating observed effects to concentrations of cadmium in the target tissues of the organism because a contaminant cannot be toxic to an animal if it is not accumulated in its tissues. For example, an acute LC_{50} or chronic LOEC water concentration is simply a surrogate for the amount of the toxicant in the organism at the site(s) of toxic action producing the observed mortality. Additionally, organisms may be exposed to trace metals such as cadmium through food or sediments even when cadmium concentrations in the water column are low. Thus the consequences of elevated tissue residues or effects of dietary exposures may be important when estimating protective thresholds for cadmium and other pollutants (McCarty and Mackay, 1993; Meyer and others, 2005). At least on a whole-body bioaccumulation basis, for some organisms cadmium uptake from prey is as important or more important than uptake from water. Stephenson and Turner (1993) examined the relation between dietary uptake of cadmium from periphyton and bioaccumulation in *Hyalella azteca* in a cadmium contaminated lake and a control lake. Dietary sources accounted for about 58 percent of cadmium accumulated. To determine the relative importance of food and water as cadmium sources for benthic insects, Roy and Hare (1999) measured cadmium accumulation by the predatory alderfly *Sialis velata* (Megaloptera) from either water alone or from chironomid prey. There, cadmium uptake from prey was far more important than that from water (Roy and Hare, 1999).

Further, if a maximum acceptable dietary intake or tissue concentration for fish or other aquatic-dependent life such as amphibians could be demonstrated based on observations of survival, growth, or reproduction in a chronic feeding study or a long-term field study, a "final residue value" criterion based upon tissue residues and bioaccumulation or bioconcentration factors could be defined (Stephan and others, 1985). More generally, the term "tissue reference value" is sometimes used for tissue residue concentrations above which adverse effects in aquatic biota may occur. The following data summary highlights selected studies that were relevant to the discussion of toxicity associated with cadmium tissue residues and were relevant to coldwater food chains (table 10). See Jarvinen and Ankley (1999) and Bridges and Lutz (2005) for more encyclopedic summaries.

Fish

At least for fish, waterborne cadmium accumulation on gill surfaces appears to be a primary site of toxic action. Effects associated with gill residues are emphasized here, because gill tissue was suggested as the optimum body tissue to predict cadmium toxicity and tissue-residue relation for fish (Jarvinen and Ankley, 1999). The highest concentrations of cadmium in fish usually occurs in the kidney, however it is not clear whether cadmium residues in the kidney are bioactive or if the residues are simply sequestered in the kidney. Cadmium residues in kidneys appear to be permanent and remain elevated long after the exposures to elevated cadmium from water or food end.

Several examples of cadmium tissue residues that corresponded with the absence or presence of observed adverse effects are listed in table 10. There was considerable overlap in the concentrations associated with the absence or presence of observed adverse effects for a given tissue type. Rainbow trout were able to tolerate cadmium in gill tissues of as much as 38 mg/kg dry weight (dw) without noticeable adverse effects, yet complete mortality to brook trout occurred in a study in which gill residues were only about 5 mg/kg dw. In contrast, brook trout living in a cadmium polluted river in Idaho carried an average gill tissue burden of 127 mg/kg dw. This 25 times difference between lethal and non-lethal gill residues for the same species suggests that it may be difficult to predict cadmium toxicity tissue-residue relations for fish.

Most studies with fish indicated that dietary cadmium exposures at environmentally realistic concentrations resulted in bioaccumulation but no appreciable adverse effects. Only when exorbitant concentrations of dietary cadmium were administered were obvious adverse effects observed. Scott and others (2003) observed that rainbow trout fed only 3 mg/kg through diet showed inhibition of cortisol elevation (a stress hormone) following exposure to alarm chemicals. However, the significance of this hormonal change is unclear because no behavioral changes were associated with the inhibition of cortisol elevation due to dietary cadmium exposures, unlike waterborne exposures which resulted in similar cortisol inhibition and similar whole-body cadmium residues (Scott and others, 2003).

Table 10. Examples of cadmium tissue residues co-occurring with the absence or presence of adverse effects.

[**Abbreviations:** NOEC, no observed effect concentration; LOEC, lowest observed effect concentration; Cd, cadmium. **Symbols:** mg/kg, milligram per kilogram; L/kg, liter per kilogram; mg/L, milligram per liter; µg/g, microgram per gram; µg/L, microgram per liter]

Organism	Residue concentration (mg/kg dry weight)	Tissue	Effect	Source
Aquatic communities	2.9 – 14.6	Whole body	TRV a tissue residue in aquatic biota above which adverse ecological effects may occur. Calculated as the product of the chronic water-based criterion over a hardness range of 20 – 250 mg/L (0.22 – 1.1 µg/L) and a bioconcentration factor of 2623 L/kg	This report
	0.4 – 2.0	Whole body	TSC, calculated as the product of the chronic water-based criterion over a hardness range of 20 – 250 mg/L (0.22 – 1.1 µg/L) and an accumulation factor (352 L/kg) that was adjusted to factor out background concentrations	This report
Tissue residues corresponding with the absence of apparent adverse effects (highest NOECs from each study)				
Amphipod (*Hyalella azteca*)	0.1–37	Whole body	No apparent effects; range of concentration from lakes in central Ontario, Canada with no direct pollution sources	Stephenson and Mackie, 1988
	0.32	Diet	Low reductions in growth (EC_{25}) in *Hyalella* fed a diet of Cd contaminated green algae. Corresponding Cd exposure in water producing the Cd in algae EC_{25} was about 0.57 µg/L cadmium at hardness 125 mg/L. No NOEC obtained	Ball and others, 2006
	13–62	Whole body	No effect on growth, survival, or reproduction	Borgmann and others, 1991
	1–3	Whole body	No effect; values from control treatments	Borgmann and others, 1991
Atlantic salmon (Salmo salar	~0.6	Whole body	Minimal reduction in alevin biomass, corresponding to a water LOEC of 0.47 µg/L, see table 16	Rombough and Garside, 1982
Brook trout (*Salvelinus fontinalis*)	10	Kidney	No effect	Benoit and others, 1976
	~2.5	Gill	No effect	Benoit and others, 1976
	0.5	Gill	No effect; mean value from wild fish collected from an unpolluted reference river, North Fork Coeur d'Alene River, Idaho	Farag and others, 1998
Crustacea (multiple taxa)	<0.3–53	Whole body	No apparent effects; range of concentrations from a variety of marine sites around the world with no direct pollution sources	Rainbow, 2002
Lake trout (*Salvelinus namaycush*)	20–45	Kidney	No apparent effects; concentrations following 3+ years of whole-lake Cd dosing to near chronic criterion levels	Malley, 1996
Longnose dace (*Rhinichthys cataractae*)	3–8	Gill	No effect	Riddell and others, 2005a
Mallard (*Anas platyrhynchus*)	200	Diet	No obvious deleterious effects after 13 weeks	Eisler, 1985
Mayfly (*Baetis tricaudatus*)	27	Whole body	No apparent effect on population densities	Cain and others, 2004
Mayfly (*Rhithrogena* sp.)	6–34	Whole body	No apparent effects; range of concentration from least-disturbed Colorado streams with apparently healthy fish and invertebrate communities)	Besser and others, 2001

Table 10. Examples of cadmium tissue residues co-occurring with the absence or presence of adverse effects.—Continued

Organism	Residue concentration (mg/kg dry weight)	Tissue	Effect	Source
Tissue residues corresponding with the absence of apparent adverse effects (highest NOECs from each study)–Continued				
Mayfly (*Serratella tibialis*)	35	Whole body	Apparently near the limit of tolerance since this was the highest concentration observed in *Serratella* collected along a gradient of contamination. No obvious adverse effects were associated with this concentrations	Cain and others, 2004
Rainbow trout (*Oncorhynchus mykiss*)	30	Gill	No apparent effect on population density and biomass in comparison to reference streams	Farag and others, 2003
	7.5	Gill	No effect. 99 percent of whole-body Cd was accumulated in the gills, liver, and kidney	Thomas and others, 1983
	8.5	Gill	No effect on survival or growth; residue resulting from 28-day exposure to 3 µg/L Cd at hardness of 140 mg/L	Franklin and others, 2005
	11	Gill	No effect on survival or growth; residue resulting from 36-day exposure to 2 µg/L Cd at hardness of 140 mg/L[1]	Szebedinszky and others, 2001
	19.5	Gill	Saturation concentration following 65 to100-day exposures to 3 µg/L Cd at hardness of 140 mg/L. No effects on growth, survival or swimming speed[1]	McGeer and others, 2000a, 2000b
	94	Kidney	Saturation concentration following 65 to100-day exposures to 3 µg/L Cd at hardness of 140 mg/L. No effects on growth, survival or swimming speed[1]	McGeer and others, 2000a, 2000b
	4.75	Whole body	Saturation concentration following 65 to100-day exposures to 3 µg/L Cd at hardness of 140 mg/L. No effects on growth, survival or swimming speed[1]	McGeer and others, 2000a, 2000b
	30	Diet (spiked trout chow)	No effect on survival or growth; fed Cd spiked food for 12-weeks and monitored for an additional 6-weeks	Kumada and others, 1973
	500	Diet (spiked trout chow)	No effect; after 14-days gill residues reached 5 µg/g dw from dietary exposure (no waterborne exposure)	Franklin and others, 2005
	786	Diet (spiked trout chow)	No effect on survival or growth	Szebedinszky and others, 2001
	2.2	Diet (live prey)	No effects on growth or survival. Exposed via live diet of *Lumbriculus*, an aquatic oligochaete that had been reared in Cd contaminated river sediments	Hansen and others, 2004
	55	Diet (live prey)	No significant effects on growth or survival; trout fry were fed live brine shrimp (*Artemia* sp.) enriched with Cd	Mount and others, 1994
	300	Diet (spiked trout chow)	No significant effects on growth, survival, or total body calcium accumulation	Baldisserotto and others, 2005
Snail (*Potamopyrgus antipodarum*)	7.5	Whole body (without shell)	No significant decreases in population growth rate of clones from different areas of northern Europe or New Zealand	Jensen and others, 2001
Zooplankton *(Daphnia galeata mendotae)*	28.3	Whole-body	Residue concentration associated with NOEC after exposing populations to cadmium concentrations for 154 days	Marshall, 1978

Table 10. Examples of cadmium tissue residues co-occurring with the absence or presence of adverse effects.—Continued

Organism	Residue concentration (mg/kg dry weight)	Tissue	Effect	Source
Tissue residues corresponding with the presence of adverse effects (lowest LOECs from each study)				
Amphipod (*Gammarus fossarum*)	240–320	Whole body	Reduced survival in 20-day water or dietary exposures[1]	Jarvinen and Ankley, 1999
Amphipod (*Hyalella azteca*)	0.61	Diet	Severe reductions in growth (EC_{50}) in *Hyalella* fed a diet of Cd contaminated green algae. Corresponding Cd exposure in water producing the Cd in algae EC_{25} was about 1.27 µg/L cadmium at hardness 125 mg/L. No LOEC reported	Ball and others, 2006
	30	Whole body	Critical body concentration resulting in 25 percent lethality in chronic exposures, based on regression analysis of bioaccumulation against mortality from multiple tests	Norwood and others, 2003
	28–148	Whole body	Increased mortality in 42-day exposures in tap water	Borgmann and others, 1991
	1	Whole body	Increased mortality in 42-day laboratory exposures in reconstituted hard water	Stanley and others, 2005
	25	Whole body	Increased mortality in 42-day laboratory exposures in treated sewage effluent	Stanley and others, 2005
	110	Whole body	Increased mortality in 42-day *in situ* exposures in treated sewage effluent stream mesocosms.	Stanley and others, 2005
Atlantic salmon (Salmo salar	~ 1	Whole body	Reduced alevin growth corresponding to a water LOEC of 0.78 µg/L, see table 16	Rombough and Garside, 1982
Bull trout (*Salvelinus confluentus*)	0.9	Whole body	Reduced survival	Hansen and others, 2002b
Brook trout (*Salvelinus fontinalis*)	127	Gill	Mean value from wild fish collected from a polluted river, South Fork Coeur d'Alene River, Idaho	Farag and others, 1998
	5.1	Gill	Males died during spawning	Benoit and others, 1976
	50 – 65	Kidney	Males died during spawning	Benoit and others, 1976
Fish (multiple taxa)	0.75	Whole body	5th percentile value of a large database of literature relating measured whole body tissue residues to adverse toxicological or ecological effects. Value over-predicted adverse effects to fish communities[1]	Dyer and others, 2000
Mayfly (*Baetis tricaudatus*)	10	Diet (periphyton)	Reduced growth	Irving and others, 2003
	~2.2	Whole body	Behavioral changes	Riddell and others, 2005a
Mayfly (*Stenonema* sp.)	2.2	Whole body	Decreased mayfly abundance in field study of an effluent influenced stream. Co-occurring water concentrations and body burdens of Ag, Cr, Cu and Zn were also elevated	Birge and others, 2000
Rainbow trout (*Oncorhynchus mykiss*)	2.8	Whole body	Reduced growth inferred, however description of growth effects was ambiguous. Exposed to 4.8 µg/L Cd for 30-weeks, hardness not reported	Kumada and others, 1973

Table 10. Examples of cadmium tissue residues co-occurring with the absence or presence of adverse effects.—Continued

Organism	Residue concentration (mg/kg dry weight)	Tissue	Effect	Source
Tissue residues corresponding with the presence of adverse effects (lowest LOECs from each study)—Continued				
Rainbow trout (*Oncorhynchus mykiss*)	5.6	Whole body	Reduced growth inferred, however description of growth effects was ambiguous. Exposed to 100 mg/kg Cd in food for 12-weeks	Kumada and others, 1973
	38	Gill	Reduced growth inferred, however description of growth effects was ambiguous. Exposed to 4.8 µg/L Cd for 30-weeks, hardness not reported. After holding in clean water for 10-weeks, gill residue declined 94 percent	Kumada and others, 1973
	80	Gill	Reduced population density and biomass in comparison to reference streams; gill edema and cell degeneration noted in 96-hour exposure of naïve fish in the same creek (dissolved Cd concentrations ~ 5 µg/L at hardness 60-70 mg/L.	Farag and others, 2003
	13.5	Gill	43 percent mortality. Gill residue resulting from being fed a trout chow diet spiked with 1,419 mg/kg	Szebedinszky and others, 2001
	100	Diet (presumably spiked trout chow)	Reduced growth inferred, however description of growth effects was ambiguous. Whole body Cd burden declined by 95 percent after 6-weeks of uncontaminated diet	Kumada and others, 1973
	1,407	Diet (spiked trout chow)	8 – 43 percent mortality. This dietary exposure produced similar gill residues as did 2 µg/L at hardness 140 mg/L. Average of two tests	Szebedinszky and others, 2001
	90–114	Diet (live prey)	No effects on survival, slight (5–10 percent) reductions in growth, Exposed via live diet of *Lumbriculus*, an aquatic oligochaete that had been reared in Cd spiked sediments	Erickson and others, 2003; Mount and others, 2001; D.R. Mount, U.S. Environmental Protection Agency, oral commun, 2005
	3	Diet (spiked trout chow)	Cortisol (an important hormone in stress response) response to alarm substance inhibited. No effect on several other behavioral endpoints. This dietary exposure produced similar whole-body residues as did 2 µg/L at hardness 120 mg/L	Scott and others, 2003
Snail (*Potamopyrgus antipodarum*)	14.5	Whole body (without shell)	Significant decreases in population growth rate of clones from different areas of northern Europe or New Zealand	Jensen and others, 2001
Southern leopard frog (*Rana sphenocephala*)	~10	Diet (periphyton)	Reduced survival, although differences from controls (15 percent) were not statistically significant	James and others, 2005
Zooplankton *(Daphnia magna)*	1,180	Whole-body	21-day LC_{50}	McCarty and Mackay, 1993
Zooplankton *(Daphnia galeata mendotae)*	42.8	Whole-body	Residue concentration associated with LOEC after exposing populations to cadmium concentrations for 154 days	Marshall, 1978

[1]Converted from wet weight to dry weight, assuming 80 percent moisture content (dry weight concentration = wet weight concentration divided by 0.2).

Invertebrates are important food items for many fish species, such as salmonids, sculpin, and dace. Cadmium concentrations in invertebrates were measured in several field surveys. Surveys from locations where cadmium concentrations in water and sediment were greater than background concentrations usually resulted in elevated concentrations in invertebrate tissues. Cadmium concentrations in tissues (whole organism) of aquatic insects at these locations ranged from about 5 to 50 mg/kg dw (Kiffney and Clements, 1993; Ingersoll and others, 1994; Farag and others, 1998; Besser and others, 2001; Maret and others, 2003). In this range of environmentally realistic concentrations of dietary cadmium, accumulation, but no overt adverse effects were demonstrated through studies in which cadmium-spiked food was fed to rainbow trout (table 10). In contrast, dietary cadmium toxicity tests with fish required greater than 90 mg/kg dw to even slightly reduce survival or growth. High mortalities were only observed at dietary concentrations greater than about 500 mg/kg (table 10). Reduced growth and survival rates were observed in studies with fish that were fed invertebrate diets collected from the wild that contained mixtures of elevated arsenic, cadmium, copper, lead, and zinc (for example, Woodward and others, 1994; Farag and others, 1999); however, subsequent studies implicated arsenic as the likely cause (Erickson and others, 2003; Hansen and others, 2004). Thus, the available information indicates that clearly adverse effects in fish such as high mortality or stunted growth seem to require environmentally unrealistic doses of cadmium in the diet.

Aquatic Invertebrates

As with fish, tissue concentrations associated with adverse effects in aquatic invertebrates overlapped those concentrations not associated with adverse effects. With the amphipod *Hyalella azteca*, the lowest whole-body tissue burden associated with adverse effects was 28 mg/kg; the highest whole-body tissue burden with no apparent adverse effects was 62 mg/kg. In mayflies collected from populations without obvious adverse effects, whole-body tissue residues of as much as about 35 mg/kg cadmium were measured. In contrast, mayflies collected from a stream that at a location where the benthic community was altered and was influenced by a mixture of metals in effluents had average cadmium whole-body tissue residues of only 2.2 mg/kg. Whether this 10 times overlap factor truly reflects species sensitivity differences or simply reflects the difficulty detecting effects and attributing causality to observed effects in field studies is unknown. Possibly, undetected adverse effects were present at the "no-effect" locations where mayflies with high tissue residues were collected, and at the location with apparent adverse effects and low tissue residues in mayflies, the adverse effects may have been influenced by factors other than cadmium concentrations in tissues.

Contradictory results have been reported from laboratory exposures with *Hyalella azteca*. Borgmann and others (1991, 2004) found that the chronic toxicity of cadmium to *Hyalella azteca* under differing water-quality conditions was more constant if toxicity was expressed as a function of cadmium tissue residues, rather than the cadmium concentrations measured in the water. In these studies, there was only a factor of 2 difference between the *Hyalella azteca* highest no-effect and lowest low-effect tissue concentrations. In contrast, Stanley and others (2005) reported greater than a factor of 100 difference between low-effects thresholds in three tests using differing water-quality conditions (table 10).

The high variability between whole-body cadmium tissue residues and the presence or absence of apparent effects is consistent with theoretical explanations of the biological significance of metals in tissue residues. Rainbow (2002) argued that the biological significance of tissue residues of cadmium (and other trace metals) varies greatly depending on the specific organisms and tissues involved. All aquatic invertebrates take up and accumulate trace metals, whether or not they are essential such as copper and zinc or non-essential such as cadmium. However, the significance of an accumulated concentration depends on the specific tissue and invertebrate involved and whether an accumulated trace metal concentration is high or low cannot be assessed on an absolute scale. Even in the absence of anthropogenic metal contamination and limited comparisons within the single taxon Crustacea, mean whole body cadmium tissue concentrations vary by at least a factor of 150, ranging from less than 0.3 to 53 mg/kg dw. Accumulated metal concentrations in aquatic invertebrates can be interpreted in terms of different trace metal accumulation patterns, dividing accumulated metals into two components—metabolically available metal and stored detoxified metal. Toxicity is related to a threshold concentration of metabolically available metal and not to total accumulated metal concentration. Once an invertebrate passes a threshold concentration of metabolically available metal, the invertebrate will suffer toxic effects, initially sublethal, but eventually lethal. In contrast, the amount of detoxified metal that an invertebrate can accumulate is theoretically unlimited (Rainbow, 2002).

In most short-term "chronic" laboratory toxicity tests, during most exposures (which often are much higher than most environmental availabilities) much or all of the incoming metal remains in the metabolically available pool and death will occur at about the same accumulated total body concentration, particularly if only a small proportion of total body metal is held in detoxified form. Rainbow (2002) suggested that the apparent relation between total body concentration and toxicity is a consequence of experimental design seeking toxic effects in the short-term, as opposed to field situations with indefinite exposure durations. In field situations, most accumulated metal might be expected in the detoxified state, which would contribute to the poor relations sometimes encountered between toxic effects and whole body tissue residues of trace metals in aquatic invertebrates (Rainbow, 2002).

Dietary cadmium exposures appear to be an important risk for at least some invertebrates. The data reviewed on dietary effects of cadmium to invertebrates indicated that adverse effects could occur at concentrations realistic in cadmium-polluted waters. Toxicity to mayflies from feeding on cadmium-contaminated algal mats at environmentally realistic concentrations was observed (Irving and others, 2003). Although short-term acute testing with the same mayfly species provided acute toxicity values far higher than environmentally relevant concentrations, no chronic water-only tests are known, making direct comparisons of dietary versus waterborne exposures impossible. In studies of the relative importance of water and food as cadmium uptake sources and toxicity, cadmium was more toxic to the freshwater crustacean *Daphnia magna* in water-only exposures than in combined cadmium exposures through both water and diet (Barata and others, 2002b). Barata and others (2002b) results do not appear to support generalizations about the additive effects of combined cadmium exposures via food and water.

A diet of cadmium-contaminated green algae *Chlorella* sp caused reduced growth in the amphipod *Hyalella azteca* in a recent study (Ball and others, 2006). In this study, pronounced growth reductions resulted from food contaminated with cadmium at the lowest residue concentrations reviewed (0.3 to 0.6 mg/kg dry weight, table 10). The cadmium concentrations in the medium used to produce the contaminated algae for the feeding tests were also quite low, about 0.57 and 1.27 µg/L for the contaminated diet-based EC_{25} and EC_{50} values respectively (table 10). These water concentrations overlap the chronic criterion value of 0.65 µg/L dissolved cadmium calculated at the culture hardness of 125 mg/L calcium carbonate. In this study, little or no bioaccumulation of cadmium was found in the tissues of *Hyalella azteca* that were fed contaminated food. These results demonstrate an apparent indirect or direct toxicological effect of cadmium-contaminated algae to *Hyalella azteca* that is not associated with cadmium accumulation. Indirect effects could include behavioral responses (for example, the *Hyalella azteca* detected something in the contaminated food and ate less) or decreased nutritional quality of the food. Possible direct effects could result from a negative effect on the gut lining and thus, a reduced ability to assimilate nutrients (Ball and others, 2006).

Bioconcentration and Bioaccumulation Factors

A simple approach for estimating tissue reference values (TRV) for cadmium that might result from exposure to waterborne exposure is to multiply the waterborne concentration of interest (for example, criteria value or concentration at a site) by bioconcentration (BCF) or bioaccumulation factors (BAF). When used with a chronic criteria value, the TRV may be further interpreted as a tissue residue in aquatic biota above which adverse ecological effects may occur (Shepard, 1998; Dyer and others, 2000). For example, a chronic cadmium criterion (CCC) value multiplied by a mean BCF for cadmium yields a tissue reference value of about 1 mg/kg dw for aquatic organisms as:

TRV = CCC × BCF = 0.41 µg/L × 2,623 L/kg tissue as wet weight = 1,075 µg/kg = 1.08 mg/kg ww.

The BCF is a ratio of internal concentrations of a contaminant in the organs or whole body of an organism to the concentrations in water (McGeer and others, 2003). Tissue residues are reported in the literature either as dry weight or as wet or fresh weight. Dry weight residues reflect the concentrations remaining in tissue samples after water is removed by heating or freeze drying, whereas wet or fresh weight concentrations are calculated and reported on their concentration in the original tissue matrix. Dry weight normalized concentrations may be more comparable across studies because they correct for the different water content of diets or tissues, which in studies listed in table 10 ranged from about 20 percent in trout chow to 90 percent in zooplankton. In contrast, wet or fresh weight tissue residues are more relevant for evaluating ecological risk because no animals dry their prey before eating. Because most of the data reviewed were reported on a dry weight basis, they are interpreted here on a dry weight basis. Thus for use with dry weight data, the TRV equation needs an adjustment for water content. Assuming a typical water content of 80 percent, the wet weight tissue TRV from the above example is multiplied by 5, yielding a dry weight cadmium TRV of 5.4 mg/kg dw (for example, cadmium TRV = 1.08 mg/kg ww ≈ 5.4 mg/kg dw).

Cadmium TRVs were calculated in this manner for a range of BCF estimates and chronic criteria values (table 10). These calculated values provide crude benchmarks to compare against measured concentrations of cadmium in tissues or diets that co-occurred either with adverse effects or no obvious adverse effects to various organisms. These comparisons in give some indication of whether continuous exposure to the water-based chronic cadmium concentration would be protective of from adverse effects via dietary bioaccumulation. TRVs ranged from 0.5 to 14.6 mg/kg dw when calculated over a range of BCF estimates and over a range of hardness-dependent criterion values. Qualifications such as “crude” and “some” are needed because of the significant uncertainties in calculating and interpreting cadmium TRVs calculated in this manner. McGeer and others (2003) found that BCFs for cadmium and other metals were characterized by extreme variability in mean values and a clear inverse relationship between BCF/BAF and aqueous exposure. The highest BCF values were at low and naturally occurring exposure concentrations. The inverse BCF to exposure relation illustrates that, although cadmium concentration increases with exposure, internal accumulation does not rise as quickly as exposure levels and therefore indicates a significant degree of control over cadmium accumulation. McGeer and others (2003) found that if BCFs were limited to the cadmium exposure range of 0.1 to 3 µg/L, the range where chronic effects were predicted to begin, mean BCFs were higher than BCFs calculated for all data (2,623 versus 1,866, with coefficients of variability of 230 and 260 percent, respectively). They also derived modified accumulation factors that accounted for only the increased accumulation that arises from an increase in exposure concentration by removing the preexisting concentrations (from controls) from the calculation. Their mean modified accumulation factor was considerably lower than the conventionally calculated BCFs (352 versus 1,866). McGeer and others (2003) cautioned that using the weight of evidence available, it was virtually impossible to derive a meaningful BCF value that is representative of the BCF for each metal. Even when BCFs are limited to the exposure range where chronic toxicity might be expected (based on water-quality guidelines), it was not possible to derive a precise and accurate BCF value (McGeer and others, 2003). Nonetheless, if McGeer and others (2003) modified accumulation factor was considered the best available estimate of a typical BCF, when used with the CCC for a moderate hardness of 50 mg/L, the resulting TRV (0.41 µg/L × 352 L/kg × 5 = 0.7 mg/kg dw) is lower than the lowest adverse effect associated with a dietary exposure or tissue residue in table 10.

These calculations used BCFs that were usually derived from laboratory exposures where the only route of exposure to elevated concentrations is through the water. When derived from field studies, the term bioaccumulation factors (BAFs) is used to indicate that the exposure routes include both food and water. BAFs are calculated the same way as BCFs, but in practice reliable BAFs are more difficult to determine than BCFs because under field conditions it is difficult to make enough measurements of the material in water to show it was reasonably constant for long enough period of time over the territory range of the organisms (Stephan and others, 1985). In a cursory comparison of cadmium BAFs to BCFs, BAFs based on mean water concentrations and whole body invertebrate tissues (mayflies and amphipods) were calculated from 13 exposures reported in five studies (Farag and others, 1998; Birge and others, 2000; Besser and others, 2001; Riddell and others, 2005a; Stanley and others, 2005). BAFs, calculated on a wet weight basis to be comparable to reported BCFs, ranged from 88 to 4,260 with a mean of 1,389 and a coefficient of variation of 101 percent. This mean BAF value is intermediate to McGeer and others' (2003) mean values, which ranged from 352 to 2,623 depending on the method calculated.

In summary, the data reviewed on effects of cadmium tissue-residues in fish and invertebrates were insufficient to analyze quantitatively similarly to data on the effects of waterborne cadmium. Qualitatively, the data reviewed suggest that fish can tolerate cadmium gill and kidney tissue residues as much as at least 4 and 20 mg/kg dw, respectively, without obvious adverse effects. Cadmium in the diet apparently poses little risk to rainbow trout in concentrations as much as about 50 mg/kg dw. However, although predatory fish such as rainbow trout may be able to tolerate feeding on prey with tissue residues greater than about 50 mg/kg dw, these tissue residues may be harmful to the prey species themselves. Some adverse effects with mayflies were associated with whole-body cadmium tissue residues as low as 2 mg/kg dw, however other data suggested some mayflies can tolerate cadmium tissue residues as much as about 30 mg/kg dw (table 10). Estimated tissue residues that might result from continuous exposure to chronic cadmium criteria concentrations were near the low range of residue concentrations associated with adverse effects.

In summary, the data reviewed on bioaccumulation and effects of dietary exposures to cadmium indicate that at chronic criterion concentrations, cadmium is unlikely to bioaccumulate to tissue residue levels expected to cause obvious adverse effects to aquatic invertebrates or fish. However, the data reviewed were not sufficient to define quantitatively a maximum acceptable dietary intake or tissue concentration for fish or other aquatic-dependent life based on observations of survival, growth, or reproduction in a chronic feeding study or a long-term field study. Therefore, no "final residue value" criterion based upon tissue residues and bioaccumulation or bioconcentration factors could be appropriately derived as described by Stephan and others (1985).

Behavioral Toxicity

Cadmium has been shown to cause neurotoxic effects in fish. These neurotoxic effects may manifest themselves through altered behavior, which in turn may predict more serious effects including reduced growth, reproductive failure, and death. Hyperactivity probably is the most widely observed maladaptive behavior reported from cadmium exposed fish, with several reports involving a variety of fish species during long-term cadmium exposures. Most fish that exhibited hyperactive behavior in long-term exposures ultimately died (Eaton, 1974; Benoit and others, 1976; Spehar and others, 1978b; Sullivan and others, 1978; McNicol and Scherer, 1991; Riddell and others, 2005a, 2005b). In contrast to this pattern, Schreck and Lorz (1978) noted that cadmium exposed coho salmon never behaved violently, but appeared sedated up until the time of death with no increase in serum cortisol levels, a stress indicator. Hyperactivity is detrimental to small fish because it makes them more likely to be seen and attacked by predatory fish. Similarly, hyperactive predatory fish have lower success rates in detecting, orienting to, attacking, and swallowing prey (Sullivan and others, 1978; Kislalioglu and others, 1996; Riddell and others, 2005b).

The mechanisms and sites of action of neurotoxic effects are unclear, because unlike metals such as aluminum, copper, lead, and mercury, cadmium apparently does not pass the blood-brain barrier, does not accumulate in the higher centers of the brain, and does not alter brain neurochemical activity (Beauvais and others, 2001; Scott and others, 2003; Scott and

Sloman, 2004). Several studies have found that cadmium did not accumulate in brain tissues (Kumada and others, 1973; Benoit and others, 1976; Thomas and others, 1983; Brown and others, 1986). Scott and others (2003) observed that cadmium could not be detected in the brain tissues of rainbow trout that showed maladaptive behavior following cadmium exposure. However, cadmium deposition in the olfactory system (rosette, nerve, and bulb) during waterborne exposure was greater than in all other organs of accumulation except the gill. No cadmium deposition was detected in the lateral line of the trout, despite that the lateral line has neural cells that are in direct contact with water. Lateral line function and rheotaxis was inhibited in a New Zealand galaxid following exposure to 2 µg/L cadmium. Rheotaxis, the behavior of fish to turn to face a current enabling them to hold position in moving water, depends upon lateral line function (Baker and Montgomery, 2001). Galaxids are stream-resident fish that hold a similar ecological niche to that of salmonids in the northern hemisphere.

Preference or avoidance testing with fish also has been reported for cadmium. Fish were given the choice of selecting parts of a test chamber with or without cadmium. However, of the types of behavioral testing reviewed, preference/avoidance testing may be the least relevant to natural waters. In natural waters fish likely select and move among habitats based on myriad reasons such as access to prey, shelter from predators, shade, velocity, temperature, and interactions with other fish. In contrast, laboratory preference or avoidance tests are commonly conducted under simple, highly artificial conditions to eliminate or minimize confounding variables other than the water characteristic of interest. Such tests may overestimate the protection this behavior provides fish in heterogeneous, natural environments. Using lake whitefish (*Coregonus clupeaformis*), R.E. McNicol and colleagues at the Freshwater Research Institute, Winnipeg, extensively investigated the interactions between preference and avoidance of cadmium and environmental factors such as light and temperature. Whitefish could detect cadmium concentrations in a counter-current trough as low as 0.2 µg/L. Under uniform test conditions, fish displayed a bimodal concentration-avoidance response relation. Fish generally avoided concentrations less than or equal to 5 µg/L and greater than 50 µg/L, but showed little response to concentrations in between. However, when cadmium was added to a shaded area of the trough, attraction to the shade completely suppressed cadmium avoidance for concentrations as much as 10 times those avoided under homogeneous lighting. Similarly, when cadmium was added to a thermally attractive region of the trough, avoidance of cadmium was again suppressed, with the degree of suppression varying with the strength of temperature attraction. Mildly attractive 15°C water completely suppressed cadmium avoidance up to a concentration of 50 µg/L. When cadmium was presented opposite lethal temperature water (24°C), fish displayed a strong aversion to the warm water which suppressed cadmium avoidance at all test concentrations (McNicol and Scherer, 1991; McNicol, 1997; McNicol and others, 1999). Thus, when the attractive influence of a natural factor opposed the repelling influence of a chemical stressor, avoidance of the chemical stressor was suppressed. It follows that results of simple laboratory preference/avoidance testing have little relevance in complex natural habitats.

Cadmium exposures have also been shown to reduce the ability of fish to respond to natural chemical attack alarms. After a predator breaks the skin of a fish, the fish releases a chemical alarm signal from fish skin epithelial cells. Scott and others (2003) found that waterborne exposure to 2 µg/L cadmium for 7 days eliminated the normal antipredator behaviors exhibited by rainbow trout in response to alarm substance, whereas exposures of shorter duration or lower concentration had no effect on normal behavior. Predator avoidance behavior is complimentary to predatory behavior. Riddell and others, (2005a and 2005b) observed that brook trout exposed to sublethal cadmium concentrations had reduced success in capturing their preferred mayfly prey, subsequently switching their prey preferences to less motile midges, with consequences of reduced growth. The LOEC for the behavioral alterations of 0.4 µg/L at a hardness of 156 mg/L was lower than chronic test endpoints for brook trout.

Other fish behaviors that may be disrupted by cadmium exposures include social behavior, dominance hierarchies, and prey selection (Sloman and others, 2003a, 2003b, and 2005; Scott and Sloman, 2004). Long term exposure of rainbow trout to 3 µg/L cadmium in water with a hardness of 140 mg/L had no effect on their critical swimming speed (McGeer and others, 2000a).

In summary, cadmium exposures may result in neurotoxic effects that may manifest themselves through altered behavior, which in turn may predict more serious effects including reduced growth, reproductive failure, and death. Generally, behavioral responses were observed at lower cadmium concentrations than concentrations causing mortality. Complex behaviors provide the foundation for fish population structure and aquatic communities, and if natural behaviors were fundamentally altered by cadmium or other pollutants, that would have serious implications for aquatic ecosystems.

Sensitivity of Different Life Stages of Aquatic Organisms to Cadmium

The data reviewed generally supported (or at least did not clearly refute) the assumption that juvenile fish and early instars of invertebrates were more sensitive to cadmium than older life stages. The most information was available for fish (salmonids, sculpin, and fathead minnow) and two commonly tested invertebrates, *Daphnia magna* and *Hyalella azteca*. However, the data reviewed on size or age relations to cadmium sensitivity were not always pronounced or consistent.

Results of experiments that have been reported for different life stages or ages of freshwater aquatic invertebrates commonly used for toxicity testing were equivocal regarding identification of any particular life stage that is consistently most sensitive to metals. For example, sensitivity of *Chironomus tentans* to cadmium decreased in advanced developmental instars of the midge, with the first instar being more than 100 times more sensitive than later instars (Williams and others, 1986). Conversely, the sensitivity of *Daphnia magna* to cadmium and copper was similar for organisms ranging from less than 1 to 6 days old (Nebeker and others, 1986a). Collyard and others (1994) found that the sensitivity of *Hyalella azteca* to toxicants in short-term exposures does not differ greatly as a function of age over a range of <1–26 days old at test initiation. In the latter tests, 96-hour LC_{50} values for cadmium chloride varied by about a factor of 2 among six age classes of *Hyalella azteca* ranging from 0 to 2 to 24 to 26 days. Organisms 10–12 days old at test initiation had the lowest cadmium LC_{50}, however, differences between the age groups were small (Collyard and others, 1994). Wigginton (2005) reported that the early life stages of species of crayfish in the family Cambaridae were 10 to 40 times more sensitive to cadmium than adults.

Among several life stages of Chinook salmon and steelhead (that is, the anadromous form of the rainbow trout), Chapman (1978b) found that with Chinook salmon the swim-up fry was the most sensitive life stage, but with steelhead, swim-up fry and parr life stages were similarly sensitive to acute-cadmium exposures, whereas both younger (alevin) and older and larger (smolt and adult) life stages were considerably more resistant.[2] In contrast, steelhead exposed to copper and zinc at the swim-up stage were more sensitive than parr. Chapman (1975) found a slightly different pattern with coho salmon than with Chinook and steelhead. Six rangefinding acute tests were conducted using fry hatched from the same batch of eggs to evaluate the sensitivity of different aged fish. Results suggested that the youngest swim-up fry and older juveniles were more resistant than 6-week old fry. Chapman and Stevens (1978) found that the greater resistance of adult salmonids to cadmium than juveniles was only temporary. When measured over 4 days, adults were much more resistant than juveniles, but when measured over 17 days, the adults were as sensitive as juveniles.

Hansen and others (2002a) noticed different patterns of sensitivity of rainbow trout and bull trout within the fry life stage. Older and larger (about 1 g) fish were more sensitive than younger and smaller (about 0.1 to 0.2 g) fish. Stubblefield and others (1999) reported very small (0.19 g average weight) swim-up fry rainbow trout were less sensitive to cadmium than were adults (table 17).

For mottled sculpin, Besser and others (2006) found that newly hatched fish were much more sensitive to cadmium, copper or zinc than older juveniles or yearlings. Tests with mottled sculpin conducted with 3-week, 7-week, 3-month old, and yearling fish showed a consistent pattern of increasing resistance with age. Unlike the patterns observed with salmonids for cadmium and copper, the newly hatched sculpins were even more sensitive than the swim-up fry.

Most life-stage sensitivity data reviewed was consistent with the notion that juveniles were probably more sensitive than older organisms, and salmonid fry about 6–10 weeks posthatch or about 1 g in weight may be particularly vulnerable. However, the available data were too inconsistent to consider a *priori* life stages to be sensitive or resistant. Thus, if data for only one life stage were available, it was considered usable in table 15. If data for more than one life stage were available, they were used in table 15 or rejected on a case-by-case basis. Data rejected because tests used apparently resistant life stages included: (1) all acute results for adult fathead minnows because they were consistently more resistant than fry; (2) results of a flow-through acute test with yearling brook trout that were about 1,000 times higher than an acute test with brook trout fry; and (3) results of a flow-through test with 6-month old coho salmon (table 17).

Acclimation

Exposure to sublethal concentrations of cadmium and other metals may result in pronounced increased resistance to later exposures of the organisms. Several studies concluded that acclimation is a real compensatory response that plays an important role in the existence of natural fish populations in metals polluted environments. However, the increased resistance of fish to metals exposure can be temporary and lost in as few as 7 days after return to unpolluted waters (Bradley and others, 1985; Sprague, 1985; Hollis and others, 1999; Stubblefield and others, 1999). For this reason, EPA's guidelines for deriving aquatic life criteria specify that test results from organisms that were pre-exposed to toxicants should not be used in criteria derivation (Stephan and others, 1985).

[2] Alevins are fish that have hatched but still contain a yolk sac. Swim-up fry are fish that are 4–6 weeks post hatching, have resorbed their yolk sacs and have begun feeding, and have just swum up from their incubation cups or natural gravels to swim freely in the water column. Parr are larger juveniles that may be up to two years old.

Screening results of short-term acute tests for pre-exposure to toxicants is reasonably straightforward. Evaluating the role of acclimation in longer term, chronic tests is less so. Several tests have shown that at least with fish, if toxicity tests were initiated during metals resistant life stages (adult or embryo), acclimation may occur. Later in the test when the more sensitive life stages become exposed (fry or juvenile), usually sensitive life stages may be more resistant than the same life stages of naïve fish which had no pre-exposure (Chapman, 1978a, 1994; Spehar and others, 1978b; Farag and others, 1994; Brinkman and Hansen, 2004a; De Schamphelaere and Janssen, 2004).

Chapman (1978a) exposed adult sockeye salmon (*Oncorhynchus nerka*) to a series of zinc concentrations for 3 months followed by an 18-month exposure of embryonic through smolt stages. When 9-month old parr salmon were tested for acute resistance to zinc, the acclimated fish were 2.2 times more resistant than the control fish. In further experiments, Chapman (1994) exposed different life stages of steelhead (*Oncorhynchus mykiss*) for the same duration (3 months) to the same concentration of copper (13.4 µg/L at a hardness of 24 mg/L as $CaCO_3$). The survival of steelhead that were initially exposed as embryos was no different than that of the unexposed control fish, even though the embryos developed into the usually sensitive swim-up fry stage during the exposure. In contrast, steelhead that were initially exposed as swim-up fry without the opportunity for acclimation during the embryo state suffered complete mortality.

Brinkman and Hansen (2004a) compared the responses of brown trout (*Salmo trutta*) to long-term cadmium exposures that were initiated either at the embryo stage (early-life stage tests) or the swim-up fry stage (chronic growth and survival tests). In three comparative tests, fish that were initially exposed at the swim-up fry stage were consistently 2–3 times less resistant than were the fish initially exposed at the embryo stage.

These studies support the counterintuitive conclusion that because of acclimation, longer term tests or tests that expose fish over their full life cycle are not necessarily more sensitive than shorter term tests which are initiated at the sensitive fry stage. This conclusion has important implications for relating test results to potential effects in the wild. Some groups of temperate freshwater fishes, such as salmonids and suckers, have migratory life histories. Their life histories often involve spawning migrations to headwater reaches of streams, followed by downstream movements of fry shortly after emerging from the substrates, and later seasonal movements to larger, downstream waters to over winter (Willson, 1997; Baxter, 2002; Cooke and others, 2005; Quinn, 2005). These life history patterns often correspond to human development and metals pollution patterns such that headwater reaches likely have the lowest metals concentrations and downstream increases could occur due to point source discharges or urbanization. At least for salmonids and suckers, the scenario of metals acclimation developing at the egg stage in upstream spawning reaches that would provide fry and older juvenile stages additional protection seems unlikely. Therefore, full life cycle toxicity testing or early life stage testing methods may not necessarily provide "safer" or more relevant estimates of acceptable long-term exposure concentrations than shorter term methods initiated at sensitive life stages.

Evaluation of Risks to Biological Integrity and Vulnerable Species

The preceding descriptions of effects of cadmium on species and criteria derivation focused on the results of testing single-species with cadmium under constant conditions in laboratory settings. This reductionist approach to criteria derivation was needed to eliminate confounding variables and to make comparisons of species-sensitivities internally valid. This of course is not the case in the real world where species interact and contaminants occur in mixtures and in variable concentrations.

The criteria values derived in this report were evaluated in the context of whether waters with cadmium concentrations near criteria conditions would be expected to maintain biological integrity. This effort included:

1. Evaluating the criteria concentrations that were derived from toxicity tests conducted in laboratories under constant conditions in the context of fluctuating cadmium concentrations in field conditions;

2. The assumption that the most sensitive 5 percent of taxa from a species-sensitivity distribution need not be explicitly protected in criteria derivation was critically examined by considering the role that a more-sensitive "unprotected" species has in aquatic food webs and through population modeling; and

3. Comparing apparent effects concentrations from field surveys or ecosystem studies to corresponding criteria concentrations.

Criteria in Variable Environments

One major problem with estimating chemical criteria that are neither underprotective nor overprotective is how to bridge the gap between the nearly constant concentrations used in the toxicity tests used to derive the criteria concentrations and the fluctuating concentrations that usually exist in the real world (Stephan and others, 1985). Stephan and others (1985) addressed the problem by defining averaging periods for criteria. The intent of their criteria guidelines was to avoid "unacceptable" effects, and some adverse effects such as a small reduction in the survival, growth, or reproduction even in important species was considered possible at or less than criteria. The criterion continuous concentration (CCC) was intended to be a good estimate of this threshold of unacceptable effect. If maintained continuously, any concentration greater than the CCC was expected to cause an unacceptable effect. However, they expected that the concentration of a pollutant in a water body could be greater than the CCC without causing an unacceptable effect if (1) the magnitudes and durations of the excursions were limited and (2) if there were compensating periods of time in which the concentrations were less than the CCC. The more the concentration was greater than the threshold of unacceptable effects, the shorter the period of time that it could be tolerated. But, whether there was any upper limit on the magnitude of exceedences that could be tolerated for very short period of times, such as on the order of 1 minute, was considered unimportant because concentrations in ambient waters rarely change substantially in such short periods of time (Stephan and others, 1985).

Stephan and others (1985) considered an averaging period of 1 hour most appropriate to use with the criterion maximum concentration (CMC) because high concentrations of some materials could cause death in 1 to 3 hours. Additionally, even when organisms do not die within the first few hours, few toxicity tests continue to monitor for delayed mortality after the exposure period ends. Thus it was considered inappropriate to allow concentrations greater than the CMC for more than 1 hour (Stephan and others, 1985). U.S. Environmental Protection Agency (2001) used a 24-hour averaging period for their cadmium CMC, although no explanation was given for the departure from Stephan and others (1985) guidelines.

Stephan and others (1985) guidelines on this point were based upon their collective experience and judgment rather than specific datasets. However, subsequent research on delayed toxicity in pulsed exposures and relations between metals accumulation on fish gill surfaces and toxicity is relevant to the selection of criteria averaging periods. Marr and others (1995) episodically or continuously exposed fish to a metals mixture; the concentrations and ratios of the metals and variations in water quality (pH, hardness) of which were selected to represent conditions measured during episodic storm events in the Clark Fork River, Montana. Mortality resulted from 8-hour exposures followed by 96-hour observation periods. At decreased hardness and constant pH (simulating hardness decreasing with runoff), the 8-hour LC_{50} was 4 times that of a 96-hour LC_{50} (Marr and others, 1995). In pulsed exposure chronic toxicity tests with fathead minnows, a single 6-hour pulse of cadmium at 3 times the 7-day chronic value followed for 7-days observation without further metals exposure resulted in slightly (5 percent) reduced survival (Where the 7-day chronic value was 13 µg/L (EC_{25}) and the pulse was 40 µg/L, tests were at a hardness of 90–110 mg/L). A 12-hour pulse followed for 7 days resulted in high mortality (61 percent). A single 5 times pulse for 6 hours (60 µg/L) resulted in very high mortality at the end of 7 days (Diamond and others, 2005). In other tests, amphipods (*Hyalella azteca*) were exposed to different toxicant concentrations, and the percentage of mortalities was noted both during and after the exposure ended. For copper at the conventional 48-hour LC_{50} concentrations, the predicted proportions dead after including latent mortality were 65 to 85 percent, not 50 percent. Because of latent mortality, “true” *Hyalella azteca* LC_{50s} with copper could be about one-half those calculated immediately at the end of acute exposures (Zhao and Newman, 2004).

When fish are exposed to cadmium or copper, a relatively rapid increase occurs above background levels of metal bound to the gill. This rapid increase occurs on the order of less than 3 to 24 hours, and this brief exposure has been sufficient to predict toxicity at 120 hours (Playle and others, 1993; Playle, 1998; MacRae and others, 1999; Di Toro and others, 2001). Acute exposures of 24 hours might not result in immediate toxicity, but deaths could result over the next few days. Simple examination of the time-to-death in 48- or 96-hour exposures would not detect latent toxicity from early in the exposures. Observations or predictions of appreciable mortality resulting from metals exposures on the order of only 3 to 6 hours supports the earlier recommendations by Stephan and others (1985) that the appropriate averaging periods for the CMC is on the order of 1 hour.

Stephan and others (1985) considered a 4-day averaging period appropriate to use with the CCC for two reasons. First, “chronic” responses in some species may not really be due to long-term stress or accumulation, but rather the test was simply long enough that a briefly occurring sensitive stage of development was included in the exposure (Chapman, 1978a; Barata and Baird, 2000). Second, a much longer averaging period, such as 1 month would allow for substantial fluctuations above the CCC. Substantial fluctuations may result in increased adverse effects from those expected in constant exposures (Stephan and others, 1985). A comparison of the effects of the same average concentrations of copper on developing steelhead, *Oncorhynchus mykiss*, that were exposed either through constant or fluctuating concentrations found that steelhead were about twice as resistant to constant exposures as to fluctuating exposures. In daily fluctuating copper exposures that peaked at the 96-hour LC_{50} concentration for 7 hours, the onset of growth reduction occurred at an average copper concentration of 16 µg/L copper, whereas in constant exposures, the onset of growth reduction occurred at an average concentration of 30 µg/L (Seim and others, 1984). A similar test design using cadmium found no increase in chronic toxicity resulting from fluctuating exposures of *Daphnia pulex* over constant exposures. In this case, the *Daphnia* were only exposed daily for 70 minutes to cadmium pulses at their 48-hour LC_{50} concentration that produced a daily-average cadmium exposure that was similar to the chronic cadmium criterion concentration in effect at the time (Ingersoll and Winner, 1982). These latter results indicate that fairly high cadmium exposures of about 1 hour may be tolerated by some organisms.

Measured cadmium fluctuations in several regional streams were examined to estimate how ambient fluctuations compared to experimental toxicity results from pulsed exposures to cadmium (fig. 3). Comparisons were limited to waters for which multiple cadmium detections and matched calcium and magnesium measurements were available (the latter were needed to calculate hardness and chronic criterion values). Differences between average long-term concentrations and peak-measured concentrations ranged from 1.1 to 5 times with the median average-to-peak difference for all sites of 2.2 times, with the exception of a single anomalous value from Thompson Creek that was 30 times higher than the median value for that stream. These comparisons suggest that if the results of the experiment with fluctuating copper exposures and steelhead are relevant to cadmium, then the potential increased sensitivity of about 2 times caused by fluctuating concentrations would be roughly offset by the regional pattern of peak cadmium concentrations being about 2 times higher than median concentrations.

In addition to the seasonal patterns shown in figure 3, daily fluctuations can be important in some streams. In streams draining historical mining areas in Idaho and Montana, cadmium concentrations in streams increased greater than 2 times from afternoon minimum values to maximum values shortly after sunrise (Nimick and others, 2003; Nimick and others, 2005). These patterns have important implications for water-quality monitoring because people are more apt to collect water samples in the afternoon than at sunrise. Because similar relations were observed in streams in which metals concentrations were greatly elevated above background as in streams in which metals are closer to background levels (Nimick and others, 2005), daily cycles may be important from a toxicological view.

Daily fluctuations in exposure conditions results in chemical non-equilibrium, which likely changes the toxicity of metals. However, the magnitude and even the direction of effect (more or less toxic) are unclear. The toxicological relevance of non-equilibrium exposure conditions probably has been studied best with copper and dissolved organic matter, where about 24 hours are needed for equilibrium, and the toxicity of copper decreased when added copper was given longer to equilibrate with dissolved organic matter (Kim and others, 1999: Ma and others, 1999). With cadmium, reaction kinetics with dissolved organic matter are less important than with copper. Still, the reaction of the added cadmium with inorganic materials in the water probably occurs at rates relevant to the daily cycles shown in figure 3. Davies and Brinkman (1994b) reported that in kinetic equilibrium experiments in hard, alkaline water, it took about 48 hours for cadmium activity to reach equilibrium.

Davies and Brinkman (1994b) tested the relative toxicity of cadmium to rainbow trout in non-equilibrium and equilibrium exposures. In their non-equilibrium (unaged) exposure, the cadmium and hard water were mixed in a diluter and immediately discharged directly into the test aquaria. In their equilibrium (aged) exposure, the cadmium and hard water were allowed to mix in an intermediate aquarium for about 82 hours before overflowing into the test tanks. The total contact times (time for 95 percent water replacement) for the cadmium and hard water were about 28 hours in the unaged and 135 in the aged exposure. Their hypothesis was that cadmium would be more toxic in non-equilibrium conditions because by adding cadmium as soluble, free metal ions, toxicity would be overestimated in unaged hard water where insufficient time was allowed to establish equilibrium. Their results showed the opposite. Cadmium was more toxic (lower EC*p* and LOEC values) in equilibrium conditions than in non-equilibrium conditions. Davies and Brinkman (1994b) concluded that under conditions of non-equilibrium, results from toxicity tests will significantly underestimate toxicity because of analysis of metastable forms of metals which are not biologically available. (Metastable refers to an unstable and transient but relatively long-lived state of a chemical or physical system, such as of a supersaturated solution.)

In contrast to copper, where all data indicated toxicity was likely greater in non-equilibrium than equilibrium conditions (Kim and others, 1999: Ma and others, 1999), the only known cadmium study had opposite findings (Davies and Brinkman (1994b). Thus, daily fluctuations in exposure conditions (fig. 3) likely result in chemical non-equilibrium and influence cadmium toxicity. However, the data with cadmium are sparse, making it difficult to infer from the experimental results whether comparable patterns would be expected in different environmental conditions (for example, softer water, shorter exposure times.

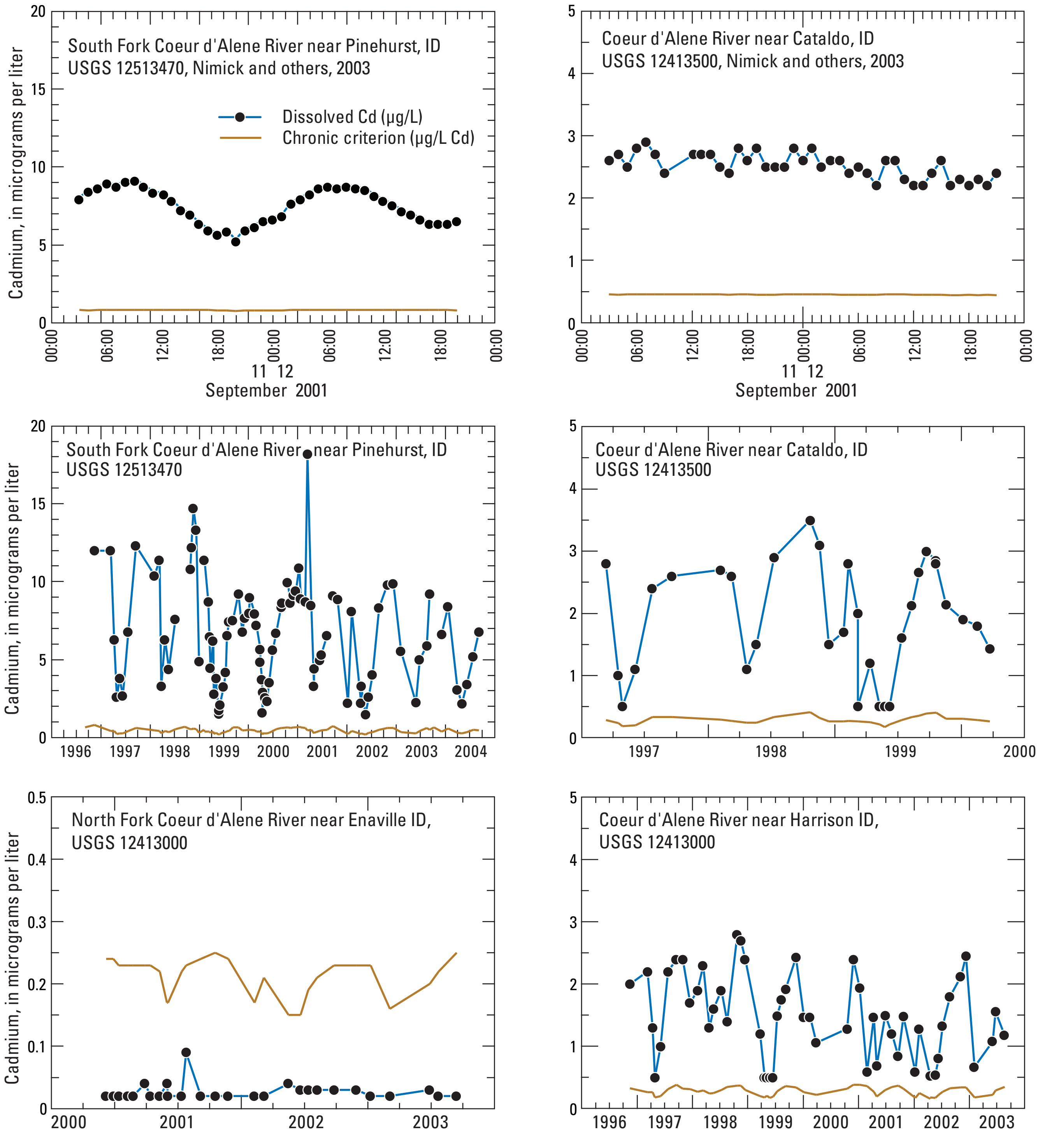


Figure 3. Variability in dissolved and chronic cadmium criterion concentrations in selected streams in Idaho and Montana, 1997–2005. "USGS" followed by a number indicates USGS station numbers that were accessed at http://waterdata.usgs.gov/nwis, April 2005 data from Thompson Creek and the Salmon River were provided by Bert Doughty, Thompson Creek Mining Company, Challis, Idaho, written commun., 2005; data from the Yankee Fork were provided by Bob Tridle, Hecla Mining Company, Coeur d'Alene, Idaho, written commun., 2005.

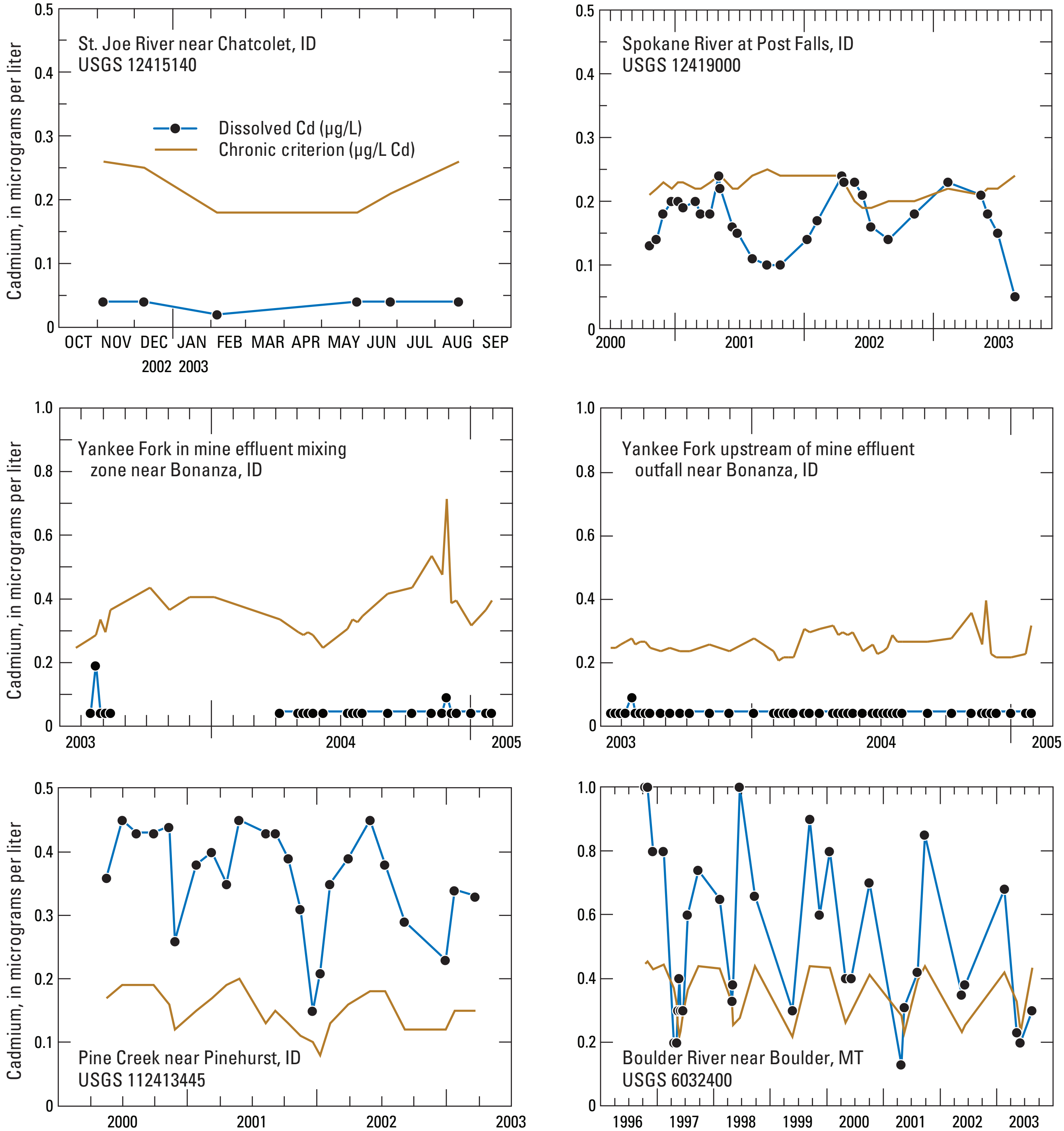


Figure 3. Continued.

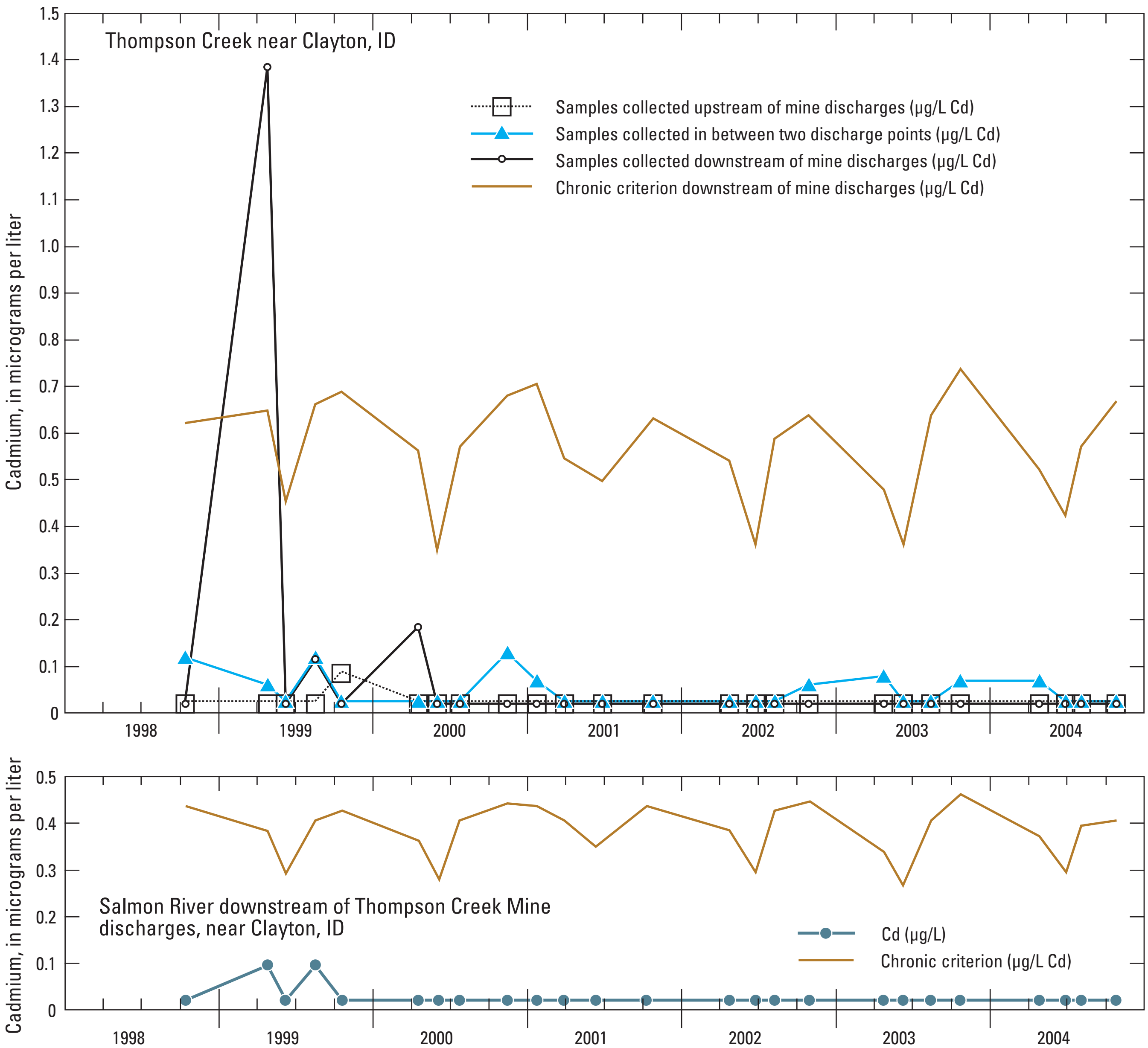


Figure 3. Continued.

Risks of the 5th Percentile Species-Sensitivity Distribution (SSD)-Based Chronic Criterion to a More Sensitive Species

One feature inherent to defining water-quality criteria on the basis of the 5th percentile of the SSD of single-species laboratory toxicity tests for the chemical is that by definition, of the species tested, species with mean response values lower than the percentile may not be protected. Aquatic ecosystems are assumed to have some functional redundancy and resiliency such that unless the more-sensitive species are judged to be "critical" species, avoiding adverse effects to 95 percent of the aquatic species tested would be sufficient to protect the biological integrity of natural ecosystems (Stephan and others, 1985).

The "5th percentile" approach has been criticized for depending on assumptions that may not be met or are untested. These include the assumptions that haphazard collections of data from single-species laboratory toxicity tests can be used to represent the sensitivities of natural ecosystems, whether any species loss from a community is acceptable, and if so, whether the 5th percentile of the SSD is the appropriate level of protection, among others (Forbes and Forbes, 1993a; Smith and Cairns, 1993; Calow and others, 1997; Power and McCarty, 1997; Newman and others, 2000; Forbes and Calow, 2002; Posthuma and others, 2002; Suter and others, 2002b; Maltby and others, 2005). Often, fewer than 20 chronic values for different taxa are available, and all measured values are greater than the 5th percentile of the SSD. This makes evaluations of risks to more sensitive but unidentified taxa somewhat theoretical and abstract. However, when the chronic dataset used to construct the SSD exceeds 20, the question of protectiveness becomes more tangible. Effects of the chemical to known species with known life histories, roles in aquatic food chains, and measured sensitivities to the chemical can be evaluated.

In the case of the acute criterion derivation, the "5th percentile" approach probably would be unacceptable for use in Idaho. This is because of the 69 species for which species mean acute values (SMAVs) were available, the SMAVs for all species of trout native to Idaho fell below the 5th percentile of the SSD: cutthroat trout, rainbow trout, and bull trout (fig. 5 in the section "Field Validation," table 7). Cutthroat trout were considered to be a "critical" species in the sense of Stephan and others (1994b) because it is a native fish of conservation concern and, as the state fish of Idaho, is a socially important sport fish. Rainbow trout are an important game fish and bull trout are listed as threatened under the Endangered Species Act. All three species are important upper trophic level carnivores in many streams and lakes. Basing the acute criterion on the lowest SMAV effectively set the acute criterion close to the 1st percentile of the SSD (that is, protecting nearly 99 percent of aquatic species).

In the case of the chronic dataset, genus mean chronic values (GMCVs) were available for 21 genera, and only one GMCV and species mean chronic value (SMCV) fell below the 5th percentile of the SSD (for the amphipod *Hyalella azteca*). In contrast to the acute dataset, for the most chronically-sensitive species, *Hyalella azteca*, no *a priori* judgment was made that *Hyalella azteca* was a "critical" species. Instead, risks of *Hyalella azteca* population declines and indirect food web consequences were evaluated.

The following describes the role of *Hyalella azteca* in aquatic food webs in the Pacific Northwest, the occurrence of *Hyalella azteca* in waters observed to exceed the chronic criterion, the degree to which *Hyalella azteca* toxicity thresholds are below the chronic criterion, and forecasts the likely consequences to *Hyalella* populations of setting the chronic criterion above their SMCV.

Role of *Hyalella azteca* in Aquatic Food Chains

Taxonomically, *Hyalella azteca* is listed as the sole species in the family Hyalellidae, within the order Amphipoda (Integrated Taxonomic Information System, 2002). However, recent genetic studies argue that *Hyalella azteca* probably is a species complex, rather than a single species (Duan and others, 1997; Wang and others, 2004). These arguments are consistent with the wide geographic range of *Hyalella azteca.* The species was originally described from specimens collected from the Aztec region of southern Mexico, and the distribution of *Hyalella azteca* occurs from at least southern Central America, where the species was originally described, to the high Arctic.

Amphipods are benthic crustaceans that occupy an intermediate position in aquatic food webs between detritus and predators, such as salamanders and salmonids (Mathias, 1971). Salmonids and other fish readily prey upon amphipods, probably consuming them in rough proportion to their abundance relative to other vulnerable invertebrates. In a lake in southern British Columbia, the fish *Oncorhynchus mykiss* (rainbow trout) and *Oncorhynchus nerka* (sockeye or kokanee salmon) and the salamanders *Taricha granulosa granulosa* (rough skinned newt) and *Ambystoma gracile* (Northwestern salamander) "feed to a significant degree on amphipods" (Mathias, 1971). In the lower Snake River in Washington and Idaho, amphipods contributed 2.7 and 7.9 percent of identifiable prey categories found in the stomachs of juvenile Chinook salmon and steelhead, respectively, from Lower Granite Reservoir (n=379 and 204, respectively). For steelhead, amphipods were the 5th most important prey category after Diptera larvae (for example, midges and flies, 24 percent), insect exuviae (12 percent), mayfly nymphs (10 percent), and adult Diptera (9.1 percent). For Chinook salmon, amphipods were the 7th most important prey category after mayfly nymphs (27 percent), Diptera larvae (for example, midges and flies, 25 percent), adult Diptera (12 percent), ants

(9.1 percent), insect exuviae (7.8 percent), and caddisflies (3.4 percent) (Karchesky and Bennett, 1999). Although amphipods are often abundant in the tailwaters below reservoirs, this was not the case for the riverine Hells Canyon reach of the Snake River, Idaho. In intensive scuba and shallow water sampling of macroinvertebrates in the Hells Canyon reach of the Snake River, amphipods were relatively abundant (32 percent of the sample) in only 1 of 28 reaches sampled. Amphipods were as much as 5 to 10 percent of the samples in 4 reaches, and the remaining 23 reaches, comprised from 0 to 5 percent of the sampled assemblage. Amphipod taxa collected in the Hells Canyon reach of the Snake River were *Crangonyx*, *Gammarus*, and *Hyalella* (Myers and Foster, 2003). This relative scarcity suggests that amphipods unlikely are major components of the diets of Chinook salmon or other fish in the Hells Canyon reach of the Snake River.

In mountain lakes in Oregon, Strong (1972) noted that during early July rainbow trout stomachs were stuffed with *Hyalella* but by late July none could be found in trout stomachs. Amphipods other than *Hyalella* have been observed to be seasonally or periodically important in salmonid and sculpin diets and *Hyalella* could be similarly important. A larger amphipod, *Gammarus lacustris,* and *Daphnia pulex* dominated the diet of bull trout in an alpine lake in the Canadian Rockies where bull trout were the only species of fish present (Wilhelm and others, 1999). Similarly *Gammarus* sp. was the dominant food item in juvenile Chinook smolts in the lower Columbia River where Gammarus was abundant (Muir and Coley, 1996). The amphipod *Crangyonx richmondensis* was one of the dominant food forms for rainbow trout in a long-term study of the effects of pulp mill effluents on coldwater stream communities. In that study, experimental stream channels constructed along the lower Clearwater River, Lewiston, Idaho, were allowed to colonize naturally with invertebrates for several months before testing effluents. Trout stomach samples indicated the dominant food forms in control streams were Diptera (primarily Chironomidae), Amphipoda, and Gastropoda. The amphipods usually were 30 percent or more of trout food intake. Gastropods were only an important food source in autumn during stream studies (National Council of the Paper Industry for Air and Stream Improvement, Inc,, 1989). In response to the collapse of populations of the amphipod *Diporeia* sp., populations of two native benthivores, slimy sculpin (*Cottus cognatus*) and lake whitefish (*Coregonus clupeaformis*), suffered severe declines in eastern Lake Ontario, perhaps partly due to starvation, because *Diporeia* sp. was their principal prey (Owens and Dittman, 2003). Amphipods in the genera *Crangonyx, Diporeia,* and *Gammarus* appear to be less sensitive to cadmium than *Hyalella* (tables 7 and 17).

Under favorable conditions, *Hyalella azteca* can be extremely abundant. In Blue Lakes Spring, a tributary to the Snake River near Twin Falls, Idaho, *Hyalella azteca* densities reached 29,400 per m^2 (Maret and others, 2001). Although no dietary studies are known, at these densities *Hyalella azteca* likely are important in the diets of the sculpin and trout populations that are resident in this large spring.

In summary, the available literature indicates that fish and salamanders will exploit *Hyalella* and other freshwater amphipod populations when they are abundant, but if other suitable prey are available, will readily switch when amphipods are scarce or absent.

Hyalella azteca in Waters with Elevated Cadmium Concentrations

If *Hyalella azteca* are as sensitive to cadmium in the wild as laboratory studies indicate, then some correspondence between their occurrence and elevated cadmium concentrations would be expected. Biological and chemical data sets that were reasonably well matched in time and space were located from two river basins—the Clark Fork River, Montana, and the Coeur d'Alene/Spokane basin, Idaho. *Hyalella azteca* were present in the Spokane River at Post Falls, Idaho, in July 1999 (U.S. Geological Survey, 2005). Cadmium concentrations at that location were elevated well above background levels, commonly approaching and slightly exceeding the cadmium chronic criterion (maximum of 1.1 times the chronic criterion, fig. 3). Amphipods were common at some sublittoral (5–10 m depth) locations in Coeur d'Alene Lake that either were uncontaminated by cadmium, zinc, and lead wastes or that had moderately elevated concentrations (1.1 to 2.0 times the chronic criterion at the time of sampling). Amphipods were not collected at more contaminated sites, but they also were not collected from the reference lake, Priest Lake, Idaho (Ruud, 1996). Ruud (1996) did not identify amphipods below "Amphipoda," but because the only amphipod species collected by the U.S. Geological Survey from the outlet to Coeur d'Alene Lake (the Spokane River) was *Hyalella azteca*, it seems likely that the amphipods in Coeur d'Alene Lake included *Hyalella azteca.*

Amphipods were collected from several locations in the Clark Fork River and Milltown Reservoir, Montana, in 1991 as part of investigations of the ecological effects of metals contaminated sediment (Canfield and others, 1994). The amphipods were not identified below "Amphipoda," but because the only amphipod species collected from tributaries to the Clark Fork River in the 1999 USGS National Water Quality Assessment (NAWQA) surveys was *Hyalella azteca*, it seems likely that the "Amphipoda" collected by Canfield and others, 1994, included *Hyalella azteca.* Sediments collected from the sites where the amphipods were collected were contaminated by a mixture of metals which caused sublethal toxic effects to *Hyalella azteca* in laboratory exposures, although the magnitude of adverse effects was fairly low (Canfield and others, 1994). A cadmium concentration of 1.1 µg/L at a hardness of 100 mg/L was considered representative for the upper Clark Fork River (Woodward and others, 1994), which would be 1.75 times the chronic criterion derived in this report. This suggests that amphipod populations could persist at cadmium concentrations greater than the chronic criterion. An examination of data from the upper Clark Fork River near sites where Canfield and others (1994) collected amphipods showed that a maximum cadmium concentration of 1 µg/L reported in 1989 at a hardness of 172 mg/L was 1.3 times the chronic criterion. However, since 1993, when detection limits for the USGS dataset were lowered from 1 to 0.1 µg/L, the maximum cadmium concentration for the areas of the Clark Fork River where amphipods were collected was only 0.2 µg/L. The presence of amphipods in the Clark Fork River near a site where measured cadmium concentrations exceeded the chronic cadmium criteria once during the 3-year period prior to their collection is consistent with the concept that aquatic communities can tolerate infrequent exceedences of criteria. However, the evidence from this Clark Fork River case study is weak because the quality of routine USGS trace metals sampling and analyses prior to about 1993 was not as rigorous as that for subsequent data, and the more recent data show low cadmium concentrations. *Hyalella* were not abundant at any of the Clark Fork River sites sampled in 1991 (≤0.5 percent) (Canfield and others, 1994). In 1999, *Hyalella* were collected from tributaries to the Clark Fork River, but none were collected from several sites in the mainstem Clark Fork River (U.S. Geological Survey, 2005). Although failure to collect a taxa does not mean the taxa was absent from the sampled water body, it does indicate that the taxa was not common at the times and places sampled.

The Coeur d'Alene and Clark Fork River examples of amphipods being present in environments with cadmium concentrations near or exceeding the chronic criterion that was not necessarily designed to protect *Hyalella* suggests that in field conditions, *Hyalella* populations may be resilient to criterion concentrations. However, because the empirical data are limited, the potential effects of cadmium at chronic criterion concentrations to *Hyalella* populations were further evaluated through population modeling.

Simulating Effects of Cadmium to a Natural, Coldwater *Hyalella azteca* Population

With a GMCV of 0.33 µg/L versus the final chronic value 0.41 µg/L cadmium at a hardness of 50 mg/L, *Hyalella azteca* cannot be considered fully protected by the criteria. Based on the results of individual toxicity tests, continuous exposure to cadmium at the chronic criterion concentrations would result in adverse effects ranging from a 10 percent decrease in reproduction to greater than 50 percent mortality (table 11). However, in ecological risk assessments, the questions of interest usually relate to effects on the abundance, production, and persistence of populations and ecosystems, rather than the death or impairment of individual organisms. Effects of

Table 11. Effects of cadmium to the most chronically sensitive species, *Hyalella azteca,* at chronic criterion concentrations.

[**Abbreviations:** CCC, criterion continuous concentration; EC*p*, percentage (*p*) of test population adversely affected at a concentration; LOEC, lowest observed effect concentration; $CaCO_3$, calcium carbonate. **Symbols:** mg/L, milligram per liter; µg/L, microgram per liter; >, greater than]

Test No.[1] (table 16)	Hardness (mg/L as $CaCO_3$)	Chronic value (µg/L)	Chronic criterion (CCC, µg/L)	Percentage of organisms adversely affected at the CCC	Type of adverse effect	Reference
45	17	0.16	0.20	40	Increased mortality	Suedel and others, 1997
46	280	.98	1.0	10	Decreased number of young/females	Ingersoll and Kemble, 2001
47	126	.74	.65	25	Increased mortality	Chadwick Ecological Consultants, Inc., 2004a
48	153	1.02	.73	20	Increased mortality	Chadwick Ecological Consultants, Inc., 2004a
49	130	.25	.66	>50	Increased mortality	Borgmann and others, 1991
50	130	.72	.66	10	Increased mortality	Borgmann and others, 1989b
Mean				26		

[1]Except for the tests numbered 45 and 49, EC*p* values that corresponded to criterion concentrations at test hardness were calculated from the authors' data using a triangular probability distribution analysis for dichotomous mortality responses or by piecewise tailed regression analysis. EC*p* analyses were made using U.S. Environmental Protection Agency's Toxicity Response Analysis Program (Erickson, 2002). For test 45, the EC*p* value is the effect at the reported LOEC, which at 0.25 µg/L was close to the criterion concentration of 0.20 µg/L calculated for the hardness of the test water. Insufficient data were reported to calculate a response curve. For test number 49, insufficient information was reported to estimate an EC*p* value that corresponded with the CCC. However, because the 42-day EC_{50} (0.53 µg/L) was less than the CCC, the CCC would have likely resulted in >50 percent mortality for this test.

contaminants at these “higher” levels of organization can be extrapolated through mathematical simulations that integrate individual-level toxicity testing results with ecological theory. Population-level effects of cadmium at its chronic criterion concentration to *Hyalella azteca* were evaluated using life-stage or age structured matrix projection models, also known as Leslie matrix models. The model uses a matrix of survival and reproduction rates (vital rates) of the different life stages, and projects them in time using matrix algebra. The technique basically is a bookkeeping device in which the births and deaths are tabulated over a given time frame (Barnthouse, 1993; Caswell, 2001).

Population Modeling Methods

The model was constructed in two stages. Stage 1 was to construct a population model that reasonably reflected baseline conditions, absent any toxicant effects. Stage 2 was to overlay effects of cadmium observed through lifecycle toxicity testing to the baseline survival and reproduction rates of *Hyalella azteca.* To build the baseline model, a life history model for was constructed for a *Hyalella azteca* population that would reflect temperate, coldwater conditions. Vital rates and demographic characteristics for the model were taken from field and laboratory studies of the natural history of the species (Cooper, 1965; Mathias, 1971; Strong, 1972; de March, 1978; Gibbons and Mackie, 1991). The modeled population was considered to have the following annual life cycle: eggs are deposited in late spring and early summer, followed by an egg and juvenile stage lasting about three months, followed by an adult stage that overwinters and then reproduces in late spring and early summer. During the reproductive stage, females mate and produce about four broods (one every 14 days for 2 months) before dying.

This annual life cycle seemed appropriate to assume for the mid-latitude vicinity of Idaho, but it would not be relevant throughout the species range. In North America, *Hyalella azteca* occur from at least Guatemala (latitude 16°N) to the Arctic (latitude 68°), as long as the mean summer water temperature of lakes reaches 10°C. In warmer regions of the species range, it has generation times on the order of 6–8 weeks with shorter than annual life cycles (de March, 1977; Pennak, 1978). However, assuming an annual life cycle simplified the population model and was consistent with observations from coldwater populations experiencing mean summer temperatures around 15–20°C (Mathias, 1971; Strong, 1972; de March, 1977, 1978).

The demographic parameters and vital rates used in the modeling are summarized in table 12. Survival and fecundity in the model were further reduced at different cadmium concentrations by superimposing mortality and reduced reproduction from the results of Ingersoll’s and Kemble’s (2001) full life cycle study. This study was selected because of the seven chronic tests with *Hyalella azteca* that were located (table 16), it was the only one for which the concentration responses for the reproduction and survival endpoints were available. Because other studies have shown that reproduction in *Hyalella azteca* is strongly related to growth (Cooper, 1965; Strong, 1972; Ingersoll and others, 1998), results of population modeling using reproductive impairment also should have relevance to shorter “chronic” toxicity tests with *Hyalella azteca* that measured growth responses but were not continued through the full life cycle.

Forecasts of the effects of cadmium on *Hyalella azteca* populations used RAMAS® computer software (Spencer and Ferson, 1997a, 1997b). A transition matrix for the forecasts was built by recalculating the life stage based-vital rates as biweekly age-classes. Uncertainty in vital rates and environmental variation were estimated by running 500 replicate simulations of each scenario, with each replicate having variable or uncertain parameters sampled from the most appropriate statistical distribution (that is, Monte Carlo simulations).

Three basic scenarios were simulated.

1. Population growth rates (lambda, λ) calculated for a range of constant cadmium concentrations. Population growth rates have been considered more ecologically relevant than single test endpoints because they integrate both survival and reproduction (Forbes and Calow, 1999). However, they only can be calculated for deterministic projection matrices where parameters are considered to be known exactly and are constant through time, which is unrealistic for real populations.

2. Risk of population decline statistics were calculated across a range of constant cadmium concentrations. In contrast to the first deterministic scenario, these scenarios recognized the intrinsic unpredictability of natural systems (stochasticity) and used Monte Carlo simulations based on parameter variability estimates. Since these simulations produce a tangle of possible population trajectories, these projections are summarized with statistics describing the risks of the population declining by a given percentage at some time during the simulations.

3. Seasonal population trajectories for several years under

    a. Baseline conditions without any added cadmium stress;

    b. Adding constant exposure to cadmium at the chronic criterion; and

    c. An episodic exposure scenario in which the animals were exposed to 1.2 times the chronic criterion at the beginning of their reproductive cycle for 0.5 months once every 3 years with cadmium at 0.6 times the chronic criterion at other times.

Table 12. Vital rates used for modeling effects of cadmium to a coldwater *Hyalella azteca* population.

[**Abbreviations:** Cd, cadmium; EC_{50}, 50 percent of test population. **Symbols:** mg/L, milligram per liter]

Parameter	Mean	Standard deviation	Range	Parameter	Mean	Standard deviation	Range
Juvenile survival rates (overall for life stage)[1]	0.2825		0.25 – 0.32	Adult daily mortality rate during spring/summer (deaths per animal per day)[1]	0.0063		0.003 – 0.009
Juvenile half-monthly survival rates[2]	.908	0.22	0 – 1	Adult half-monthly survival, "A"[2]	0.9	0.05	0 – 1
Brood size, young per fecund female, "Y"[3]	8.35	2.28	6.3 – 15.5	Sex ratio, females:male, "S"[3]	1.7 : 1	.32	1.1 – 2.4
Proportion of females that are fecund, "F"[3]	.26	.07	0.14 – 0.430	Adult overwinter survival rate per half-monthly time step[4]	0.99		
Broods per half-month time step during reproductive season "B"[5]	1			Fecundity (=A·Y·S·F·B)[6]	1.23	.3	0.44 – 4.26
Broods per life time	4			Ricker density dependence function for fecundity (β)	-0.00105		-0.00138 – -0.00071
Cd 42-day EC50 at 280 mg/L hardness[7]	2.49			Cd reproductive impairment (broodsize) EC_{50} at 280 mg/L hardness[7]	1.96		
Slope of the probit curve at the EC_{50}[7]	.515			Slope of the probit curve at the EC_{50}[7]	0.313		

[1]Mean and range of observed survival at shallow and medium depth stations (Mathias, 1971).

[2]Mean and range of survival rates adjusted from the duration of observations to the half-monthly time steps for the life stage used in the model (assuming 3-months duration for juvenile life stage, 3 months for adult life stage, and 6-months duration for an inactive, overwintering phase).

[3]Gibbons and Mackie, 1991.

[4]Placeholder survival rate was assumed in order to adjust the model's time steps to an annual life cycle. While true survival rates during winter are undoubtedly lower, overwinter survival is probably high, based on Cooper's (1965) observations that mortality rates approached zero as fish predation declined with the onset of winter in a Michigan lake.

[5]Assuming a 14-day resting period between broods, based on de March's (1978) experimental results at 18ºC and assuming a 2-month reproductive season during late spring and early summer.

[6]Standard deviation for this compound parameter was estimated by calculating a coefficient of variation (CV) for each parameter and taking the product of the CVs.

[7]EC_{50} values and the slope of the probit curve were estimated using EPA's (1992) 2-parameter probit distribution using the raw data from the test described by Ingersoll and Kemble (2001).

This scenario is only slightly more severe than would be allowable under the frequency, magnitude, and duration for chronic criterion (the allowable duration is 4 days rather than 1 month). The scenario also is similar to that observed in the Spokane River, Idaho, where average cadmium concentrations were about half that of the maximum concentrations and where maximum concentrations slightly exceeded the chronic criteria annually (fig. 3). This scenario was too complex to model directly, so the model was run in four phases with the outputs from one phase used as the initial conditions of the next phase.

Coincidentally, at the 280 mg/L hardness value of the toxicity test results used for the simulations, the chronic criterion is 1 µg/L so that multiples of criteria exceedence can be interpreted directly as concentrations, for example, 0.6X = 0.6 µg/L. The models were projected for 6 years, that is, six generations with an annual life cycle.

A lognormal distribution was assumed for variability in baseline fecundity rates, based on patterns observed in 22 populations in lakes in Ontario (Gibbons and Mackie, 1991). Making realistic estimates of environmental variability was more problematic. Mathias's (1971) mortality rates for a *Hyalella* population that was preyed on by salamanders and salmonids in a Northwestern lake seemed most relevant for the simulations; however, only mean mortality rates per station were reported. For juvenile *Hyalella* in a Michigan lake during the summer, Cooper (1965) reported weekly mortality rates ranging from -2 to 60 percent with an average of 26 percent and a coefficient of variation of 72 percent. For adults during summer, calculated mortality rates ranged from 43 to 552 percent (Cooper, 1965). Although negative mortality rates can be assumed to be zero and reflect measurement error, how to interpret mortality rates greater than 100 percent was not obvious. Because real populations are influenced by unpredictable combinations of environmental factors such as weather, competition, and predation and sampling methods have considerable measurement error, ignoring variability and uncertainty in mortality rates would be environmentally unrealistic. Further, projections under static (deterministic) scenarios showed that the population structure was more sensitive to survival rates than to fecundity rates, so including some estimate of variability in mortality rates seemed appropriate. Thus, Cooper's mean coefficients of variation for juvenile mortality rates were used to indicate the relative variability in mortality rates that would be plausible for a natural *Hyalella* population. These coefficients of variation were multiplied by Mathias's mean values to obtain a plausible value for environmental variability in baseline mortality rates for a natural population. For adult mortality rates, the standard deviation of mean mortality rates from Mathias's different stations was used for the Monte Carlo simulations, recognizing that this probably under represents the natural variability in adult mortality rates.

Population Modeling Results

For the first scenario modeled under static, baseline conditions, the annual population growth rate (λ, lambda) for the population was calculated as between 1.01 and 1.09 (fig. 4*A*). This indicates that the model population was near equilibrium and suggests that the input parameters were plausible. A population with a λ value of 1.0 is at equilibrium with births and deaths replacing each other. Absent any compensating factors, if λ is less than 1.0, the population will eventually decline to extinction. If λ is greater than 1.0, the population will eventually explode exponentially until resources become limiting. As cadmium concentrations increase above 100 percent of the chronic criterion, (1 µg/L under all modeling scenarios), λ values fall below 1.0.

The second set of scenarios shows the likelihood of *Hyalella* population declines of at least 75, 90, 95, and 98 percent (fig. 4*B*). This range of declines encompasses declines that would be common under baseline conditions (75 percent) to declines that are severe enough to approximate a population extinction (98 percent). The choice of a 98 percent reduction as an quasi-extinction threshold was arbitrary, but probably reflects a decrease that is severe enough that even if the population did not die out completely, it would take a long time to rebound, absent immigration or extremely strong compensatory increases in survival and fecundity. The projections show that under baseline conditions, odds are even that a *Hyalella* population will experience at least 75 percent declines at some point over 6 generations. More severe declines greater than 90 percent are unlikely (5 percent risk) with almost no risk of 95 percent or greater declines in abundance. The risk curves are nearly flat from 0–0.8 µg/L, increase slightly from 0.8–1.2 µg/L, and increase sharply at greater than 1.2 µg/L cadmium. At 1 µg/L (1 times CCC) the risk of a 90-percent decline has increased to 12 percent and the risk of 95 percent or greater decrease still is 1 percent or less. Above 1 µg/L, the risk curves for severe population declines or extinction increase steeply, so that at sustained concentrations of 1.9 µg/L, severe declines are nearly certain and extinction (estimated as a >98 percent decline) is probable. At 1.9 µg/L, decreases in abundance of at least 95 percent are forecast with a 96-percent probability.

This 1.9 µg/L concentration is significant because it corresponds with several endpoints from the underlying toxicity test used in the simulations (Ingersoll and Kemble, 2001). In the 42-day toxicity test, the EC_{50} for reproduction and both the LOEC and EC_{20} values for mortality were about 1.9 µg/L, and the NOEC after 10-days exposure also was 1.9 µg/L. Using the raw data from Ingersoll's and Kemble's (2001) test, the EC_{20} for growth at 28 days was estimated through logistic regression at about 1.7 µg/L and for 42 days at >3.2 µg/L. NOEC, LOEC, and EC_{20} statistics from chronic test endpoints have been considered acceptably low levels of toxic effects by some authors (see section "Statistical Interpretation of Chronic Tests"). The results of the population forecasts indicate that if such "low-effect" concentrations were persistent, disproportionably severe declines or even local extinctions of *Hyalella* populations could potentially occur.

The third set of scenarios forecasted seasonal trends in *Hyalella* abundances through six generations (fig. 4*C*). Under the baseline scenario (natural population with no added cadmium stress), the modeling predicted generally increasing populations with seasonal abundance cycles of as much as 3 times. Over the six-generations forecast, the troughs (annual minima) in the abundance cycles increased by about 35 percent per generation. Under a scenario of episodic exposures to 1.2 times the CCC every third generation at the beginning of the reproductive season, the population also trended upward but only by about 15 percent per year. Under a scenario of constant exposure to 1 times the CCC, little or no growth in the population was predicted. Of the scenarios modeled, the episodic exposure scenario probably is the one most relevant to potential effects of cadmium to *Hyalella* under the magnitude, frequency, and duration of allowable criteria exceedences. The timing of the assumed exceedences mattered. Assuming that the exceedences occurred at the beginning of the reproductive season reduced reproductive potential more than did exceedences late in the reproductive season, or than exceedences during non-reproductive times of the year (not shown).

The seasonally cyclical abundance patterns produced by the model are qualitatively similar to patterns observed in seasonal field studies, although the patterns in the field studies were not as regular as in the model. (Cooper, 1965; Mathias, 1971). The model likely underestimates interannual variability. *Hyalella* densities measured over a 12-year period at a USGS

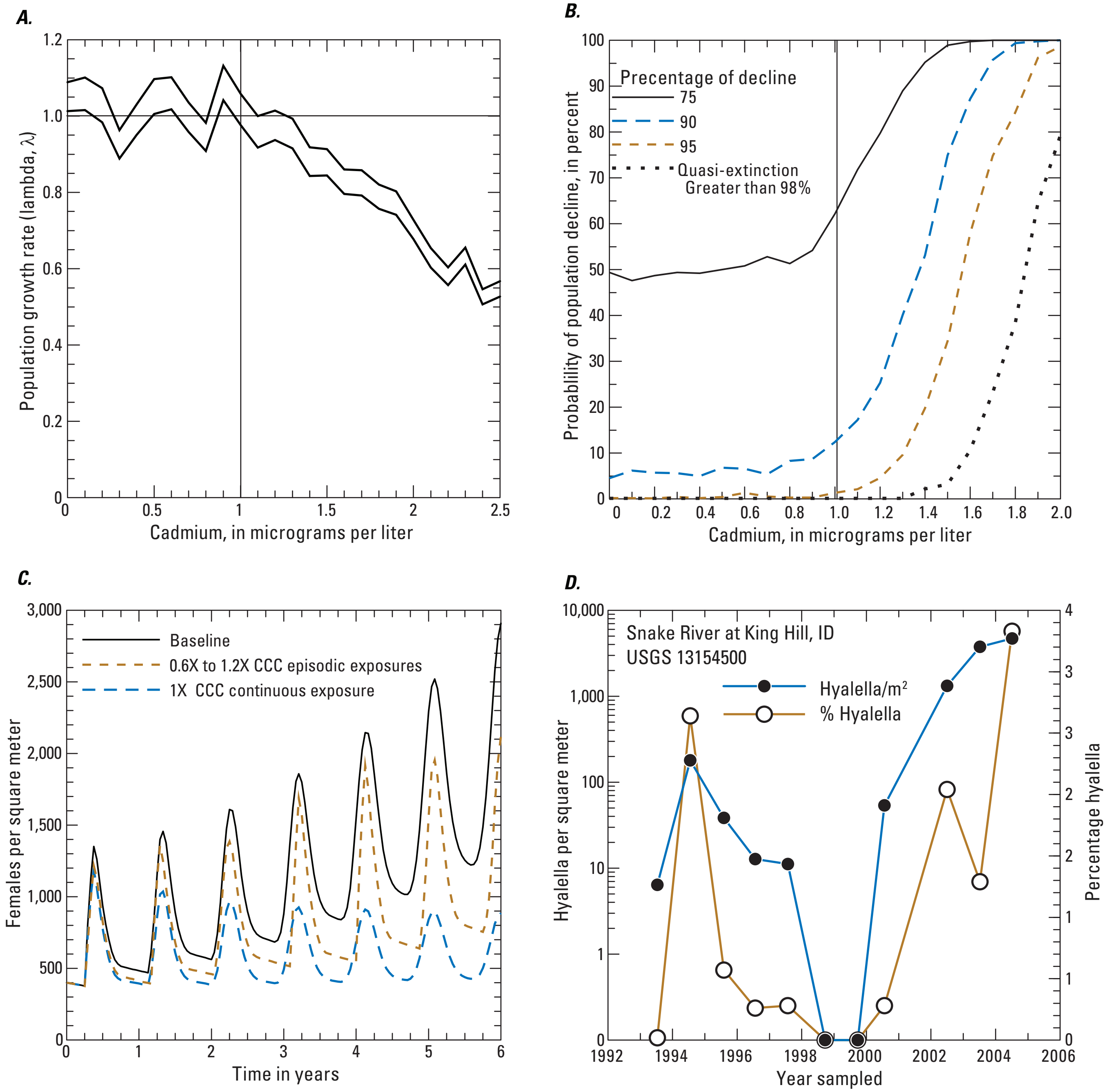


Figure 4. Risks to *Hyalella azteca* populations exposed to the cadmium chronic criterion. ***A.*** Population growth rate (lambda, λ) decline with increasing cadmium concentrations. Solid lines show λ for using min. and max. survival and fecundity estimates. Horizontal and vertical dashed lines show threshold for population decline (λ<1) and chronic criterion respectively; ***B.*** Risks of population declines up to the specified percentages occurring at some point during the 6-year population projections (mean probabilities following 500 replicate simulations); ***C.*** Projected *Hyalella azteca* densities (mean of 500 replicates) under scenarios of baseline conditions; exposure to 0.6X the chronic criterion concentrations (CCC) with episodic increases to 1.2X CCC at the beginning of the reproductive season for one 15-day time steps, once every 3 years ; and continuous exposure to 1.0X the chronic criterion concentration; ***D.*** Measured *Hyalella azteca* densities over a 12-year period or record at a USGS NAWQA biological trends monitoring station.

NAWQA long-term biomonitoring site in the Snake River, Idaho, ranged from 0 to 4707 per m^2 (fig. 4*D*). Although some of this apparent variability undoubtedly reflects measurement error, these USGS NAWQA data probably were close to the best case in regard to minimizing uncertainty owing to differences in sampling and processing. All samples shown in figure 4*D* were collected using the same methods, equipment, and all samples were analyzed by the same laboratory, which for the most part used the same taxonomists during this time. Most of the samples also were collected by the same individuals and from the same riffle year-to-year.

The population modeling forecasts of the effects of chronic cadmium exposure to *Hyalella azteca* populations could under- or over-predict the effects for several reasons. First, the toxic effects of cadmium are not constant across populations or conditions. The hardness-adjusted mortality observed by Ingersoll and Kemble (2001) was lower than mortalities observed in some other long-term exposures of *Hyalella* to cadmium (table 11). Although the tests resulting in higher mortalities were not used directly in the simulations because reproductive endpoints were not reported, this suggests that some populations could be more vulnerable to cadmium than those used in the present simulations.

Second, the modeling assumed that the growth *Hyalella* populations were density-independent. Density-dependence refers to the fact that populations cannot increase without limits, because expanding populations are all eventually self-limiting. Population growth is limited by density-dependent factors such as food and space limitations, waste buildup, disease transmission, or large populations may attract more predators. In population modeling, some mathematical function is applied to either or both fecundity or survival so that as densities increase or decrease, survival and fecundity rates also increase or decrease in compensation. Mathematically, the density-dependent compensatory changes in vital rates tends to lessen the effects of toxicants in population forecasts. For long-term population forecasts, assuming density-independence, as was the case here, is unrealistic. However, for shorter term forecasts, reasonable projections and comparisons can be made while ignoring density dependence (Barnthouse, 1993; Caswell, 2001). Further, even among intensely studied populations, the ability to reconstruct from data the density-dependence relations governing the natural dynamics may be limited because of changing environmental conditions. Under moderate or weak density dependence, the computed extinction risks are lower than when density dependence is included in the model. When available data sets are insufficient for reconstructing reliable measurements of density dependence, conservative estimates of extinction probabilities can be made from models that simply omit density dependence (Ginzburg and others, 1990). In *Hyalella* , weak density-dependent reduction in broodsizes has been observed in laboratory cultures (Borgmann and others, 1989b), however Strong (1972) observed that over the ranges encountered in nature, gross reproductive rates of individual *Hyalella* were uninfluenced by density. Strong (1972) suggested that resource-limited density-dependence is expected only in environments free of vertebrate predators. These predators appear generally able to crop *Hyalella* populations back to levels where resources do not limit reproduction. Predation that was biased against larger amphipods was implicated as the selective force shaping the small size, slow individual growth, and hence reduced fecundities of some populations (Strong, 1972).

Some of the field studies were consistent with the idea that density-dependent compensation probably occurs with *Hyalella* populations. Cooper (1965) observed that *Hyalella* populations in a Michigan lake could tolerate as much as about 28 percent mortality per week. This mortality rate is considerably higher than the mortality rates used here (about 5 percent mortality per week) that resulted in a slowly growing baseline population (fig. 4*C*). Strong (1972) observed periodic mass mortality of *Hyalella* in a hot springs population that were apparently due to intrusions of noxious subsurface water. The periodic mass mortalities were frequent enough that the hot springs population spent most of its time below resource-determined carrying capacity (Strong, 1972). This recovery from acutely toxic episodes by an actual population complements the episodic chronic exposures to a model population (fig. 4*C*). Finally, the dramatic increase in *Hyalella* density in Snake River samples from 1999 to 2003 of 0 to 3,766 animals/m^2 (fig. 4*D*) indicates that *Hyalella* populations can grow quickly under favorable environmental conditions. High flows in the Snake River during the late 1990s may have reduced and displaced *Hyalella*, and in following years the flows were low and stable and accompanied by lush macrophyte beds.

A regression of mean *Hyalella* brood sizes and abundances that were reported by Gibbons and Mackie (1991) did result in a significant negative relation, suggesting density-dependent compensation in fecundity was present. This relation was used with the Ricker recruitment function to consider the effects of density dependent compensation in the population modeling. The Ricker function can be used to model situations where the sharing of resources among individuals is roughly equal. Therefore, as the population density becomes very high, no individual gets enough resources, and at high densities, all individuals suffer equally. This is sometimes described as “scramble competition” (Ricker, 1975; Spencer and Ferson, 1997b; Caswell, 2001). The Ricker function is $P = exp(-\beta A)$ where P is the multiplier applied to maximum fecundity, A is the abundance in females per square meter, and β is a measure of the strength of density dependence in square meters per female. For fecundity, β was estimated by regression as -0.001046 with 95-percent confidence intervals of -0.00138 to -0.00071). Extinction risk forecasts were sensitive to the density-dependence function. In figure 4*B*, the risk curves for severe population decline or quasi-extinction begin to bend upward at around 0.9 µg/L and by about 2.0 µg/L, severe declines are almost inevitable and quasi-extinction is probable. If density-dependent compensation in fecundity is incorporated into the model as described above, the risk curves only begin to bend upward at around 1.6 µg/L. By 2.0 µg/L, instead of an 80-percent risk of a 98-percent population decline, there was about an 80-percent risk of a 75-percent decline and very little risk (<1 percent) of severe declines of greater than 90 percent (density-dependent results are not graphed).

However, the physiological mechanisms responsible for reproductive failure from cadmium exposure and reduced reproduction associated with high densities in the absence of toxic effects are different. Reduced reproduction in natural environments at high densities is likely related to factors such as reduced growth from competition for food or space, or from selective predation on large reproductively fit individuals. These factors would not seem relevant to reduced reproduction occurring in toxicity tests that are conducted in environments with ample food and no predation. No theoretical or experimental basis was obvious for why density-dependent compensatory increases in fecundity should be expected to offset cadmium-caused reproductive impairment equally. Because giving density-dependence and toxicity the same mathematical weight in adjusting the vital rates in the population simulations did not seem warranted, and because no basis was apparent for some other weighting, only the results of the density-independent population simulations are presented in figure 4.

However, the lack of experimental testing of density dependence of *Hyalella* in combination with exposures to toxicants and the inability to describe mathematically density-dependent compensation in polluted environments does mean that none exists. Such compensation seems most plausible in episodic exposure scenarios. It follows that under these scenarios, the actual effects of cadmium exposures should be less severe than the density independent simulations carried out here.

In summary, the empirical field data and population modeling suggest that for *Hyalella azteca*, a species that is both the most sensitive species tested to date with chronic cadmium exposures as well as a species that by design is not necessarily protected by the chronic cadmium criteria, populations could be limited by cadmium at the chronic criterion. These population limitations are unlikely to be of a magnitude that would result in local extinctions. In montane stream food webs, *Hyalella* probably are not dominant enough for invertivores such as salmonids, sculpin, dace, or salamanders to depend on them as a food source. However, in springbrooks (and presumably other streams) that have stable, low velocity flows, *Hyalella* can dominate benthic macroinvertebrate communities. A crash of a prey species in such a system could have large effects on predatory fish populations. *Hyalella* and other amphipods could be important components of lake food chains, and a contaminant-caused population crash could have important adverse ecological effects.

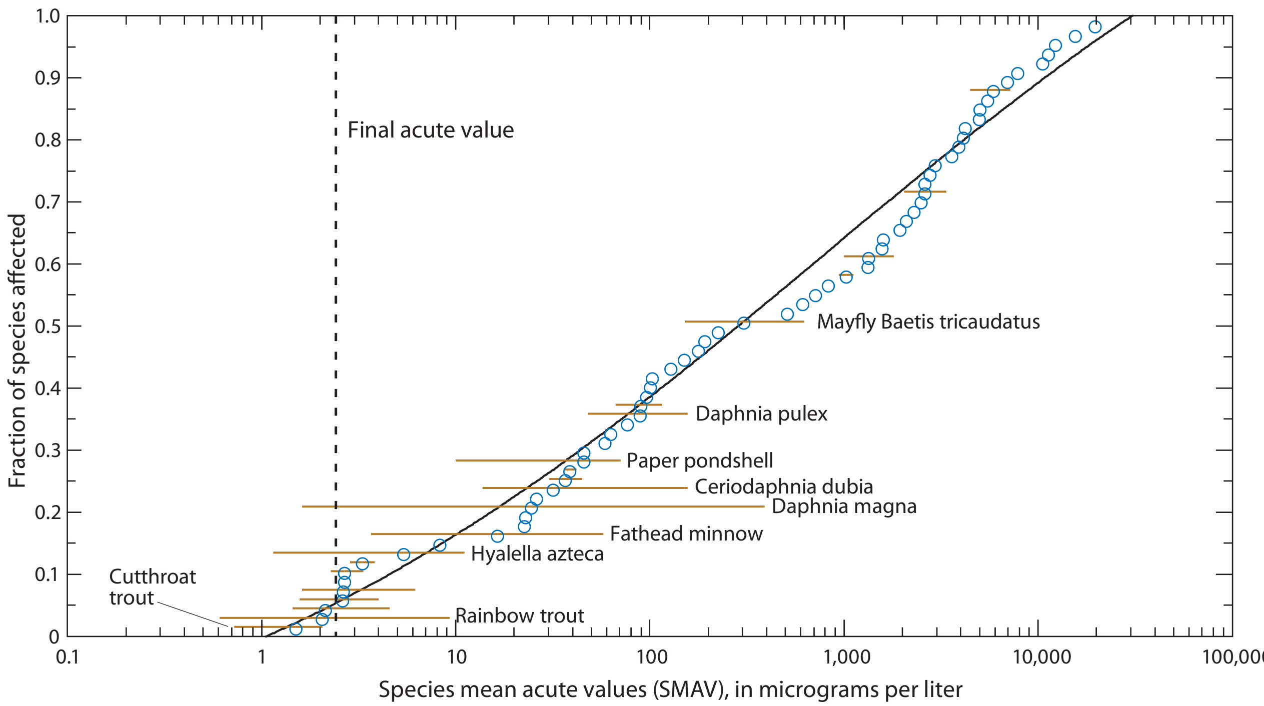
Field Validation

The goals of the Clean Water Act include maintaining and restoring the biological, chemical, and physical integrity of the waters of the United States. Because the goal of chemical integrity is central to the reasons for developing aquatic life criteria, it seems logical to evaluate the acceptability of chemical criteria by whether water allowed to have chemicals at criteria conditions would compromise the "biological integrity" of waters. Frey (1977) defined waters that have biological integrity as those that have

> *"… a species composition, diversity, and functional organization comparable to that of natural habitats of the region."*

Frey's definition has been widely used in the biological condition of waters including development of indexes of biological integrity (IBIs). IBIs are empirical additive indexes consisting of various metrics describing an assemblage such as species composition, diversity, functional organization, presence or absence of pollution sensitive or tolerant taxa (Karr, 1991). This definition is used in this field validation section to evaluate whether apparent effects that co-occurred with cadmium concentrations at or less than criteria concentrations were "unacceptable."

EPA's National guidelines were developed on the premise that effects that occur on a species in appropriate laboratory tests generally will occur on the same species in comparable field situations (Stephan and others, 1985). When enough species test values are available to describe a species sensitivity distribution, the distribution is at least implicitly considered to represent the distribution of sensitivities in natural ecosystems. The species sensitivity distribution of species mean values can be visualized as a cumulative distribution function (fig. 5). Assuming that the chronic distribution curve approximates responses in field settings, at sustained cadmium concentrations of less than 0.4 µg/L, less than 5 percent of species are predicted to be adversely affected; at about 3 µg/L, 50 percent are predicted to be adversely affected, and above about 25 µg/L, 90 percent of all species are predicted to be adversely affected. Comparisons to field studies can help evaluate if these predicted species responses seem to represent "unacceptable" or indiscernible effects.



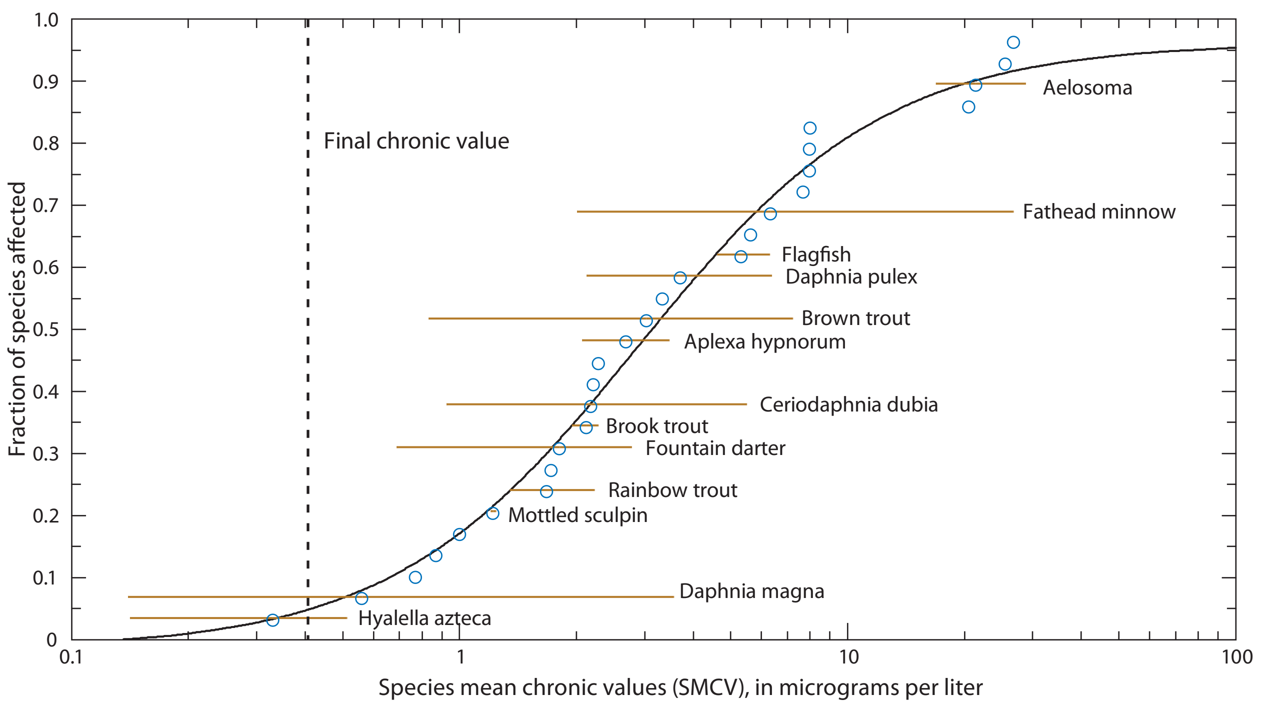


Figure 5. Species sensitivity distributions of cadmium values showing species mean values (open circles), 5th percentile final acute and chronic values (vertical dashed lines), and their fitted logistic distribution curves. All values adjusted to a hardness of 50 milligrams per liter $CaCO_3$.

An approach to determine if criteria would protect natural aquatic communities is to compare the occurrence of apparent instream effects to criteria exceedences at that location. The absence of apparent effects at sites that do not often exceed criteria, or the presence of apparent effects at those sites where criteria are frequently exceeded would support the relevance of the criteria. The converse of either would call in question the protectiveness or relevance of criteria.

Limitations, however, are in the ability to “validate” criteria in this manner. “Validation” is the process of comparing the overall result or output of a method, toxicity test, or model with observed effects in natural systems (Cairns and others, 1995). In this case, the criteria that resulted from many toxicity tests, calculations and decisions are compared to field observations of ambient biological conditions and criteria exceedences. Toxicity test results can never truly be validated or refuted based on field comparisons and vice versa. Toxicity tests are conducted in ecologically unrealistic environments, where all variables other than that being tested are held constant. By this means, causality can be assigned to the test variable. In field conditions, multiple biological, physical, and chemical variables interact. With many variables changing, strict causality may be difficult to establish.

The best possible outcome when seeking to “validate” toxicity test results with field observations is that similar types of apparent effects are observed in the field as resulted in toxicity tests at similar concentrations (Chapman, 1995; Clements and Kiffney, 1996; de Vlaming and Norberg-King, 1999; Clements and others, 2002; Suter and others, 2002a). For example, if concentrations of cadmium caused mortality to cutthroat trout relative to controls in toxicity tests, and if similar concentrations were measured in a stream location with low abundance of cutthroat trout relative to abundance at reference conditions with low cadmium concentrations but otherwise similar habitat conditions, then the toxicity-test predictions would be considered “validated.” Although powerful, there are limitations to these comparisons. First, the absence of instream effects can never be proven; effects may be present that are too subtle to detect by field surveys. Second, proving that the presence of apparent effects was caused by the stressor of interest may be difficult, because other unmeasured or correlated variables could be the cause. Thus, here biological effects in lakes and streams are qualified as “apparent” effects.

Cadmium Whole-Lake Ecosystem Experiment

In the late 1980s, Canadian researchers began a remarkable series of whole-lake ecosystem studies with cadmium at the Experimental Lakes Area of western Ontario. The purpose of the studies was to gain knowledge on the behavior and effects of metals in real aquatic ecosystems. At the time, most information on the effects of cadmium on aquatic organisms was based on single-species laboratory tests and field observations. In contrast, Canadian water-quality guidelines were intended for ecosystems, with a national goal of no observable effects on aquatic life over the long term with particular attention to the protection of sensitive species and life stages. To investigate whether the chronic water-quality guidelines that were based on single-species tests were protective in the wild, a 6-year study of the fate and effects of experimental cadmium enrichment on Lake 382 of the Experimental Lakes Area was undertaken (Schindler, 1988; Lawrence and others, 1996; Malley, 1996). The whole-lake studies are particularly relevant for the present analysis because (1) the extrapolation of results of single-species laboratory tests to the wild is a question; (2) the concentrations of cadmium actually achieved in Lake 382 were close to the present hardness-adjusted chronic cadmium criteria values (0.153 to 0.185 µg/L, which were about 1.25 to 1.5 times the chronic criterion 0.12 µg/L calculated for a hardness of 8 mg/L); and (3) prior to the cadmium additions, Lake 382 was in a near-pristine condition so that changes in aquatic populations and communities from baseline or reference-lake conditions could be attributed to the experimental cadmium additions. The last point is in contrast to field surveys of polluted systems because these tend to have co-occurring mixtures of metals and other disturbances, greatly reducing the ability to assign causes to effects.

Background cadmium concentrations in Lake 382 before cadmium additions were about 0.0016 µg/L (1.6 ng/L). Cadmium was added to the epilimnion during the ice-free seasons such that concentrations were gradually ramped up for the first 3 years of additions, and were maintained through the ice-free season at 0.153 to 0.185 µg/L for 3 years. When the lake was ice covered, no additions were made and concentrations decreased, so annual average concentrations were about 33–60 percent of the concentrations maintained during the ice-free season. Recovery was monitored for an additional 3 years after the cadmium additions ended (Lawrence and others, 1996).

Previous ecosystem manipulations with various stressors indicated that among the earliest of responses to stress are changes in species composition of small, rapidly-reproducing species with wide dispersal powers such as phytoplankton, and the disappearance of sensitive organisms from aquatic communities (Schindler, 1987). However, during or following the near-criterion cadmium additions, no overt effects were detected on phytoplankton, zooplankton, macrobenthos, or fish assemblages. Population monitoring of crayfish (*Orconectes virilis*), lake trout (*Salvelinus namaycush*), white sucker (*Catostomus commersoni*), and minnows detected no differences from baseline or reference conditions attributable to cadmium treatments.

Bioaccumulation occurred at every trophic level. Concentrations stabilized in short-lived *Hyalella* at 17X background but in both the long-lived floater mussel *Pyganodon grandis* (whole body) and lake trout (kidney) concentrations reached 5–9X and 4–5X, respectively, after 6 years but were still trending upward when cadmium additions were discontinued. Except for metallothionein induction, biomarkers of stress (for example, biochemical, histological, physiological, and behavioral) monitored in the large fish and floater mussels were unremarkable. Malley (1996) speculated that if the experiment had run longer, adverse effects likely would have later become evident, based on the increasing trends in fish tissue residues and sediment contamination.

The monitoring of *Hyalella azteca* and lake trout populations in Lake 382 are of particular interest because adverse effects were observed with these species in laboratory exposures at concentrations less than the chronic criterion derived in this report. Although the inability to detect effects to *Hyalella* populations does not demonstrate that no-effects occurred, it does indicate that no population crash of *Hyalella* occurred. With lake trout, Scherer and others' (1997) "black box" experiment found that in a tank with cadmium exposed lake trout and rainbow trout fingerlings, more rainbow trout remained at the end of the test than in the control tank. From this they inferred that cadmium exposures reduced the ability of lake trout to capture prey (Scherer and others, 1997). The foreseeable implications of an impaired ability to capture prey include reduced growth or condition factor, which in turn implies less energy available for reproduction and lower fat reserves for winter, resulting in expected population declines and a shift in age class structure. Although none of these overt effects were obvious in cadmium-dosed Lake 382, this pattern of effect was clearly documented through similar monitoring efforts in acidification and other studies at the Experimental Lakes Area by the same team of researchers (Schindler and others, 1985; Schindler, 1987, 1988). Therefore, it appears unlikely that the lake trout exposed to elevated cadmium exposures in the wild for 6+ years in Lake 382 had appreciably impaired ability to capture food. These differences between tank experiments and ecosystem experiments are parallel to those observed with zooplankton. In limnocorral mesocosm exposures in preparation for the whole-lake experiments, marked reductions in the cladocerans *Daphnia galeata* and *Holopedium gibberum* were observed following exposure for 3 weeks to 0.2 µg/L cadmium, yet these predictions of adverse effects in resident cladocerans in Lake 382 were not borne out (Lawrence and Holoka, 1991; Malley, 1996).

Co-Occurrence of Cadmium Criterion Exceedences and Apparent Effects in Streams

The vicinity of the Coeur d'Alene River, Idaho has recently been the focus of several environmental assessments, and large chemical and biological datasets have been collected in the study area in the last several years. Here, chemical and biological data over a gradient of conditions were compiled and matched to seek associations between criteria exceedences and apparent adverse effects. The primary dataset evaluated was from the USGS National Water Quality Assessment, Northern Rockies and Intermontane Valleys study unit "NROK data" (http://id.water.usgs.gov/nrok/index.html). Two main strengths of this dataset for comparing instream conditions to criteria exceedences were that (1) chemical, physical habitat, macroinvertebrate, and electrofishing collection methods were synoptic, clearly described, suitably rigorous, and consistent; and (2) sample sites were selected to include a range of conditions from nearly undisturbed reference sites to highly disturbed mining sites. Additional biological and chemical data from the IDEQ and fish surveys conducted by natural resource trustees as part of a natural resource damage assessment were also used. These data were originally compiled in Dillon and Mebane (2002) and were presented in more detail there.

Several biological metrics (that is biological endpoints that are expected to respond to a stressor in a predictable way) were compared to cadmium concentrations. Because an almost limitless number of biological endpoints could potentially be examined, the comparisons focused on six metrics that were expected to be sensitive to metals, and could be calculated with the available data: trout density, percent sculpins, a stream fish index of biological integrity (IBI), total invertebrate taxa richness, mayfly taxa richness, and density of metals intolerant taxa. Brief descriptions of each metric follow and the apparent responses associated with chronic cadmium criteria exceedence factors are shown in figure 6. Biological metrics values are plotted with cadmium chronic criterion unit exceedences where an exceedence factor

$$EF = \frac{Cd\ (\mu g/L)}{Cd\ CCC\ (\mu g/L)}$$

was calculated for the hardness the sample. If no sample hardness were available, the CCC was calculated with the average hardness for the location. The interpretations of these comparisons were limited to (1) if there was no adverse response apparent for a metric at exceedence factors less than 1.0, then sub-criterion cadmium concentrations did not cause an "unacceptable" adverse effect to that biological metric in this dataset; and (2) if there was an apparent adverse response threshold for a metric at an exceedence factor, then cadmium exceedences greater than that threshold could cause an "unacceptable" adverse effect to that biological metric in this dataset. However, in the latter case, since zinc also occurred at these sites in roughly equi-toxic proportions to cadmium, it would not be valid to assign causality for effects solely to cadmium since the effects could have been caused by zinc, their combination, or other factors.

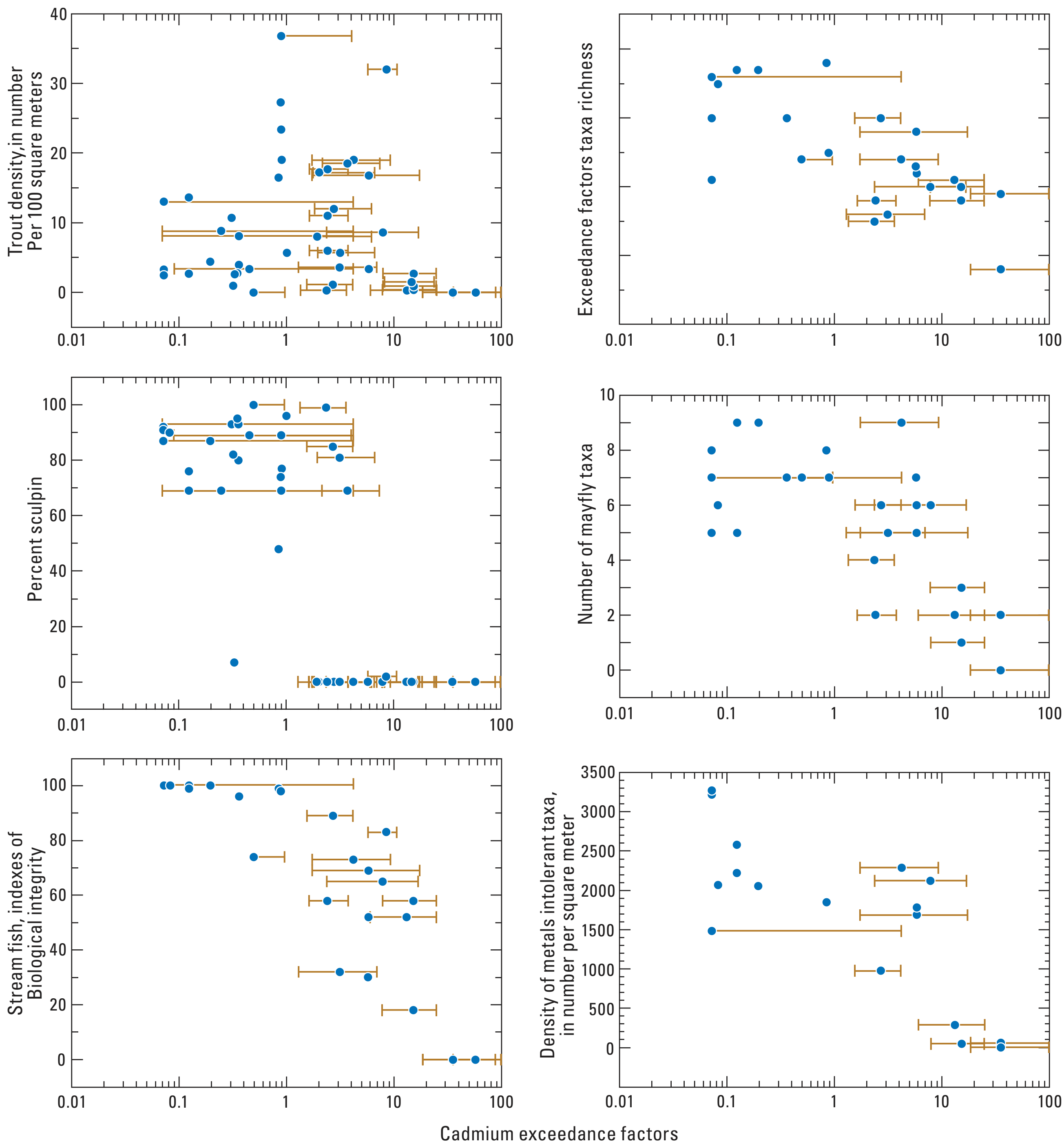


Figure 6. Comparison of measured cadmium chronic criterion exceedence factors, and corresponding instream biological condition metrics. Points indicate exceedence factors at the time of biological sampling, or for sites not sampled concurrently, median exceedence factors. Error bars show the 5th and 95th percentile exceedence factors for sites with multiple values. Exceedence factors calculated after adjusting the criterion for the sample or median hardness values for each site. Data from streams in the Coeur d'Alene and St. Regis River basins, Idaho and Montana (Dillon and Mebane, 2002).

Trout density—Salmonids were the most acutely sensitive organisms to cadmium in the available data, and were fairly sensitive in chronic exposures (lower one-half of the species sensitivity distribution, table 7; fig. 5). Thus, in locations with cadmium concentrations greater than those toxic in controlled testing, trout densities would be expected to be depressed. The chronic criteria are intended to delineate levels safe for sensitive organisms based on testing at the individual level of organization. In nature, the collective effects of toxicity to individuals should be reflected at the population level of organization. However, quantitative relations between trout populations and environmental quality are notoriously elusive. Relations between environmental variables and trout densities are often unclear because of interactions among water quality, habitat, species interactions, and management manipulations (Fausch and others, 1988; Rose, 2000; Mebane and others, 2003).

In the Coeur d'Alene dataset, trout densities were consistently low at locations with cadmium exceedence factors greater than about 10 (fig. 6). At sites with lower exceedence factors, densities were variable. At sites with exceedence factors less than 1.0 and lower trout densities than some other sites, the variability in densities was presumably owing to factors other than cadmium such as physical habitats. Whether an apparent peak in the trout densities at sites with cadmium exceedence factors of about 2 to 5 is real or an artifact of the dataset is unclear.

Percent sculpin—Sculpin were among the most sensitive organisms in both acute and chronic testing with cadmium (table 7). Sculpin are ubiquitous and often numerically dominant in mid-sized forest streams in the Pacific Northwest. Although they tend to become less dominant as streams increase in size to become rivers, they are almost always present and are usually common in unpolluted, coldwater rivers in the inland Northwest (Maret and MacCoy, 2002; Mebane, 2002b; Mebane and others, 2003). Sculpins and other small-bodied fish may be superior to large-bodied fish as "sentinel" species for detecting effects from discrete disturbances in open receiving water environments. Smaller fish species, such as cottids, exhibit limited mobility relative to many larger species and typically possess a smaller home range. Many small species also show territorial behavior, particularly in lotic systems. This characteristic increases the probability that a sentinel species will not move extensively, and the observed response of that species will more likely reflect the local environment in which it was caught. In addition, small fish species tend to be more numerous than larger, more predatory species, which facilitates sampling; they have a shorter life span and therefore show alterations in reproduction and growth faster than longer-lived species; and they are not subject to commercial or sport fishing (Gibbons and others, 1998; Munkittrick and McMaster, 2000).

Sculpin showed a remarkably consistent threshold response to elevated cadmium (fig. 6). Sculpin were abundant at almost all sites where the cadmium criterion seldom exceeded 2 times. Sculpin were almost completely extirpated at sites where the median cadmium concentrations exceeded about 2 times cadmium chronic criterion. Because cadmium and zinc occurred in this study area in roughly similar proportions (on a toxic unit basis) at many sites the extirpation of sculpin could have been caused by either cadmium or zinc. An examination of the raw data behind figure 6 showed that among the sites at which sculpin were absent and at which median zinc exceedence factors were <1.0, the lowest cadmium exceedence factor was 2.4.

Index of biological integrity —The index of biotic integrity (IBI) concept was developed to address the need for operational definitions of Clean Water Act terms such as "biological integrity" and "unreasonable degradation." The IBI was intended to provide a broadly based and ecologically sound tool to evaluate biological conditions in streams, incorporating many attributes of stream communities to evaluate human effects on a stream and its watershed. Those attributes cover the range of ecological levels from the individuals through population, community and ecosystem (Karr, 1991). An assumption behind a multimetric additive index such as the IBI is that at least some metrics would respond to most stressors. The IBI framework was used to develop an index for coldwater forest streams in Idaho that gauges a stream against an expectation based on minimal disturbance in the ecoregion. The IBI developed for fish communities in Idaho streams is an additive index consisting of the following six metrics: (1) coldwater native species; (2) percent coldwater individuals; (3) percent sensitive native individuals; (4) trout age classes; (5) sculpin age classes, and (6) catch per unit effort of coldwater individuals (Mebane, 2002b).

Most IBI scores at sites with median cadmium exceedence factors less than 1 were high, indicating the fish community composition at study sites that usually met cadmium criteria was similar to that at reference streams. IBI scores showed a general graded decrease with further increases in cadmium exceedence factors greater than 2.0.

Macroinvertebrate taxa richness—Total taxa richness measures are widely used in field assessments of streams and may be sensitive measures of ecosystem disturbance. Many studies have reported declines in taxa richness in response to elevated metals concentrations in streams (Clements and Kiffney, 1996; Clements and others, 2000; Fore, 2002; Kiffney and Clements, 2002; Mebane, 2002a; Maret and others, 2003). Carlisle and Clements (1999) found that in terms of sensitivity, variability, and statistical power, richness measures were superior to other commonly used metrics.

In the Coeur d'Alene vicinity data, a general pattern of declining invertebrate richness corresponding to increasing exceedences of cadmium criteria is apparent (fig. 6). The sites with the highest species richness all had exceedence factors of 1.0 or less, the sample with the highest richness (38 taxa) was collected from a site with an exceedence factor of about 1.0, and sites with exceedence factors of about 2.0 or more tended to have lower species richness. This suggests a possible exceedence factor response threshold between 1.0 and 2.0. Overall, the invertebrate richness and exceedence factor comparison indicates that at locations where the chronic cadmium criterion is not exceeded, few if any taxa would be lost owing to cadmium toxicity.

Mayfly taxa richness—This richness measure is limited to mayfly (Ephemeroptera) taxa. As a group, mayflies generally are sensitive to metals and fewer taxa tend to be found in areas with elevated metals, and some studies have reported that the effects of metals generally were greater on mayflies than other macroinvertebrate groups (Clements and Kiffney, 1996; Clements and others, 2000; Fore, 2002; Kiffney and Clements, 2002; Mebane, 2002a; Maret and others, 2003). Similarly to the overall taxa richness metric, a general pattern of declining mayfly richness corresponding to increasing exceedences of cadmium criteria is apparent. The highest number of mayfly taxa (9), we collected from a site with an exceedence factor of about 3 (fig. 6).

Metals-intolerant macroinvertebrate density—Several taxa were consistently sensitive to heavy metals in surveys of Colorado mountain streams (Clements and others, 2000; Fore, 2003). This metric measures the abundance of only these metals-intolerant taxa. Abundance is measured instead of taxa richness in this metric, because taxa richness reflects only presence of a taxon. The metric consists of the sum of the densities of the following 10 genera in 4 orders: the mayflies *Cinygmula*, *Drunella*, *Epeorus*, *Paraleptophlebia*, and *Rhithrogena*; the stoneflies *Skwala*, *Suwallia*, and *Sweltsa*; the caddisfly *Rhyacophila*, and the dipteran *Pericoma*. Conceptually, this metric should be more sensitive than taxa richness metrics because if a taxon is severely depressed but still present in reduced numbers, taxa richness counts will not reflect that. However, for the data analyzed in figure 6, this metric appears less sensitive than the richness measures. Densities of the 10 "metals-intolerant" taxa were >1,500 per square meter at sites with exceedence factors of about 8 or less (fig. 6).

In summary, for these streams which included a gradient of cadmium concentrations, at sites where the cadmium chronic criteria values derived in this report were seldom exceeded (EF <1.0) the macroinvertebrate and fish assemblages seemed to have "species composition, diversity, and functional organization comparable to that of natural habitats of the region." In other words, for these data biological integrity was unlikely to have been compromised owing to cadmium concentrations so long as the chronic criterion derived in this report was seldom exceeded.

A second case study of the co-occurrence of cadmium criterion exceedences and apparent effects in streams gives some insight to the effects of a single pulse of cadmium to a stream macroinvertebrate community. In April 1999, a pulse of cadmium of about 30X background concentrations and with a chronic criterion EF of about 2.6 and an acute criterion EF of about 1.0 was detected in Thompson Creek, Idaho (fig. 3). Because the episode was detected through weekly effluent monitoring, the duration of the pulse is assumed to have been longer than 1 day and shorter than 1 week. By August 1999 when the benthic macroinvertebrate community was sampled, few if any adverse effects were apparent. Overall macroinvertebrate density and taxa richness were similar at the exposed and upstream site. Densities and taxa richness of mayflies were somewhat higher in the downstream (exposed) sampling site than the upstream reference site, a pattern which also was observed at two companion sites located at similar elevations on a nearby stream that was not influenced by the release. The mayfly *Heptagenia* was only collected upstream of the exposed area in Thompson but was collected from both sites on the nearby unaffected stream (Chadwick Ecological Consultants, Inc., 2000). Because *Heptagenia* have been reported to be sensitive to elevated metals in streams (Clements and others, 2000; Fore, 2002; Clements, 2004), it is plausible that their absence from the survey was related to the earlier pulse. However, the distributions of macroinvertebrates in streams may be both spatially and temporally patchy, and the possibility that the correspondence between the distribution of *Heptagenia* and the cadmium pulse was merely coincidental cannot be excluded. Further, other mayflies in the family Heptageniidae that have also been considered sensitive to metals showed no such pattern. The Heptageniid mayflies *Cinygmula* and *Epeorus* were abundant upstream and downstream. No corresponding fish community data were collected in the August 1999 survey. Overall, this episode suggests that the macroinvertebrate communities in coldwater, mountain streams similar to Thompson Creek may be resilient to or recover quickly from, infrequent cadmium pulses with durations of few days and a magnitude as much as about 2.5 times the chronic criterion concentration and 1 times the acute criterion concentration.

Other comparisons of chronic criterion values and apparent effects values from ecosystem studies and field surveys are given in table 13. Some comparisons were ambiguous, but most of the comparisons indicated the onset of adverse effects occurred at cadmium concentrations greater than the relevant chronic criterion concentrations. Two studies reported clearly adverse effects at sub-criterion concentrations. Juvenile hatchery brook trout exposed to cadmium for 30 days at 50 percent of the chronic criterion had reduced success at capturing mayfly prey. Riddell and others (2005a) concluded that impaired foraging by affected individuals has the potential to alter the structure of aquatic communities by modifying energy flow and trophic interactions. When combined with increased activity levels, as observed in exposed brook trout, the energy budget of such organisms could become further compromised, with potentially negative effects on growth, survival, and reproduction (Riddell and others, 2005a, 2005b). Whether brook trout or other salmonids exposed to sub-criterion cadmium concentrations in field conditions also have impaired foraging is ambiguous. Brook trout were reasonably common in the co-occurrence surveys from the Coeur d'Alene vicinity described previously. No obvious reductions in several indictors of growth, survival, and reproduction were observed in trout surveys from sites at which cadmium concentrations were only moderately elevated (chronic exceedence factors less than about 2) (Dillon and Mebane, 2002; Maret and MacCoy, 2002). Lake trout populations that were exposed in the wild to higher cadmium chronic criterion exceedence factors than were the brook trout tested by Riddell and others (2005a, 2005b) showed no obvious decrease in condition or numbers (Malley, 1996; Vince Palace, Freshwater Research Institute, Winnipeg, oral commun., 2005). The absence of discernable effects of cadmium with lake trout in a long-term whole lake study and the presence of adverse effects in shorter term tests with brook trout could be related to the exposure and testing differences or species differences. Brook and lake trout are closely related, although their life histories differ markedly. Tests with lake trout in simplified food chains have shown adverse effects at similar cadmium concentrations causing the adverse effects in the brook trout studies (Behavioral Toxicity section; table 17). This suggests differing cadmium exposures or test differences during whole-lake or field studies may be more important than species differences for resolving the apparently contradictory results with brook trout and lake trout.

In summary, the "field validation" comparisons showed mostly good agreement between the laboratory-based predictions and effects observed in the field surveys or ecosystem experiments (fig. 7). Adverse effects at concentrations lower than the calculated chronic criterion were reported from small scale, simplified model ecosystem experiments. In these experiments, adverse behavioral effects were observed with two species in the genus *Salvelinus,* brook trout and lake trout. In a large-scale ecosystem study and field surveys, no obvious adverse effects to lake trout or brook trout were observed at cadmium chronic criterion concentrations. The disparity between effects observed in small-scale or large-scale ecosystem experiments or surveys could be real or artifacts of differing study designs and data types.

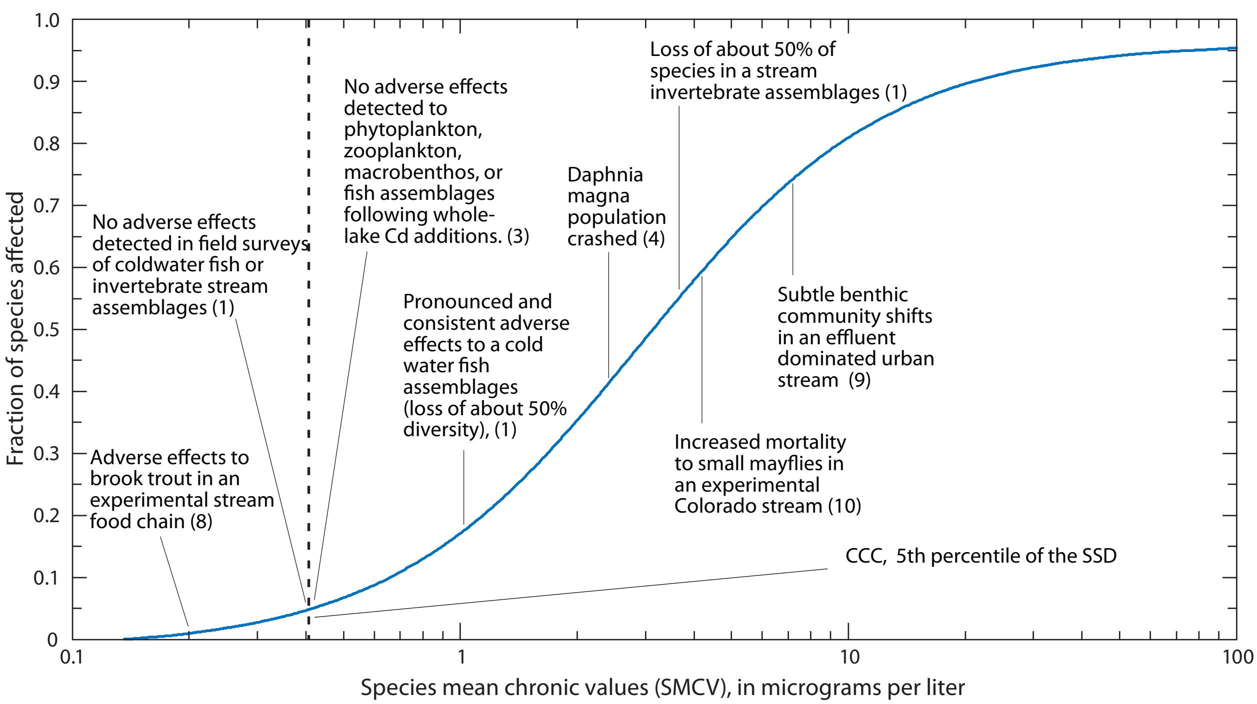


Figure 7. Comparison of cadmium effects concentrations estimated from ecosystem studies or field assessments to proportions of adversely affected species based on a species-sensitivity distribution (SSD) of laboratory-derived species mean chronic values (SMCVs). Solid curved line illustrates a SSD fitted from SMCVs using logistic regression; dashed line indicates chronic criterion concentration (CCC), numbers in parentheses refer to studies described in table 13. All values were adjusted to a hardness of 50 milligrams per liter $CaCO_3$.

Table 13. Comparison of chronic criterion values with results of ecosystem studies and field surveys.

[Underlined values are lower than corresponding chronic criterion values. Hardness as mg/L $CaCO_3$, all other concentrations are for measured values in microgram per liter. **Abbreviations:** Cd, cadmium; Cu, copper; Pb, lead; L, liter; m, meter; NOEC, no observed effect concentration; LOEC, lowest observed effect concentration; CCC, criterion continuous concentration. **Symbols:** µg/L, microgram per liter]

Study type	Study location	Exposure duration	Method notes	Effects/Notes	Hardness (mg/L as $CaCO_3$)	Lowest adverse effect value	Corresponding chronic criterion value	Reference
Field assessments	Northern Idaho and Western Montana	Indefinite	Co-occurrence comparisons	Extirpation of sculpin (*Cottus* sp.) from all sites where median cadmium concentration exceeded ≈ 2.4 times chronic criterion. Value listed was lowest concentration with an associated adverse effect (sculpin extirpated), which did not also have zinc criterion exceedences	64	1.20	0.44	Dillon and Mebane, 2002
Field assessments	Western Kentucky	Indefinite	Co-occurrence comparisons	Reduced macroinvertebrate taxa richness and density, particularly for mayflies. Cd body burden highest in the least resistant taxa (*Stenonema* mayflies). LOEC range shown is the mean and estimated maximum Cd concentrations (mean plus S.D. of 0.55 µg/L). Mean Cu concentrations ~0.5 times CCC and mean Pb concentrations ~1 times CCC, suggesting that the combined exposures may have influenced the observed effects	63	<u>.32</u> to .87	.44	Birge and others, 2000
Field assessments	Northwestern Québec, Canada	Indefinite	Co-occurrence comparisons sediment-water interface	Floater mussel (*Pyganodon grandis*) population status was negatively correlated with modeled free-cadmium ion concentrations at the water-sediment interface. Cd concentrations in water were estimated from sediment core extracts using a geochemical model. Effects of Cd were confounded by variable temperatures, other natural factors, and marked decreases in metal contamination over the 10 years prior to sampling	15	<u>.07</u>	.19	Perceval and others, 2004
Lake experiment	Experimental Lakes Area, W. Ontario, Canada	6 years	Epilimni-ondosed during ice-free season	No overt effects detected on phytoplankton, zooplankton, macrobenthos, or fish assemblages. Bioaccumulation occurred at every trophic level. Concentrations stabilized in short-lived *Hyalella* at 17 times background but in both the long-lived floater mussel *Pyganodon grandis* (whole body) and lake trout (kidney) concentrations both reached 9 times after 6-years but were still trending upward. Increasing trends of fish bioaccumulation and sediment contamination suggesting effects might eventually occur in experiment had run longer. Cd value is the mean of the concentrations maintained during the final two ice-free seasons	8	.19	.12	Lawrence and others, 1996; Malley, 1996

Table 13. Comparison of chronic criterion values with results of ecosystem studies and field surveys.—Continued

Study type	Study location	Exposure duration	Method notes	Effects/Notes	Hardness (mg/L as $CaCO_3$)	Lowest adverse effect value	Corresponding chronic criterion value	Reference
Lentic mesocosm	Burlington, Ontario (Lake Ontario water)	250 days	4.5 m deep × 1 m wide columns	*Daphnia* population crashed at 9 weeks, apparently could adapt to shorter exposures. No effect at 1 µg/L	130	4.70	0.69	Borgmann and others, 1989a
Lentic mesocosm	Experimental Lakes Area, W. Ontario	14 days	In situ continuous flow vessels	Decrease in abundance of cladocerans *Daphnia galeata* and *Holopedium gibberum*	8	.20	.12	Lawrence and Holoka, 1991
Pond	Blacksburg, Virginia	28 days	Suspended colonization substrates	Decrease in protozoan assemblage colonization (species richness). NOEC 0.4 and LOEC 1.4 µg/L	70	.75	.47	Niederlehner and others, 1985
Stream mesocosm	Saskatoon, Saskatchewan, Canada	30 days	Recirculating stream channels	Juvenile brook trout success at capturing mayfly prey impaired. Activity by stonefly *Kogotus nonus* nymphs decreased. Mayfly *Baetis tricaudatus* drift increased	156	.4	.77	Riddell and others, 2005a
Stream mesocosm	Saskatoon, Saskatchewan, Canada	30 days	Recirculating stream channels	Reduced condition factor in juvenile brook trout. Brook trout prey preference shifted from mayflies to less motile midges	156	.4	.77	Riddell and others, 2005b
Stream mesocosm	Denton, Texas	10 days	Once-through, effluent dominated stream channels	Subtle benthic community shifts (increase in Chironomid abundance, decrease in aquatic moth larvae)	146	15	.74	Brooks and others, 2004
Stream mesocosm	Fort Collins, Colorado	10 days	Once-through stream channels	Increased mortality to smaller individuals of the mayflies *Baetis tricaudatus (Baetidae), Ephemerella infrequens* (Ephemerellidae), and *Rhithrogena hageni* (Heptageniidae), and the stonefly *Pteronarcella badia* (Pteronarcyidae)	29	3.20	.27	Kiffney and Clements, 1996
Stream mesocosm	Aiken, South Carolina		91.5-m-long stream	Periphyton and invertebrate community structure affected. No effects to free-ranging fish observed, reported value is the NOEC	9.1	.5	.13	Versteeg and others, 1999
Stream microcosm	Glen Lyn, Virginia		20-L volume, 1.5-m-long stream	No observed effects on benthic microbial and invertebrate communities	57	2.5	.41	Versteeg and others, 1999

Cadmium and Threatened and Endangered Species

The National guidelines for developing criteria include assumptions that species can tolerate some adverse effects, and that populations and ecosystems have resiliency and can probably recover from periodic adverse effects (Stephan and others, 1985, p. 8–12). These assumptions may not hold in cases of especially vulnerable species that are at risk of extinction through most of their range and that have less capacity to withstand or recover from additional stressors. These species include those listed as threatened or endangered under the Endangered Species Act (ESA). The goals of the ESA include conserving the ecosystems that endangered species and threatened species depend, to conserve those endangered and threatened species, and promoting their recovery. It is necessary that aquatic life criteria for cadmium or any substance be sufficiently protective that the criteria would not allow conditions that impede the recovery of threatened or endangered species.

Risks of direct effects of cadmium on threatened or endangered species were evaluated using a simple risk quotient approach (table 14). In this approach, the highest no-observed-effect-concentration (NOEC) available for a species is the denominator, and the corresponding chronic criterion is the numerator. If the quotient is less than one, the criterion is presumed to be adequately protective, and if the quotient is greater than one, the criterion is presumably insufficiently protective (although the biological effect of exceeding a no-observed-effect value is admittedly ambiguous).

Chronic test results were available for 3 of the 11 threatened or endangered aquatic species that occur in Idaho. For most species for which no direct information is available, estimates need to be based on one or more taxonomic or toxicological surrogate species. For sockeye salmon and the Snake River physa, surrogate species were considered to be closely enough related that estimated cadmium NOECs were taken directly from the surrogate species, without any further adjustments. For white sturgeon, a two-step estimate of a cadmium NOEC was made. The relative sensitivity of three other sturgeon species and rainbow trout to copper, "a representative metal," were compared (Dwyer and others, 2005b). The most sensitive sturgeon response was 0.8 times as sensitive as rainbow trout to copper. The mean rainbow trout cadmium NOEC was then multiplied by this rainbow-sturgeon sensitivity factor to obtain a proportional estimate of white sturgeon sensitivity to chronic cadmium exposures. In comparative testing of 17 diverse threatened or endangered species and standard toxicity test fish species, rainbow trout were consistently the most sensitive of the standard test fish species, and were the best estimate of sensitivity of cold- or warm-water threatened or endangered species (Sappington and others, 2001; Dwyer and others, 2005b). The rainbow trout was equal to or more sensitive than listed and related species 81 percent of the time. Of the 19 percent of test results with listed species and rainbow trout in which the listed species were more sensitive, the geometric mean of the sensitivity factor between the rainbow trout and listed species was 0.63 (Dwyer and others, 2005b). More conservative factors than the geometric mean can be determined using variance estimates (0.46 based on 1 standard deviation (SD) of the mean and 0.33 based on 2 standard deviations of the mean). Using the most conservative of the sensitivity factors and assuming that the rainbow trout NOECs and LC50s are approximately proportional, in table 14 the rainbow trout NOEC is multiplied by 0.33 to estimate the low range of potential NOECs for snails (except for those in the families Physidae and Hydrobiidae, discussed separately). These potential NOEC values are shown as "greater than" values because the sensitivity factor of 0.33 is based on an extreme estimate of sensitivity differences, rather than a central tendency estimate. In a normally distributed population, the mean minus 2 standard deviations corresponds to about the 2.5th percentile. Thus about 97.5 percent of the values would be greater than a value based on the mean minus 2 standard deviations. Using the most conservative factor as described, the approximate NOEC would be about equal to the CCC (0.37 versus 0.38 µg/L, respectively). Using less conservative sensitivity factors of 0.46 based on 1 standard deviation of the mean, or 0.63 based on the geometric mean of the differences in the sensitivity factor between the rainbow trout and listed species, would result in NOEC estimates that are greater than the CCC (0.5 or 0.7 µg/L, respectively).

For Hydrobiidae snails, NOEC estimates also were made using interspecies correlation between fathead minnow (surrogate species) and *Amnicola* (predicted species) using EPA's ICE model (Asfaw and others, 2004) with the fathead minnow mean chronic NOEC. *Amnicola* was a factor 14 more acutely resistant than fathead minnow to 5 chemicals. Fathead species mean NOEC was 4.5 µg/L at hardness of 50 mg/L. Assuming proportional acute and chronic responses, 4.5 µg/L × 14 = 63 µg/L.

This latter estimate for Hydrobiidae snails probably is exorbitantly high, based on species-sensitivity distributions (fig. 4) and on chronic values reported with Physid or Lymnaeaid snails. The estimate is included simply to explore some various approaches to estimating chronic values for untested species. From the information reviewed here, the best estimate of a chronic NOEC value for all Snake River snails is probably around 0.5 to 0.7 µg/L at a hardness of 50 mg/L.

Table 14. Comparison of estimated no-effect concentrations for threatened and endangered species occurring in Idaho to chronic criteria concentrations.

[**Abbreviations:** NOEC, no observed effect concentration; CCC, criterion continuous concentration; LC50, 50 percent of life cycle; –, not applicable. **Symbols:** mg/L, milligram per liter; µg/L, microgram per liter; <, less than; >, greater than]

Species of interest	Family	Hardness (mg/L)	Test NOEC (µg/L)	CCC	Hazard quotient (CCC/NOEC)	NOEC basis	Family of surrogate	Surrogate species
Bull trout, *Salvelinus confluentus*	Salmonidae	29	0.38	0.27	0.71	Direct	–	–
Chinook salmon, *Oncorhynchus tshawytscha*	Salmonidae	25	.96	.25	.26	Direct	–	–
Steelhead, *Oncorhynchus mykiss*[1]	Salmonidae	50	1.2	.38	.32	Direct	–	–
Sockeye salmon, *Oncorhynchus nerka*	Salmonidae	50	1.2	.38	.32	Surrogate	Salmonidae	*Oncorhynchus mykiss*
White sturgeon, *Acipenser transmontanus*[2]	Acipenseridae	50	.9	.38	.4	Surrogate	Acipenseridae	Atlantic sturgeon *(Acipenser oxyrhynchus),* Shortnose sturgeon *(Acipenser brevirostrum),* Shovelnose sturgeon *(Scaphirhynchus platorynchus)*
Bliss Rapids snail, *Taylorconcha serpenticola*[3]	Hydrobiidae	50	63	.38	.006	Surrogate	Hydrobiidae	Spire snail, *Amnicola sp.*
Idaho springsnail, *Pyrgulopsis idahoensis*[3]	Hydrobiidae	50	63	.38	.006	Surrogate	Hydrobiidae	Spire snail, *Amnicola sp.*
Bruneau Hot Springs snail, *Pyrgulopsis bruneauensis*[3]	Hydrobiidae	50	63	.38	.006	Surrogate	Hydrobiidae	Spire snail, *Amnicola sp.*
Desert valvata, *Valvata utahensis*[4,5]	Valvatidae	50	>.39	.38	<.97	Surrogate	Salmonidae	*Oncorhynchus mykiss*
Banbury Springs lanx, *Lanx* sp.[5]	Lymnaeidae	50	>.39	.38	<.97	Surrogate	Salmonidae	*Oncorhynchus mykiss*
Snake River physa, *Physa natricina*	Physidae	45	1.9	.35	.19	Surrogate	Physidae	*Aplexa hypnorum*

[1]Geometric mean of six hardness-adjusted NOEC values.

[2]In comparative testing, the lowest acute value of the three surrogate species tested to copper was 0.8 times the rainbow trout acute value to copper (Dwyer and others, 2005b). Despite their caution that complications with the sturgeon testing may have exaggerated their apparent sensitivity, they are used here in this comparison. Assuming acute-chronic responses are approximately proportional, the chronic rainbow trout NOEC was multiplied by 0.8 for the comparison here.

[3]Estimate made using interspecies correlation between fathead minnow (surrogate species) and Amnicola (predicted species) using U.S. Environmental Protection Agency's ICE model (Asfaw and others, 2004) with the fathead minnow mean chronic NOEC. Amnicola was 14 times more acutely resistant than fathead minnow to 5 chemicals. Fathead species mean NOEC = 4.5 µg/L at hardness 50. Assuming proportional acute and chronic responses, 4.5 µg/L times 14 = 95 µg/L.

[4]Also known as "Utah valvata."

[5]Rainbow trout used as a taxonomic-independent surrogate since in comparisons of 3 standard test species to 17 listed species for 5 chemicals representing a broad range of toxic modes of action, rainbow trout were more sensitive than listed species 81 percent of the tests. In tests with copper, a "representative" metal, rainbow trout were equal to or more sensitive than all the listed species tested (Dwyer and others, 2005b). Of the 19 percent of test results with listed species and rainbow trout in which the listed species were more sensitive, the geometric mean of the sensitivity factor between the rainbow trout and listed species was 0.63. Here, assuming that the rainbow trout NOECs and LC_{50s} are approximately proportional, the rainbow trout NOEC is multiplied by 0.63 to estimate listed snail NOECs.

Indirect Effects to Listed Salmonids Via Effects to Sensitive Prey Items

Indirect effects on listed species through food chain alterations are conceivable because by design, adverse effects could be allowed to as much as 5 percent of the taxa in an ecosystem. If a more sensitive taxa was disproportionately important in the food web of a listed species, indirect adverse effects could result to the listed species. The amphipod *Hyalella azteca* is the only taxa for which data were available to evaluate this potential scenario directly. *Hyalella azteca's* species mean chronic value was slightly lower than the calculated chronic criterion value (0.33 versus 0.41 µg/L cadmium), making it the most sensitive species tested to date with chronic cadmium exposures as well as a species that by design is not necessarily protected by the chronic cadmium criteria.

Field data and population modeling suggested that *Hyalella azteca* populations could be limited by cadmium at the chronic criterion concentration. These population limitations are unlikely to be of a magnitude that would result in local extinctions. In montane stream food webs, *Hyalella* probably are not dominant enough for invertivores such as salmonids, sculpin, dace, or salamanders to depend on them as a food source. However, in springbrooks and presumably other streams that have stable, low velocity flows, *Hyalella* can dominate benthic macroinvertebrate communities. A crash of a prey species in such a system could have large effects on predatory fish populations. *Hyalella* and other amphipods could be important components of lake food chains, and a contaminant-caused population crash could have important adverse ecological effects (see section "Risks of the 5th Percentile Species-Sensivity Distribution (SSD)-Based Chronic Criterion to a More Sensitive Species: population-level effects to the amphipod *Hyalella azteca*").

These scenarios suggest that if cadmium concentrations in receiving waters approach chronic criterion values, biomonitoring could be prudent to evaluate if the diversity and abundance of potential salmonid prey items are similar to reference or baseline conditions. For example, a salmonid prey metric was defined for stream macroinvertebrates that could be useful for interpreting general biomonitoring when risks of indirect food chain effects to salmonids is a concern (Suttle and others, 2004).

Potential Site-Specific Adjustments to Cadmium Criteria Values

The information presented to this point suggested several scenarios where the criteria values derived could be under- or overprotective for a specific water body. Potential scenarios where the criteria values calculated in this report might be under- or overprotective, and possible steps to reduce the uncertainties in the degree of protectiveness include:

Possibly underprotective—In behavioral studies following extended exposures of salmonids in the genus *Salvelinus* to sub-criterion cadmium concentrations, reductions in the ability of salmonids to capture prey, shifts in prey utilization, and reductions in condition factors have been observed (Kislalioglu and others, 1996; Scherer and others, 1997; Riddell and others, 2005b). These results suggest that if typical ambient cadmium concentrations were only slightly less than the chronic criterion concentrations, similar types of effects could occur in field conditions. However, reduced size or condition factors in *Salvelinus* were not obvious in two field situations with fish populations having long-term exposures to proportionally higher cadmium concentrations (Malley, 1996; Maret and MacCoy, 2002). These apparent differences between laboratory and field responses could be real or they could simply reflect the difficulty in detecting biological changes in field studies that are small in relation to natural variability. If biomonitoring of fish communities in waters that receive cadmium discharges show shifts in prey utilization or decreases in body condition that are congruent with those observed by Riddell and others (2005b) in their experimental streams; it follows that the chronic cadmium criterion should be adjusted downward. The magnitude of such a potential adjustment is unclear, because Riddell and others (2005b) did not observe a threshold of effects.

Possibly underprotective—If a critical species at a site were sensitive to cadmium at lower than criterion concentrations, then a lower criterion it might be needed. For example, if cadmium were to be discharged to a waterbody in which the amphipod *Hyalella azteca* was an important component of fish diets, and in which *Hyalella azteca* dominated the invertebrate assemblage such that alternative prey items were limited, then under these circumstances *Hyalella azteca* might be considered a critical species that warrants a downward adjustment to the chronic cadmium criterion. For example, a chronic criterion could be calculated based upon the species mean chronic value for *Hyalella azteca*, rather than the 5th percentile of the species-sensitivity distribution, as was done in this report.

Possibly overprotective—Chemical characteristics of site water could make cadmium less toxic than in the typical test waters for the datasets that were used to derive criteria values in this reports. Most information reviewed indicated that cadmium toxicity is mitigated by increasing calcium concentrations. Since calcium concentrations are often correlated with other factors that may influence toxicity such as hardness, pH and alkalinity, to some extent these factors are often accounted for in the hardness-adjusted criteria values. However, these relations may not hold at all sites, and other factors that may influence toxicity such as high calcium to magnesium ratios or dissolved organic carbon concentrations may be important at some locations. Several tests have shown that cadmium is less toxic in effluent influenced stream waters than it is in reconstituted test waters (Diamond and others, 1997; Brooks and others, 2004; Stanley and others, 2005).

However this may not be a good comparison for site waters, because standard recipes for reconstituted water may produce poor surrogates for natural waters (Welsh and others, 2000; 2001), and most of the data compiled in tables 15 and 16 of this report used waters originating from natural sources rather than reconstituted test waters.

Possibly overprotective—If the species occurring in a particular waterbody are likely less sensitive than those species that influenced the criteria derivation (that is, the four most sensitive genera), then the criteria values calculated here could be more stringent than needed to protect aquatic life. For example, the acute criterion value was lowered to protect critical species. If none of those species occur at a site that lowering may not be warranted. Similarly, two of the four most chronically sensitive genera were amphipods; if amphipods are unlikely to occur in a particular waterbody then the criteria might be more stringent than needed.

U.S. Environmental Protection Agency (1994) suggests that for waters where sensitive species used in the general criterion calculation and comparably sensitive species do not occur, or a narrower mix of species than that found in the general criterion data occurs at a site, then it may be appropriate to recalculate the criterion after excluding nonresident species from the dataset. In Idaho and many other jurisdictions, protected aquatic life uses are often classified according to perceived thermal regimes, such as coldwater or warmwater fisheries. Accordingly, Chadwick Ecological Consultants, Inc. (2004b) classified organisms as “coldwater” or “warmwater” and recalculated cadmium criteria for warmwater uses after excluding coldwater taxa, and vice versa. In a related approach, in lieu of a thermal classification, Dillon and Mebane (2002) used a lotic/lentic classification of organisms for inclusion in site-specific criteria datasets for coldwater streams in northern Idaho. Because some taxa such as amphipods and cladocerans were eurythermal, but were only found in lentic (slow moving) waters, they were excluded from consideration in criteria datasets (Dillon and Mebane, 2002). Both approaches have some risk that by further limiting the diversity of the criteria dataset, sensitive but untested taxa might not be fully protected. Dwyer and others (2005b) compared the relative sensitivities of commonly tested fish species to 17 endangered and threatened species and concluded that rainbow trout data likely represent the response of sensitive warmwater species and not merely responses of coldwater species. Procedures that exclude species because of temperature would likely not be protective of sensitive warmwater species (Dwyer and others, 2005b). In a similar analysis, the standard effluent toxicity test species *Ceriodaphnia dubia* (a lentic species), was almost always more sensitive than either the endangered species or the fathead minnow in comparative testing with seven endangered fish species with diverse effluents. In contrast, the fathead minnow, a species that occurs in both lentic and lotic waters, was less sensitive than the endangered species in about 80 percent of the comparisons (Dwyer and others, 2005a).

Summary

1. Cadmium is rare in aquatic environments, but it may be the most acutely toxic of all the atomically stable metals in the periodic table.

2. A large body of data on the effects of cadmium to aquatic animals has been reported. About 278, 93, and 102 acute, chronic, and other, respectively, test values were summarized in this report. Because previous reviews indicated that aquatic plants tended to be less sensitive to cadmium than sensitive aquatic animals, data searches were focused on studies with aquatic animals.

3. Most of data on the effects of cadmium to aquatic species were generated from tests where cadmium was tested as the single contaminant of concern. However, in ambient waters, cadmium probably seldom occurs as a single contaminant of concern and commonly occurs with zinc and copper. Data on the toxicity of cadmium in chemical mixtures or interactive effects were equivocal. In tests with binary cadmium and zinc mixtures, most data indicate cadmium is no more toxic than it is when tested singly. Although toxicity data with cadmium and copper mixtures were contradictory, some data indicate that adding copper may result in greater toxicity than cadmium alone or than cadmium and zinc mixtures.

4. Species mean acute values (SMAVs) and species mean chronic values (SMCVs) were determined for 69 and 28 aquatic species respectively. These values were assumed to represent a subset or surrogates for the species-sensitivity distribution (SSD) of the approximately 1,200 aquatic animal species in Idaho.

5. Data on the effects of cadmium through bioaccumulation into tissues residues were ambiguous. Tissue residue concentrations that co-occurred with the presence of observed adverse effects- and those that corresponded with the absence of observed adverse effects overlapped substantially. The data were insufficient to define maximum acceptable tissue concentrations and bioconcentration or bioaccumulation factors that could be used to back calculate water column concentrations that would reliably predict bioaccumulation risk.

6. Hardness-toxicity relations were determined that explained between 30–99 percent of the variability in toxicity for different species and data sets.

7. When normalized to a hardness of 50 mg/L as calcium carbonate, a Final Acute Value (FAV) and Final Chronic Value (FCV) were calculated from the 5^{th} percentile of SSD models, based on genus mean values. The FAV is divided by 2.0 to extrapolate from a concentration that caused unacceptable adverse effects to acutely sensitive taxa (killed 50 percent) to a criterion maximum

concentration expected to kill few if any organisms following acute exposures. In contrast, the FCV was calculated from chronic test values which reflected less severe effects than deaths of 50 percent of the test organisms. For fish, the chronic values corresponded with deaths or adverse effect to around 10–15 percent of the individuals, and for invertebrates around 20–30 percent. No further extrapolation of the FCV from low-effect to "no-effect" values was made.

8. Three species in the acute dataset had species mean acute values lower than the 5th percentile of the SSD (cutthroat trout, rainbow trout, and bull trout). The three species are the only species of trout native to Idaho and as the top carnivores in many coldwater ecosystems in Idaho, the three species were considered "critical" species. The species are also highly valued socially. Accordingly, the 5th percentile of SSD-based FAV was lowered to protect these sensitive species.

9. One species in the smaller chronic dataset had a SMCV lower than the 5th percentile of the SSD (the amphipod *Hyalella azteca*). Unlike the case with native trout in the acute dataset, *Hyalella azteca* was not considered a priori to be a critical species. Instead, the potential population-level and food chain consequences of exposure to a chronic criterion that was not designed to protect *Hyalella azteca* were evaluated. A population model was constructed that simulated *Hyalella azteca*'s seasonal life cycle in temperate, coldwater ecosystems. A model of cadmium toxicity to *Hyalella azteca* was nested with the baseline population model to forecast the effects of continuous or episodic cadmium exposures. The forecasts suggested that extinctions of *Hyalella azteca* populations were unlikely, although population declines were expected.

10. The role of *Hyalella azteca* in aquatic food chains is probably proportional to its relative abundance in benthic communities. In a food chain study in an impounded section of the Snake River in Washington and Idaho, *Hyalella azteca* contributed about 3–8 percent of the diets of juvenile Chinook salmon and steelhead. If benthic communities are diverse and alternate prey species are abundant, declines in *Hyalella azteca* populations seem unlikely to have a disproportionate effect throughout aquatic food webs.

11. The chronic criterion concentration for cadmium that was calculated using the 5th percentile of a SSD based on single-species laboratory experiments was compared with the presence or absence of apparent effects due to cadmium from ecosystem experiments or field surveys. In general, there was reasonable agreement between the laboratory-based predictions and effects observed in the field surveys or ecosystem experiments. Adverse effects at concentrations lower than the calculated chronic criterion were reported from small scale, simplified model ecosystem experiments. In these experiments, adverse behavioral effects were observed with two species in the genus *Salvelinus,* brook trout and lake trout. In a large-scale ecosystem study and field surveys, no obvious adverse effects to lake trout, brook trout, or other fishes were observed at cadmium chronic criterion concentrations. The disparity between effects observed in small-scale or large-scale ecosystem experiments or surveys could be real or artifacts of differing study designs and data types.

12. In streams in and around Idaho, maximum weekly or maximum monthly cadmium concentrations tended to be about 2 times higher than long-term average concentrations, suggesting that if streams met the chronic criterion, long-term average cadmium concentrations would be around 50 percent of criteria concentrations at many sites. This suggests that in practice, the chronic criterion might be more protective than would appear solely based on species-sensitivity distributions, and assuming indefinite exposures.

13. If ambient cadmium concentrations approached the SSD-based cadmium chronic criterion and if species that might not be fully protected are present, uncertainties in the criteria extrapolation and protectiveness could be reduced through monitoring of aquatic communities at exposed and reference sites. Field monitoring endpoints that correspond to experimental effects include the diversity and abundance of macroinvertebrate taxa, relative importance of prey items in the stomach contents of fish, condition factors of fish, fish species composition, age class distribution and relative abundances of fish species.

References Cited

Aldenberg, T., and Jaworska, J.S., 2000, Uncertainty of the hazardous concentration and fraction affected for normal species sensitivity distributions: Ecotoxicology and Environmental Safety, v. 46, no. 1, p. 1–18.

Ankley, G.T., Benoit, D.A., Hoke, R.A., Leonard, E.N., West, C.W., Phipps, G.L., Mattson, V.R., and Anderson, L.A., 1993, Development and evaluation of test methods for benthic invertebrates and sediments: effects of flow rate and feeding on water quality and exposure conditions: Archives of Environmental Contamination and Toxicology, v. 25, no. 1, p. 12–19.

Ankley, G.T., Phipps, G.L., Leonard, E.N., Benoit, D.A., Mattson, V.R., Kosian, P.A., Cotter, A.M., Dierkes, J.R., Hansen, D.J., and Mahony, J.D., 1991, Acid volatile sulfide as a factor mediating cadmium and nickel bioavailability in contaminated sediments: Environmental Toxicology and Chemistry, v. 10, p. 1299–1307.

Arillo, A., Calamari, D., Margiocco, C., Melodia, F., and Mensi, P., 1984, Biochemical effects of long-term exposure to cadmium and copper on rainbow trout (*Salmo gairdneri*): validation of water quality criteria: Ecotoxicology and Environmental Safety, v. 8, p. 106–117.

Asfaw, A., Ellersieck, M.R., and Mayer, F.L., 2004, Interspecies correlation estimations (ICE) for acute toxicity to aquatic organisms and wildlife, II. User manual and software: U.S. Environmental Protection Agency, EPA/600/R-03/105, 20 p.

American Society for Testing and Materials, 1997, Standard guide for conducting acute toxicity tests on test materials with fishes, macroinvertebrates, and amphibians. Method E729-96, Annual Book of ASTM Standards: West Conshohocken, PA, American Society for Testing and Materials, 22 p.

American Society for Testing and Materials, 1998, Standard guide for conducting early life-stage toxicity tests with fishes. Method E1241-98, Annual Book of ASTM Standards: West Conshohocken, Pa., American Society for Testing and Materials, 29 p.

Attar, E.N., and Maly, E.J., 1982, Acute toxicity of cadmium, zinc, and cadmium-zinc mixtures to *Daphnia magna*: Archives of Environmental Contamination and Toxicology, v. 11, no. 3, p. 291–296.

Baird, D.J., 1992, Predicting population response to pollutants: in praise of clones. A comment on Forbes and Depledge: Functional Ecology, v. 6, no. 5, p. 616–619.

Baird, D.J., Barber, I., Bradley, M., Soares, A.M.V.M., and Calow, P., 1991, A comparative study of genotype sensitivity to acute toxic stress using clones of *Daphnia magna* Straus: Ecotoxicology and Environmental Safety, v. 21, p. 257–265.

Baird, D.J., Barber, I., and Calow, P., 1990, Clonal variation in general responses of *Daphnia magna* Straus to toxic stress. I. Chronic life-history effects: Functional Ecology, v. 4, no. 4, p. 399–407.

Baker, C.F., and Montgomery, J.C., 2001, Sensory deficits induced by cadmium in banded kokopu, *Galaxias fasciatus*, juveniles: Environmental Biology of Fishes, v. 62, no. 4, p. 455–464

Baldisserotto, B., Chowdhury, M.J., and Wood, C.M., 2005, Effects of dietary calcium and cadmium on cadmium accumulation, calcium and cadmium uptake from the water, and their interactions in juvenile rainbow trout: Aquatic Toxicology, v. 72, no. 1-2, p. 99–117.

Ball, A.L., Borgmann, U., and Dixon, D.G., 2006, Toxicity of a cadmium-contaminated diet to *Hyalella azteca*: Environmental Toxicology and Chemistry, v. 25, no. 9, p. 2526–2532.

Barata, C., and Baird, D.J., 2000, Determining the ecotoxicological mode of action of chemicals from measurements made on individuals: results from instar-based tests with *Daphnia magna* Straus: Aquatic Toxicology, v. 48, no. 2/3, p. 195–209.

Barata, C., Baird, D.J., and Markich, S.J., 1998, Influence of genetic and environmental factors on the tolerance of *Daphnia magna* Straus to essential and non-essential metals.: Aquatic Toxicology, v. 42, no. 2, p. 115–137.

Barata, C., Baird, D.J., Miñarro, A., and Soares, A.M.V.M., 2000, Do genotype responses always converge from lethal to nonlethal toxicant exposure levels? Hypothesis tested using clones of *Daphnia magna* Straus: Environmental Toxicology and Chemistry, v. 19, no. 9, p. 2314–2322.

Barata, C., Baird, D.J., Mitchell, S.E., and Soares, A.M.V.M., 2002a, Among- and within-population variability in tolerance to cadmium stress in natural populations of *Daphnia magna* –implications for ecological risk assessment: Environmental Toxicology and Chemistry, v. 21, no. 5, p. 1058–1064.

Barata, C., Markich, S.J., Baird, D.J., and Soares, A.M.V.M., 2002b, The relative importance of water and food as cadmium sources to *Daphnia magna* Straus: Aquatic Toxicology, v. 61, no. 3-4, p. 143–154.

Barnthouse, L.W., 1993, Population-level effects, *in* Suter, G.W., II, ed., Ecological Risk Assessment: Chelsea, Michigan, Lewis Publishers, p. 247–274.

Barnthouse, L.W., 2004, Quantifying population recovery rates for ecological risk assessment: Environmental Toxicology and Chemistry, v. 23, no. 2, p. 500–508.

Barnthouse, L.W., Suter, G.W., II, and Rosen, A.E., 1990, Risks of toxic contaminants to exploited fish populations: influence of life history, data uncertainty and exploitation intensity: Environmental Toxicology and Chemistry, v. 9, p. 297–311.

Baxter, C.V., 2002, Fish movement and assemblage dynamics in a Pacific Northwest riverscape: Corvallis, Oregon State University, Ph.D. dissertation, 188 p.

Beauvais, S.L., Jones, S.B., Parris, J.T., Brewer, S.K., and Little, E.E., 2001, Cholinergic and behavioral neurotoxicity of carbaryl and cadmium to larval rainbow trout (*Oncorhynchus mykiss*): Ecotoxicology and Environmental Safety, v. 49, no. 1, p. 84–90.

Benoit, D.A., Leonard, E.N., Christensen, G.M., and Fiandt, J.T., 1976, Long-term effects of cadmium exposure on three generations of brook trout (*Salvelinus fontinalis*): Transactions of the American Fisheries Society, v. 108, no. 4, p. 550–560.

Besser, J.M., Brumbaugh, W.G., May, T.W., Church, S.E., and Kimball, B.A., 2001, Bioavailability of metals in stream food webs and hazards to brook trout (*Salvelinus fontinalis*) in the upper Animas River watershed, Colorado: Archives of Environmental Contamination and Toxicology, v. 40, no. 1, p. 49–59.

Besser, J.M., Mebane, C.A., Ivey, C.D., Kunz, J.L., Greer, E.I., May, T.W., and Ingersoll, C.G., 2006, Relative sensitivity of mottled sculpins (*Cottus bairdi*) and rainbow trout (*Onchorhynchus mykiss*) to toxicity of metals associated with mining activities: Columbia, Mo., U.S. Geological Survey, Columbia Environmental Research Laboratory, Administrative Report to U.S. Environmental Protection Agency, Project No. DW14939902-01-0. 37 p.

Besser, J.M., Wang, N., Dwyer, F.J., Mayer, F.L., and Ingersoll, C.G., 2005, Assessing contaminant sensitivity of endangered and threatened fishes: 2. Chronic toxicity of copper and pentachlorophenol to two endangered species and two surrogate species: Archives of Environmental Contamination and Toxicology, v. 48, no. 2, p. 155–165.

Birge, W.J., Black, J.A., and Ramey, B.A., 1981, The reproductive toxicology of aquatic contaminants, *in* Saxena, J., and Fisher, F., eds., Hazard Assessment of Chemicals: Current Developments: New York, Academic Press, p. 59–115.

Birge, W.J., Price, D.J., Shaw, J.R., Spromberg, J.A., Wigginton, A.J., and Hogstrand, C., 2000, Metal body burden and biological sensors as ecological indicators: Environmental Toxicology and Chemistry, v. 19, no. 4, p. 1199–1212.

Bishop, W.E., and McIntosh, A.W., 1981, Acute lethality and effects of sublethal cadmium exposure on ventilation frequency and cough rate of bluegill (*Lepomis macrochirus*): Archives of Environmental Contamination and Toxicology, v. 10, no. 5, p. 519–530.

Boreman, J., 1997, Methods for comparing the impacts of pollution and fishing on fish populations: Transactions of the American Fisheries Society, v. 126, no. 2, p. 506–513.

Borgmann, U., Couillard, Y., Doyle, P., and Dixon, D.G., 2005, Toxicity of sixty-three metals and metalloids to *Hyalella azteca* at two levels of water hardness: Environmental Toxicology and Chemistry, v. 24, no. 3, p. 641–652.

Borgmann, U., Millard, E.S., and Charlton, C.C., 1989a, Effect of cadmium on a stable, large volume, laboratory ecosystem containing *Daphnia* and phytoplankton: Canadian Journal of Fisheries and Aquatic Sciences, v. 46, p. 399–405.

Borgmann, U., Norwood, W.P., and Babirad, I.M., 1991, Relationship between chronic toxicity and bioaccumulation of cadmium in *Hyalella azteca*: Canadian Journal of Fisheries and Aquatic Sciences, v. 48, no. 6, p. 1055–1060.

Borgmann, U., Norwood, W.P., and Dixon, D.G., 2004, Re-evaluation of metal bioaccumulation and chronic toxicity in *Hyalella azteca* using saturation curves and the biotic ligand model: Environmental Pollution, v. 131, no. 3, p. 469–484.

Borgmann, U., Ralph, K.M., and Norwood, W.P., 1989b, Toxicity test procedures for *Hyalella azteca*, and chronic toxicity of cadmium and pentachlorophenol to *H. azteca*, *Gammarus fasciatus*, and *Daphnia magna*: Archives of Environmental Contamination and Toxicology, v. 18, no. 5, p. 756–764.

Bradley, R.W., DuQuesnay, C., and Sprague, J.B., 1985, Acclimation of rainbow trout, *Salmo gairdneri* Richardson, to zinc – kinetics and mechanism of enhanced tolerance induction: Journal of Fish Biology, v. 27, p. 367–379.

Bridges, T.S., and Lutz, C.H., 2005, The Environmental Residue-Effects Database (ERED), accessed December 2005 at http://el.erdc.usace.army.mil/ered/.

Brinkman, S.F., and Hansen, D., 2004a, Effect of hardness on cadmium toxicity to brown trout (*Salmo trutta*) embryos, larvae, and fry, Water Pollution Studies, Federal Aid in Fish and Wildlife Restoration Project F-243-R11: Fort Collins, Colorado Division of Wildlife, p. 4–20.

Brinkman, S.F., and Hansen, D., 2004b, Effect of hardness on zinc toxicity to Colorado River cutthroat trout (*Oncorhynchus clarki pleuriticus*) and rainbow trout (*Oncorhynchus mykiss*) embryos and fry, Water Pollution Studies, Federal Aid in Fish and Wildlife Restoration Project F-243-R11: Fort Collins, Colorado Division of Wildlife, p. 22–35.

Brix, K.V., DeForest, D.K., Burger, M., and Adams, W.J., 2005, Assessing the relative sensitivity of aquatic organisms to divalent metals and their representation in toxicity datasets compared to natural aquatic communities Human and Ecological Risk Assessment, v. 11, no. 6, p. 1139–1156.

Brooks, B.W., Stanley, J.K., White, J.C., Turner, P.K., Wu, K.B., and La Point, T.W., 2004, Laboratory and field responses to cadmium: an experimental study in effluent dominated stream mesocosms: Environmental Toxicology and Chemistry, v. 23, no. 4, p. 1057–1064.

Brown, M.W., Thomas, D.G., Shurben, D., Solbé, J.F.d.L.G., Kay, J., and Cryer, A., 1986, A comparison of the differential accumulation of cadmium in the tissues of three species of freshwater fish, *Salmo gairdneri*, *Rutilus rutilus* and *Noemacheilus barbatulus*: Comparative Biochemistry and Physiology Part C, Comparative Pharmacology, v. 84C, no. 2, p. 213–217.

Brown, V., Shurben, D., Miller, W., and Crane, M., 1994, Cadmium toxicity to rainbow trout *Oncorhynchus mykiss* Walbaum and brown trout *Salmo trutta L.* over extended exposure periods: Ecotoxicology and Environmental Safety, v. 29, no. 1, p. 38–46.

Bruce, R.D., and Versteeg, D.J., 1992, A statistical procedure for modeling continuous toxicity data: Environmental Toxicology and Chemistry, v. 11, no. 10, p. 1485–1494.

Buhl, K.J., 1997, Relative sensitivity of three endangered fishes, Colorado squawfish, bonytail, and razorback sucker, to selected metal pollutants: Ecotoxicology and Environmental Safety, v. 37, no. 2, p. 186–192.

Buhl, K.J., and Hamilton, S.J., 1991, Relative sensitivity of early life stages of Arctic grayling, coho salmon, and rainbow trout to nine inorganics: Ecotoxicology and Environmental Safety, v. 22, no. 2, p. 184–197.

Cain, D.J., Luoma, S.N., and Wallace, W.G., 2004, Linking metal bioaccumulation of aquatic insects to their distribution patterns in a mining-impacted river: Environmental Toxicology and Chemistry, v. 23, no. 6, p. 1463–1473.

Cairns, J., Jr., Niederlehner, B.R., and Smith, E.P., 1995, Ecosystem effects: functional endpoints, *in* Rand, G.M., ed., Fundamentals of aquatic toxicology: effects, environmental fate, and risk assessment, (2nd ed.): Washington, D.C., Taylor and Francis, p. 589–608.

Calow, P., Sibly, R.M., and Forbes, V.E., 1997, Risk assessment on the basis of simplified life-history scenarios: Environmental Toxicology and Chemistry, v. 16, no. 9, p. 1983–1989.

Cambridge Scientific Abstracts, 2005a, ASFA: Aquatic Sciences and Fisheries Abstracts [database], accessed January–May 2005, at http://www.csa.com/e_products/databases-collections.php.

Cambridge Scientific Abstracts, 2005b, Water Resources Abstracts [database], accessed January–May 2005, at http://www.csa.com/e_products/databases-collections.php.

Campbell, P.G.C., and Stokes, P.M., 1985, Acidification and toxicity of metals to aquatic biota: Canadian Journal of Fisheries and Aquatic Sciences, v. 42, p. 2034–2049.

Canfield, T.J., Kemble, N.E., Brumbaugh, W.G., Dwyer, F.J., Ingersoll, C.G., and Fairchild, J.F., 1994, Use of benthic invertebrate community structure and the sediment quality triad to evaluate the metals-contaminated sediment in the upper Clark Fork River, Montana: Environmental Toxicology and Chemistry, v. 13, no. 12, p. 1999–2012.

Canton, J.H., and Sloof, W., 1982, Toxicity and accumulation studies of cadmium (Cd^{2+}) with freshwater organisms of different trophic levels: Ecotoxicology and Environmental Safety, v. 6, p. 113–128.

Carlisle, D.M., and Clements, W.H., 1999, Sensitivity and variability of metrics used in biological assessments of running waters: Environmental Toxicology and Chemistry, v. 18, no. 2, p. 285–291.

Carroll, J.J., Ellis, S.J., and Oliver, W.S., 1979, Influences of total hardness constituents on the acute toxicity of cadmium to brook trout (*Salvelinus fontinalis*): Bulletin of Environmental Contamination and Toxicology, v. 22, p. 575–581.

Castillo, V., III, and Longley, G., 2001, Comparison of EPA target toxicity aquatic test organisms to the fountain darter: 7 day chronic toxicity test using cadmium chloride, performed 11/12/99–3/6/00 (5 parts): San Marcos, Tex., Edwards Aquifer Research and Data Center (EARDC), Southwest Texas State University, Fed. Assist. Agree. No. X-986345-01 (5 parts), 179 p.

Caswell, H., 2001, Matrix Population Models: Construction, analysis, and interpretation: Sunderland, Mass., Sinauer Associates, Inc., 722 p.

Chadwick Ecological Consultants, Inc., 2000, Aquatic biological monitoring of Thompson Creek and Squaw Creek, Custer County, Idaho, Prepared for Thompson Creek Mining, Challis, Idaho by Chadwick Ecological Consultants, Littleton, Colo., 32 p.

Chadwick Ecological Consultants, Inc., 2004a, Chronic toxicity of cadmium to freshwater crustaceans at different water hardness concentrations, Prepared for Thompson Creek Mining, Challis, Idaho by Chadwick Ecological Consultants, Littleton, Colo., 62 p.

Chadwick Ecological Consultants, Inc., 2004b, U.S. EPA cadmium water quality criteria document—technical review and criteria update, Report to the Association of Metropolitan Sewerage Agencies (AMSA) by Chadwick Ecological Consultants, Inc., Littleton, Colo., 45 p, Chapman, G.A., 1975, Toxicity of copper, cadmium, and zinc to Pacific Northwest salmonids: Corvallis, Oreg., U.S. Environmental Protection Agency, Western Fish Toxicology Station, National Water Quality Laboratory, Interim Report Task 002 ROAP 10CAR., 27 p.

Chapman, G.A., 1978a, Effects of continuous zinc exposure on sockeye salmon during adult-to-smolt freshwater residency: Transactions of the American Fisheries Society, v. 107, no. 6, p. 828–836.

Chapman, G.A., 1978b, Toxicities of cadmium, copper, and zinc to four juvenile stages of chinook salmon and steelhead: Transactions of the American Fisheries Society, v. 107, no. 6, p. 841–847.

Chapman, G.A., 1982, Chinook salmon early life stage tests with cadmium, copper, and zinc, Letter of December 6, 1982, to Charles Stephan, U.S. Environmental Protection Agency Environmental Research Laboratory, Duluth: Corvallis, Oreg., U.S. EPA Environmental Research Laboratory.

Chapman, G.A., 1994, [Acclimation, life stage differences in lethality and behavioral effects of chronic copper exposures with steelhead], Letter of July 5, 1994 to Chris Mebane, [NOAA liaison to] EPA Region 10, Seattle, Wash: Newport, Oreg., U.S. Environmental Protection Agency Coastal Ecosystems Team.

Chapman, G.A., Ota, S., and Recht, F., 1980, Effects of water hardness on the toxicity of metals to *Daphnia magna*: Corvallis, Oreg., U.S. Environmental Protection Agency, Office of Research and Development, 12 p.

Chapman, G.A., and Stevens, D.G., 1978, Acutely lethal levels of cadmium, copper, and zinc to adult male coho salmon and steelhead: Transactions of the American Fisheries Society, v. 107, no. 6, p. 837–840.

Chapman, P.M., 1995, Extrapolating laboratory toxicity test results to the field: Environmental Toxicology and Chemistry, v. 14, p. 927–930.

Chapman, P.M., Farrell, M.A., and Brinkhurst, R.O., 1982, Relative tolerances of selected aquatic oligochaetes to individual pollutants and environmental factors: Aquatic Toxicology, v. 2, no. 1, p. 47–67.

Clark, G.M., 2002, Occurrence and transport of cadmium, lead, and zinc in the Spokane River basin, Idaho and Washington, water years 1999-2001: U.S. Geological Survey Water-Resources Investigations Report 02-4183, 37 p., accessed October 2003 at http://id.water.usgs.gov/public/reports.html.

Clements, W.H., 2004, Small-scale experiments support causal relationships between metal contamination and macroinvertebrate community composition: Ecological Applications, v. 14, no. 3, p. 954–967.

Clements, W.H., Carlisle, D.M., Courtney, L.A., and Harrahy, E.A., 2002, Integrating observational and experimental approaches to demonstrate causation in stream biomonitoring studies: Environmental Toxicology and Chemistry, v. 21, no. 6, p. 1138–1146.

Clements, W.H., Carlisle, D.M., Lazorchak, J.M., and Johnson, P.C., 2000, Heavy metals structure benthic communities in Colorado mountain streams: Ecological Applications, v. 10, no. 2, p. 626–638.

Clements, W.H., and Kiffney, P.M., 1996, Validation of whole effluent toxicity tests: integrated studies using field assessments, microcosms, and mesocosms, *in* Grothe, D.L., Dickson, K.L., and Reed-Judkins, D.K., eds., Whole effluent toxicity testing: an evaluation of methods and prediction of receiving system impacts: Pensacola, Fla., Society of Environmental Toxicology and Chemistry (SETAC), p. 229–244.

Coeurdassier, M., De Vaufleury, A., and Badot, P.-M., 2003, Bioconcentration of cadmium and toxic effects on life-history traits of pond snails (*Lymnaea palustris* and *Lymnaea stagnalis*) in laboratory bioassays: Archives of Environmental Contamination and Toxicology, v. 45, no. 1, p. 102–109.

Coeurdassier, M., De Vaufleury, A., and Badot, P.-M., 2004, Effects of cadmium on the survival of three life-stages of the freshwater pulmonate *Lymnaea stagnalis* (Mollusca: Gastropoda): Bulletin of Environmental Contamination and Toxicology, v. 72, p. 1083–1090.

Collyard, S.A., Ankley, G.T., Hoke, R.A., and Goldenstein, T., 1994, Influence of age on the relative sensitivity of *Hyalella azteca* to diazinon, alkylphenol ethoxylates, copper, cadmium, and zinc: Archives of Environmental Contamination and Toxicology, v. 26, no. 1, p. 110–113.

Cooke, S.J., Bunt, C.M., Hamilton, S.J., Jennings, C.A., Pearson, M.P., Cooperman, M.S., and Markle, D.F., 2005, Threats, conservation strategies, and prognosis for suckers (Catostomidae) in North America: insights from regional case studies of a diverse family of non-game fishes: Biological Conservation, v. 121, no. 3, p. 317–478.

Cooper, W.E., 1965, Dynamics and production of a natural population of a fresh-water amphipod, *Hyalella azteca*: Ecological Monographs, v. 35, no. 4, p. 377–394.

Couture, P., and Kumar, P.R., 2003, Impairment of metabolic capacities in copper and cadmium contaminated wild yellow perch (*Perca flavescens*): Aquatic Toxicology, v. 64, p. 107–120.

Crane, M., and Newman, M.C., 2000, What level of effect is a no observed effect?: Environmental Toxicology and Chemistry, v. 19, no. 2, p. 516–519.

Cusimano, R.F., Brakke, D.F., and Chapman, G.A., 1986, Effects of pH on the toxicities of cadmium, copper, and zinc to steelhead trout (*Salmo gairdneri*): Canadian Journal of Fisheries and Aquatic Sciences, v. 43, no. 8, p. 1497–1503.

Davies, P.H., and Brinkman, S.F., 1994a, Appendix I: Effects of pre-exposure to sublethal waterborne cadmium on cadmium toxicity, metallothionein concentrations, and subcellular distribution of cadmium in the gill and kidney of brown trout, *in* Davies, P.H., ed., Water Pollution Studies, Federal Aid in Fish and Wildlife Restoration, Project #33: Fort Collins, Colo., Colorado Division of Wildlife, p. I-11–I-31.

Davies, P.H., and Brinkman, S.F., 1994b, Appendix II: Cadmium toxicity to rainbow trout: bioavailability and kinetics in waters of high and low complexing capacities, *in* Davies, P.H., ed., Water Pollution Studies, Federal Aid in Fish and Wildlife Restoration, Project #33: Fort Collins, Colo., Colorado Division of Wildlife, p. II-33–II-59.

Davies, P.H., and Brinkman, S.F., 1994c, Toxicology and chemical data on unregulated pollutants, Water Pollution Studies, Federal Aid in Fish and Wildlife Restoration, Project #33: Fort Collins, Colo., Colorado Division of Wildlife, p. 5–10.

Davies, P.H., Gorman, W.C., Carlson, C.A., and Brinkman, S.F., 1993, Effect of hardness on bioavailability and toxicity of cadmium to rainbow trout: Chemical Speciation and Bioavailability, v. 5, no. 2, p. 67–77.

de Bruin, J.H.M., and Hof, M., 1997, How to measure no effect. Part IV: How acceptable is the ECx from an environmental policy point of view?: Environmetrics, v. 8, no. 3, p. 263–267.

de March, B.G.E., 1977, The effects of photoperiod and temperature on the induction and termination of reproductive resting stage in the freshwater amphipod *Hyalella azteca* (Saussure): Canadian Journal of Zoology, v. 55, no. 10, p. 1595–1600.

de March, B.G.E., 1978, The effects of constant and variable temperatures on the size, growth, and reproduction of the freshwater amphipod *Hyalella azteca* (Saussure): Canadian Journal of Zoology, v. 56, no. 8, p. 1801–1806.

De Schamphelaere, K.A.C., and Janssen, C.R., 2004, Bioavailability and chronic toxicity of zinc to juvenile rainbow trout (*Oncorhynchus mykiss*): comparison with other fish species and development of a Biotic Ligand Model: Environmental Science and Technology, v. 38, no. 23, p. 6201–6209.

de Vlaming, V., and Norberg-King, T.J., 1999, A review of single species toxicity tests: Are the tests reliable predictors of aquatic ecosystem community response?: Duluth, Minn., U.S. Environmental Protection Agency, EPA 600/R/97/114.

Detenbeck, N.E., DeVore, P.W., Niemi, G.J., and Lima, A., 1992, Recovery of temperate-stream fish communities from disturbance - a review of case studies and synthesis of theory: Environmental Management, v. 16, no. 1, p. 33–53.

Di Toro, D.M., Allen, H.E., Bergman, H.L., Meyer, J.S., Paquin, P.R., and Santore, R.C., 2001, Biotic ligand model of the acute toxicity of metals. 1. Technical basis: Environmental Toxicology and Chemistry, v. 20, no. 10, p. 2383–2396.

Diamond, J.M., Bowersox, M., Latimer, H., Barbour, C., Bearr, J., and Butcher, J.B., 2005, Effects of pulsed contaminant exposures on early life stages of the fathead minnow: Archives of Environmental Contamination and Toxicology, v. 49, no. 4, p. 511–519.

Diamond, J.M., Koplish, D.E., McMahon, J., III, and Rost, R., 1997, Evaluation of the water-effect ratio procedure for metals in a riverine system: Environmental Toxicology and Chemistry, v. 16, no. 3, p. 509–520.

Dillon, F.S., and Mebane, C.A., 2002, Development of site-specific water quality criteria for the South Fork Coeur d'Alene River, Idaho: application of site-specific water quality criteria developed in headwater reaches to downstream waters: Seattle, Wash., Prepared for and in conjunction with the Idaho Department of Environmental Quality. Windward Environmental. 95 p, Accessed January 2005 at http://www.deq.state.id.us/water/data_reports/surface_water/monitoring/site_specific_criteria.cfm.

Drever, J.I., 1995, The geochemistry of natural waters (3rd ed.): Englewood Cliffs, N.J., Prentice Hall, 436 p.

Duan, Y., Guttman, S.I., and Oris, J.T., 1997, Genetic differentiation among laboratory populations of *Hyalella azteca*: implications for toxicology: Environmental Toxicology and Chemistry, v. 16, no. 3, p. 691–695.

Dwyer, F.J., Hardesty, D.K., Henke, C.E., Ingersoll, C.G., Whites, D.W., Augspurger, T., Canfield, T.J., Mount, D.R., and Mayer, F.L., 2005a, Assessing contaminant sensitivity of endangered and threatened fishes: III. Effluent toxicity tests: Archives of Environmental Contamination and Toxicology, v. 48, no. 2, p. 174–183.

Dwyer, F.J., Mayer, F.L., Sappington, L.C., Buckler, D.R., Bridges, C.M., Greer, I.E., Hardesty, D.K., Henke, C.E., Ingersoll, C.G., Kunz, J.L., Whites, D.W., Mount, D.R., Hattala, K., and Neuderfer, G.N., 2005b, Assessing contaminant sensitivity of endangered and threatened fishes: I. Acute toxicity of five chemicals: Archives of Environmental Contamination and Toxicology, v. 48, no. 2, p. 143–154.

Dyer, S.D., White-Hull, C.E., and Shepard, B.K., 2000, Assessments of chemical mixtures via toxicity reference values overpredict hazard to Ohio fish communities: Environmental Science and Technology, v. 34, p. 2518–2524.

Eaton, J.G., 1974, Chronic cadmium toxicity to the bluegill (*Lepomis macrochirus* Rafinesque): Transactions of the American Fisheries Society, v. 103, no. 4, p. 729–735.

Eaton, J.G., 1978, Metal toxicity to embryos and larvae of seven freshwater fish species-I. cadmium: Bulletin of Environmental Contamination and Toxicology, v. 19, p. 95–103.

Eisler, R., 1985, Cadmium hazards to fish, wildlife, and invertebrates: a synoptic review: Laurel, Md., U.S. Fish and Wildlife Service, Patuxent Research Station, Contaminant Hazard Reviews, Report No. 2. Biological Report 85(1.2). 46 p.

Erickson, R.J., 2002, Toxicity Response Analysis Program, version 1.0: Duluth, MN U.S. Environmental Protection Agency, National Health and Environmental Research Laboratory, Mid-Continent Ecological Division.

Erickson, R.J., Highland, T.L., Hockett, J.R., Leonard, E.N., Mattson, V.R., and Mount, D.R., 2003, Effects of dietary copper, zinc, lead, cadmium, and arsenic on growth and survival of juvenile fish using live food organisms, *in* Society of Environmental Toxicology and Chemistry 24th Annual Meeting Abstracts, November 9-13, 2003 Austin, Texas, Society of Environmental Toxicology and Chemistry, Pensacola, Fla.

Erickson, R.J., and Stephan, C.E., 1988, Calculation of the final acute value for water quality criteria for aquatic organisms: Duluth, Minn., Environmental Research Laboratory-Duluth, EPA/600/3-88/018. 62 p.

EVS Environment Consultants, 1996, Technical memorandum—Results of range-finding tests: Seattle, Wash., Prepared for the Idaho Division of Environmental Quality, EVS Environment Consultants, [42 p].

Farag, A.M., Boese, C.J., Woodward, D.F., and Bergman, H.L., 1994, Physiological changes and tissue metal accumulations of rainbow trout exposed to foodborne and waterborne metals: Environmental Toxicology and Chemistry, v. 13, no. 12, p. 2021–2030.

Farag, A.M., Skaar, D., Nimick, D.A., MacConnell, E., and Hogstrand, C., 2003, Characterizing aquatic health using salmonid mortality, physiology, and biomass estimates in streams with elevated concentrations of arsenic, cadmium, copper, lead, and zinc in the Boulder River watershed, Montana: Transactions of the American Fisheries Society, v. 132, no. 3, p. 450–467.

Farag, A.M., Woodward, D.F., Brumbaugh, W.G., Goldstein, J.N., McConnell, E., Hogstrand, C., and Barrows, F.T., 1999, Dietary effects of metals-contaminated invertebrates from the Coeur d'Alene River, Idaho on cutthroat trout: Transactions of the American Fisheries Society, v. 129, p. 578–592.

Farag, A.M., Woodward, D.F., Goldstein, J.N., Brumbaugh, W.G., and Meyer, J.S., 1998, Concentrations of metals associated with mining waste in sediments, biofilm, benthic macroinvertebrates, and fish from the Coeur d'Alene River Basin, Idaho: Archives of Environmental Contamination and Toxicology, v. 34, no. 2, p. 119–127.

Fausch, K.D., Hawkes, C.L., and Parsons, M.G., 1988, Models that predict standing crop of stream fish from habitat variables: 1950-85: Portland, Pacific Northwest Research Station, U.S. Forest Service, General Technical Report PNW-GTR-213, 52 p.

Fisher, D.J., and Burton, D.T., 2003, Comparison of two U.S. Environmental Protection Agency species sensitivity distribution methods for calculating ecological risk criteria: Human and Ecological Risk Assessment, v. 9, no. 3, p. 675–690.

Finlayson, B.J., and Verrue, K.M., 1982, Toxicities of copper, zinc, and cadmium mixtures to juvenile Chinook salmon: Transactions of the American Fisheries Society, v. 111, p. 645–650.

Forbes, T.L., and Forbes, V.E., 1993a, A critique of the use of distribution-based extrapolation models in ecotoxicology: Functional Ecology, v. 7, no. 3, p. 249–254.

Forbes, V.E., and Calow, P., 1999, Is the per capita rate of increase a good measure of population-level effects in ecotoxicology?: Environmental Toxicology and Chemistry, v. 18, no. 7, p. 1544–1556.

Forbes, V.E., and Calow, P., 2002, Species sensitivity distributions revisited: a critical appraisal: Human and Ecological Risk Assessment, v. 8, no. 3, p. 473-492.

Forbes, V.E., and Depledge, M.H., 1992, Predicting population response to pollutants: the significance of sex: Functional Ecology, v. 6, no. 4, p. 376–381.

Forbes, V.E., and Depledge, M.H., 1993, Testing vs. research in ecotoxicology: a response to Baird and Calow: Functional Ecology, v. 7, no. 5, p. 509–512.

Forbes, V.E., and Forbes, T.L., 1993b, Ecotoxicology and the power of clones: Functional Ecology, v. 7, no. 5, p. 509–512.

Fore, L.S., 2002, Biological assessment of mining disturbance of stream invertebrates in mineralized areas of Colorado, *in* Simon, T.P., and Stewart, P.M., eds., Biological Response Signatures: Multimetric index patterns for assessment of freshwater aquatic assemblages: Boca Raton, Fla, CRC Press, p. 341–364.

Franklin, N.M., Glover, C.N., Nicol, J.A., and Wood, C.M., 2005, Calcium/cadmium interactions at uptake surfaces in rainbow trout: waterborne versus dietary routes of exposure: Environmental Toxicology and Chemistry, v. 24, no. 11, p. 2954–2964.

Frey, D.G., 1977, Biological integrity of water-an historical approach, *in* The integrity of water, Washington, D.C., U.S. Environmental Protection Agency, p. 127–140.

García, G.G., Nandini, S., and Sarma, S.S.S., 2004, Effect of cadmium on the population dynamics of *Moina macrocopa* and *Macrothrix triserialis* (Cladocera): Bulletin of Environmental Contamination and Toxicology, v. 72, p. 717–724.

Gibbons, W.N., and Mackie, G.L., 1991, The relationship between environmental variables and demographic patterns of *Hyalella azteca* (Crustacea: Amphipoda): Journal of the North American Benthological Society, v. 10, no. 4, p. 444–454.

Gibbons, W.N., Munkittrick, K.R., and Taylor, W.D., 1998, Monitoring aquatic environments receiving industrial effluent using small fish species 1: response of spoonhead sculpin (*Cottus ricei*) downstream of a bleached-kraft pulp mill: Environmental Toxicology and Chemistry, v. 17, no. 11, p. 2227–2237.

Ginzburg, L.R., Ferson, S., and Akçakaya, H.R., 1990, Reconstructibility of density dependence and the conservative assessment of extinction risks: Conservation Biology, v. 4, no. 1, p. 63–70.

Gomot, A., 1998, Toxic effects of cadmium on reproduction, development, and hatching in the freshwater snail *Lymnaea stagnalis* for water quality monitoring: Ecotoxicology and Environmental Safety, v. 41, p. 288–297.

Gossiaux, D.C., Landrum, P.F., and Tsymbal, V.N., 1992, Response of the amphipod *Diporeia* spp. to various stressors: cadmium, salinity, and temperature: Journal of Great Lakes Research, v. 18, no. 3, p. 364–371.

Goulden, C.E., 1999, A critique of effluent bioassays, *in* Schulze, P.C., ed., Measures of environmental performance and ecosystem condition: Washington, D.C., National Academy Press, p. 191–198.

Grafe, C.S., Mebane, C.A., McIntyre, M.J., Essig, D.A., Brandt, D.H., and Mosier, D.T., 2002, The Idaho Department of Environmental Quality Water Body Assessment Guidance, Second Edition: Boise, Idaho, Idaho Department of Environmental Quality, 230 p., accessed May 2005 at http://www.deq.state.id.us/water/data_reports/surface_water/monitoring/overview.cfm#wbag.

Grosell, M., and Brix, K.V., 2004, Why are freshwater pulmonate snails so sensitive to chronic metal exposure?: SETAC Globe, v. 5, no. 3, p. 43–45.

Grosell, M., Gerdes, R.M., and Brix, K.V., 2006, Chronic toxicity of lead to three freshwater invertebrates - *Brachionus calyciflorus, Chironomus tentans,* and *Lymnaea stagnalis*: Environmental Toxicology and Chemistry, v. 25, no. 1, p. 97–104.

Gust, K.A., 2006, Joint toxicity of cadmium and phenanthrene in the freshwater amphipod *Hyalella azteca*: Archives of Environmental Contamination and Toxicology, v. 50, no. 1, p. 7–13.

Hansen, J.A., Lipton, J., and Welsh, P.G., 1999, Sensitivity of bull trout *(Salvelinus confluentus)* to cadmium and zinc in water characteristic of the Coeur d'Alene River Basin: Boulder, Colo., Report to U.S. Environmental Protection Agency Region 10. Stratus Consulting, variously paged.

Hansen, J.A., Lipton, J., Welsh, P.G., Cacela, D., and MacConnell, B., 2004, Reduced growth of rainbow trout (*Oncorhynchus mykiss*) fed a live invertebrate diet pre-exposed to metal-contaminated sediments: Environmental Toxicology and Chemistry, v. 23, no. 8, p. 1902–1911.

Hansen, J.A., Welsh, P.G., Lipton, J., Cacela, D., and Dailey, A.D., 2002a, Relative sensitivity of bull trout *(Salvelinus confluentus)* and rainbow trout *(Oncorhynchus mykiss)* to acute exposures of cadmium and zinc: Environmental Toxicology and Chemistry, v. 21, no. 1, p. 67–75.

Hansen, J.A., Welsh, P.G., Lipton, J., and Suedkamp, M.J., 2002b, The effects of long-term cadmium exposure on the growth and survival of bull trout (*Salvelinus confluentus*): Aquatic Toxicology, v. 58, no. 3-4, p. 165–174.

Hatakeyama, S., and Yasuno, M., 1981, Effects of cadmium on the periodicity of parturation and brood size of *Moina macrocopa* (Cladocera): Environmental Pollution (Series A), v. 26, p. 111–120.

Heijerick, D.G., De Schamphelaere, K.A.C., and Janssen, C.R., 2002, Predicting acute zinc toxicity for *Daphnia magna* as a function of key water chemistry characteristics: development and validation of a biotic ligand model: Environmental Toxicology and Chemistry, v. 21, no. 6, p. 1309–1315.

Hem, J.D., 1992, Study and interpretation of the chemical characteristics of natural water: U.S. Geological Survey Water-Supply Paper 2254, 263 p.

Hertzberg, R.C., and MacDonell, M.M., 2002, Synergy and other ineffective mixture risk definitions: Science of the Total Environment, v. 288, no. 1, p. 31–42.

Hickey, C.W., and Golding, L.A., 2002, Response of macroinvertebrates to copper and zinc in a stream mesocosm: Environmental Toxicology and Chemistry, v. 21, no. 9, p. 1854–1863.

Holcombe, G.W., Phipps, G.L., and Fiandt, J.T., 1983, Toxicity of selected priority pollutants to various aquatic organisms: Ecotoxicology and Environmental Safety, v. 7, no. 4, p. 400-409.

Holcombe, G.W., Phipps, G.L., and Marier, J.W., 1984, Methods for conducting snail (*Aplexa hypnorum*) embryo through adult exposures: effects of cadmium and reduced pH levels: Archives of Environmental Contamination and Toxicology, v. 13, no. 5, p. 627–634.

Hollis, L., McGeer, J.C., McDonald, D.G., and Wood, C.M., 1999, Cadmium accumulation, gill Cd binding, acclimation, and physiological effects during long term sublethal Cd exposure in rainbow trout: Aquatic Toxicology, v. 46, p. 101–119.

Hollis, L., McGeer, J.C., McDonald, D.G., and Wood, C.M., 2000a, Effects of long term sublethal Cd exposure in rainbow trout during soft water exposure: implications for biotic ligand modelling: Aquatic Toxicology, v. 51, p. 93–105.

Hollis, L., McGeer, J.C., McDonald, D.G., and Wood, C.M., 2000b, Protective effects of calcium against chronic waterborne cadmium exposure to juvenile rainbow trout: Environmental Toxicology and Chemistry, v. 19, no. 11, p. 2725–2734.

Hounslow, A.W., 1995, Water quality data: analysis and interpretation: Boca Raton, Fla., CRC Press, 397 p.

Ingersoll, C.G., Brumbaugh, W.G., Dwyer, F.J., and Kemble, N.E., 1994, Bioaccumulation of metals by *Hyallella azteca* exposed to contaminated sediments from the upper Clark Fork River, Montana.: Environmental Toxicology and Chemistry, v. 13, no. 12, p. 2013–2020.

Ingersoll, C.G., Brunson, E.L., Dwyer, F.J., Hardesty, D.K., and Kemble, N.E., 1998, Use of sublethal endpoints in sediment toxicity tests with the amphipod *Hyalella azteca*: Environmental Toxicology and Chemistry, v. 17, no. 8, p. 1508–1523.

Ingersoll, C.G., and Kemble, N.E., 2001, Revised description of toxicity data on cadmium: Chronic water-only exposures with the amphipod *Hyalella azteca* and the midge *Chironomus tentans.*, Letter of January 11, 2001, to Cindy Roberts, U.S. Environmental Protection Agency, Washington, D.C., U.S. Geological Survey, Columbia Ecological Research Center, Columbia, Mo., 5 p.

Ingersoll, C.G., and Winner, R.W., 1982, Effect on *Daphnia pulex* (De Geer) of daily pulse exposures to copper or cadmium: Environmental Toxicology and Chemistry, v. 1, no. 4, p. 321–327.

Irving, E.C., Baird, D.J., and Culp, J.M., 2003, Ecotoxicological responses of the mayfly *Baetis tricaudatus* to dietary and waterborne cadmium: implications for toxicity testing: Environmental Toxicology and Chemistry, v. 22, no. 5, p. 1058–1064.

Integrated Taxonomic Information System, 2002, Integrated Taxonomic Information System (ITIS) on-line database: accessed January 2005, at http://www.itis.usda.gov.

Jackson, B.P., Lasier, P.J., Miller, W.P., and Winger, P.W., 2000, Effects of calcium, magnesium, and sodium on alleviating cadmium toxicity to *Hyalella azteca*: Bulletin of Environmental Contamination and Toxicology, v. 64, p. 279–286.

James, S.M., and Little, E.E., 2003, The effects of chronic cadmium exposure on American toad (*Bufo americanus*) tadpoles: Environmental Toxicology and Chemistry, v. 22, no. 2, p. 377–380.

James, S.M., Little, E.E., and Semlitsch, R.D., 2005, Metamorphosis of two amphibian species after chronic cadmium exposure in outdoor mesocosms: Environmental Toxicology and Chemistry, v. 24, no. 8, p. 1994–2001.

Jarvinen, A.W., and Ankley, G.T., 1999, Linkage of effects to tissue residues: development of a comprehensive database for aquatic organisms exposed to inorganic and organic chemicals: Pensacola, Fla., Society of Environmental Toxicology and Chemistry (SETAC), 364 p.

Jensen, A., and Forbes, V.E., 2001, Interclonal variation in the acute and delayed toxicity of cadmium to the European prosobranch gastropod *Potamopyrgus antipodarum* (Gray): Archives of Environmental Contamination and Toxicology, v. 40, no. 2, p. 230–235.

Jensen, A., Forbes, V.E., and Parker, E.D., Jr, 2001, Variation in cadmium uptake, feeding rate, and life-history effects in the gastropod *Potamopyrgus antipodarum*: linking toxicant effects on individuals to the population level: Environmental Toxicology and Chemistry, v. 20, no. 11, p. 2503–2513.

Jop, K.M., Askew, A.M., and Foster, R.B., 1995, Development of a water-effect ratio for copper, cadmium, and lead for the Great Works River in Maine, using *Ceriodaphnia dubia* and *Salvelinus fontinalis*: Bulletin of Environmental Contamination and Toxicology, v. 54, p. 29–35.

Karchesky, C.M., and Bennett, D.H., 1999, Dietary overlap between introduced fishes and juvenile salmonids in lower Granite Reservoir, Idaho-Washington, *in* Oregon Department of Fish and Wildlife and National Marine Fisheries Service 1999 Management Implications of Co-occurring Native and Introduced Fishes: Proceedings of the Workshop, October 27-28, 1998, Portland, Oreg., p. 145-154.

Karntanut, W., and Pascoe, D., 2000, A comparison of methods for measuring acute toxicity to *Hydra vulgaris*: Chemosphere, v. 41, no. 10, p. 1543–1548.

Karntanut, W., and Pascoe, D., 2002, The toxicity of copper, cadmium and zinc to four different *Hydra* (Cnidaria: Hydrozoa): Chemosphere, v. 47, no. 10, p. 1059–1064.

Karr, J.R., 1991, Biological integrity: a long neglected aspect of water resource management: Ecological Applications, v. 1, no. 1, p. 66–84.

Keller, A.E., and Zam, S.G., 1991, The acute toxicity of selected metals to the freshwater mussel, *Anodonta imbecilis*: Environmental Toxicology and Chemistry, v. 10, no. 4, p. 539–546.

Kiffney, P.M., and Clements, W.H., 1993, Bioaccumulation of heavy metals by benthic invertebrates at the Arkansas River, Colorado: Environmental Toxicology and Chemistry, v. 12, p. 1507–1518.

Kiffney, P.M., and Clements, W.H., 1996, Size-dependent response of macroinvertebrates to metals in experimental streams: Environmental Toxicology and Chemistry, v. 15, no. 8, p. 1352–1356.

Kiffney, P.M., and Clements, W.H., 2002, Ecological effects of metals on benthic invertebrates, *in* Simon, T.P., and Stewart, P.M., eds., Biological Response Signatures: Multimetric index patterns for assessment of freshwater aquatic assemblages: Boca Raton, Fla, CRC Press, p. 135–154.

Kim, S.D., Ma, H., Allen, H.E., and Cha, D.K., 1999, Influence of dissolved organic matter on the toxicity of copper to *Ceriodaphnia dubia:* effect of complexation kinetics: Environmental Toxicology and Chemistry, v. 18, no. 11, p. 2433–2437.

Kislalioglu, M., Scherer, E., and McNicol, R.E., 1996, Effects of cadmium on foraging behaviour of lake charr, *Salvelinus namaycush*: Environmental Biology of Fishes, v. 46, no. 1, p. 75–82.

Kuhn, A., Munns, W.R., Jr., Serbst, J., Edwards, P., Cantwell, M.G., Gleason, T., Pelletier, M.C., and Berry, W., 2002, Evaluating the ecological significance of laboratory response data to predict population-level effects for the estuarine amphipod *Ampelisca abdita*: Environmental Toxicology and Chemistry, v. 21, no. 4, p. 865–874.

Kumada, H., Kimura, S., Yokote, M., and Matida, Y., 1973, Acute and chronic toxicity, uptake and retention of cadmium in freshwater organisms: Bulletin of the Freshwater Fisheries Research Laboratory of Tokyo, v. 22, no. 2, p. 157–165.

Lawrence, S.G., and Holoka, M.H., 1991, Response of crustacean zooplankton impounded in situ to cadmium at low environmental concentrations: Verhandlungen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie, v. 24, p. 2254–2259.

Lawrence, S.G., Holoka, M.H., Hunt, R.V., and Hesslein, R.H., 1996, Multi-year experimental additions of cadmium to a lake epilimnion and resulting water column cadmium concentrations: Canadian Journal of Fisheries and Aquatic Sciences, v. 53, p. 1876–1887.

Lewis, P.A., and Weber, C.I., 1985, A study of the reliability of *Daphnia* acute toxicity tests, *in* Cardwell, R.D., Purdy, R., and Bahner, R.C., eds., Aquatic Toxicology and Hazard Assessment: Seventh Symposium, ASTM STP 854: Philadelphia, Pa., American Society for Testing and Materials (ASTM), p. 73–86.

Ma, H., Kim, S.D., Cha, D.K., and Allen, H.E., 1999, Effects of kinetics of complexation by humic acid on toxicity of copper to *Ceriodaphnia dubia*: Environmental Toxicology and Chemistry, v. 18, no. 5, p. 828–837.

MacRae, R.K., Smith, D.E., Swoboda-Colberg, N., Meyer, J.S., and Bergman, H.L., 1999, Copper binding affinity of rainbow trout (*Oncorhynchus mykiss*) and brook trout (*Salvelinus fontinalis*) gills: implications for assessing bioavailable metal: Environmental Toxicology and Chemistry, v. 18, no. 6, p. 1180–1189.

Malley, D.F., 1996, Cadmium whole-lake experiment at the Experimental Lakes Area: an anachoronism?: Canadian Journal of Fisheries and Aquatic Sciences, v. 53, no. 8, p. 1862–1870.

Maltby, L., Blake, N., Brock, T.C.M., and Van den Brink, P.J., 2005, Insecticide species sensitivity distributions: importance of test species selection and relevance to aquatic ecosystems: Environmental Toxicology and Chemistry, v. 24, no. 2, p. 379–388.

Maret, T.R., Cain, D.J., MacCoy, D.E., and Short, T.M., 2003, Response of benthic invertebrate assemblages to metals exposure and bioaccumulation associated with hard-rock mining in northwestern streams, U.S.A.: Journal of the North American Benthological Society, v. 22, no. 4, p. 598–620.

Maret, T.R., and MacCoy, D.E., 2002, Fish assemblages and environmental variables associated with hard-rock mining in the Coeur d'Alene River basin, Idaho: Transactions of the American Fisheries Society, v. 131, no. 5, p. 865–884.

Maret, T.R., MacCoy, D.E., Skinner, K.D., Moore, S.E., and O'Dell, I., 2001, Evaluation of macroinvertebrate assemblages in Idaho rivers using multimetric and multivariate techniques, 1996-1998: U.S. Geological Survey Water-Resources Investigations Report 01-4145, 69 p.

Marr, J.C.A., Bergman, H.L., Parker, M., Lipton, J., Cacela, D., Erickson, W., and Phillips, G., 1995, Relative sensitivity of brown and rainbow trout to pulsed exposures of an acutely lethal mixture of metals typical of the Clark Fork River, Montana: Canadian Journal of Fisheries and Aquatic Sciences, v. 52, no. 9, p. 2005–2015.

Marshall, J.S., 1978, Population dynamics of *Daphnia galeata mendotae* as modified by chronic cadmium stress: Journal of the Fisheries Research Board of Canada, v. 35, no. 4, p. 461–469.

Martin, T.R., and Holdich, D.M., 1986, The acute lethal toxicity of heavy metals to peracarid crustaceans (with particular reference to fresh-water asellids and gammarids): Water Research, v. 20, no. 9, p. 1137-1147.

Mathias, J.A., 1971, Energy flow and secondary production of the amphipds *Hyalella azteca* and *Crangonyx richmondensis occidentalis* in Marion Lake, British Columbia: Journal of the Fisheries Research Board of Canada, v. 28, no. 5, p. 711–726.

Matthews, W.J., 1998, Patterns in freshwater fish ecology: New York, Chapman and Hall, 732 p.

McCarty, L.S., and Mackay, D., 1993, Enhancing ecotoxicological modeling and assessment: body residues and modes of toxic action: Environmental Science and Technology, v. 27, no. 9, p. 1718–1728.

McDonald, D.G., and Wood, C.M., 1993, Branchial mechanisms of acclimation to metals in freshwater fish, *in* Rankin, J.C., and Jensen, F.B., eds., Fish Ecophysiology: London, Chapman and Hall, p. 297–321.

McGeer, J.C., Brix, K.V., Skeaff, J.M., DeForest, D.K., Brigham, S.I., Adams, W.J., and Green, A.S., 2003, Inverse relationship between bioconcentration factor and exposure concentration for metals: implications for hazard assessment of metals in the aquatic environment: Environmental Toxicology and Chemistry, v. 22, no. 5, p. 1017–1037.

McGeer, J.C., Szebedinszky, C., McDonald, D.G., and Wood, C.M., 2000a, Effects of chronic sublethal exposure to waterborne Cu, Cd or Zn in rainbow trout. 1: Iono-regulatory disturbance and metabolic costs: Aquatic Toxicology, v. 50, p. 231–243.

McGeer, J.C., Szebedinszky, C., McDonald, D.G., and Wood, C.M., 2000b, Effects of chronic sublethal exposure to waterborne Cu, Cd or Zn in rainbow trout. 2: tissue specific metal accumulation: Aquatic Toxicology, v. 50, p. 245–256.

McKim, J.M., 1977, Evaluation of tests with early life stages of fish for predicting long-term toxicity: Journal of the Fisheries Research Board of Canada, v. 34, no. 8, p. 1148–1154.

McKim, J.M., 1985, Early life stage toxicity tests, *in* Rand, G.M., and Petrocelli, S.R., eds., Fundamentals of Aquatic Toxicology: Methods and Applications: New York, Hemisphere Publishing, p. 58-95.

McNicol, R.E., 1997, The influence of environmental factors on the preference-avoidance responses of lake whitefish (*Coregonus clupeaformis*) to cadmium: Winnipeg, Manitoba, University of Manitoba, Ph.D. thesis, 146 p.

McNicol, R.E., Bégout-Anras, M.-L., and Scherer, E., 1999, Influence of light preferences on the avoidance responses of lake whitefish, *Coregonus clupeaformis*, to cadmium: Environmental Biology of Fishes, v. 55, no. 3, p. 295-306.

McNicol, R.E., and Scherer, E., 1991, Behavioural responses of lake whitefish (*Coregonus clupeaformis*) to cadmium during preference-avoidance testing: Environmental Toxicology and Chemistry, v. 10, no. 2, p. 225-234.

McNulty, E.W., Dwyer, F.J., Ellersieck, M.R., Greer, E.I., Ingersoll, C.G., and Rabeni, C.F., 1999, Evaluation of ability of reference toxicity tests to identify stress in laboratory populations of the amphipod *Hyalella azteca*: Environmental Toxicology and Chemistry, v. 18, no. 3, p. 544-548.

Mebane, C.A., 2002a, Effects of metals on freshwater macroinvertebrates: a review and case study of the correspondence between a multimetric index, toxicity testing, and copper concentrations in sediment and water, *in* Simon, T.P., ed., Biological response signatures: indicator patterns using aquatic communities: Boca Raton, Fla., CRC Press, p. 281-306.

Mebane, C.A., 2002b, Stream Fish Indexes, *in* Grafe, C.S., ed., Idaho Small Stream Ecological Assessment Framework: an Integrated Approach: Boise, Idaho Department of Environmental Quality, p. 4-1 to 4-65.

Mebane, C.A., 2003, Comparisons of cadmium criteria to the results of toxicity testing with species resident to the South Fork Coeur d'Alene River: Boise, Idaho, Idaho Department of Environmental Quality. 44 p, accessed March 2005 at http://www.deq.state.id.us/water/data_reports/surface_water/monitoring/site_specific_criteria.cfm.

Mebane, C.A., Maret, T.R., and Hughes, R.M., 2003, An index of biological integrity (IBI) for Pacific Northwest rivers: Transactions of the American Fisheries Society, v. 132, no. 2, p. 239-261.

Meyer, J.S., 1999, A mechanistic explanation for the ln (LC50) vs ln (hardness) adjustment equation for metals: Environmental Science and Technology, v. 33, p. 908-912.

Meyer, J.S., Adams, W.J., Brix, K.V., Luoma, S.N., Mount, D.R., Stubblefield, W.A., and Wood, C.M., eds., 2005, Toxicity of dietborne metals to aquatic organisms: Pensacola, Fla., Society of Environmental Toxicology and Chemistry (SETAC), 303 p.

Mount, D.R., Ankley, G.T., Brix, K.V., Clements, W.H., Dixon, D.G., Fairbrother, A., Hickey, C.W., Lanno, R.P., Lee, C.M., Munns, W.R., Ringer, R.K., Staveley, J.P., Wood, C.M., Erickson, R.J., and Hodson, P.V., 2003, Effects assessment, *in* Reiley, M.C., Stubblefield, W.A., Adams, W.J., Di Toro, D.M., Hodson, P.V., Erickson, R.J., and Keating, F.J., eds., Reevaluation of the State of the Science for Water-Quality Criteria Development: Pensacola, Society of Environmental Toxicology and Chemistry (SETAC), p. 53–118.

Mount, D.R., Barth, A.K., Garrison, T.D., Barten, K.A., and Hockett, J.R., 1994, Dietary and waterborne exposure of rainbow trout *(Oncorhynchus mykiss)* to copper, cadmium, lead and zinc using a live diet: Environmental Toxicology and Chemistry, v. 13, no. 12, p. 2031–2042.

Mount, D.R., Mattson, V.R., Highland, T.L., Leonard, E.N., Erickson, R.J., Dawson, T., and Lott, K., 2001, Exposure of rainbow trout to dietary metals using live prey, *in* Society of Environmental Toxicology and Chemistry 21th Annual Meeting Abstracts, November 11–15, 2001 Baltimore, Society of Environmental Toxicology and Chemistry, Pensacola, Fla., Pensacola, FL,.

Muir, W.D., and Coley, T.C., 1996, Diet of yearling Chinook salmon and feeding success during downstream migration in the Snake and Columbia Rivers: Northwest Science, v. 70, no. 4, p. 298–305.

Munkittrick, K.R., and McMaster, M.E., 2000, Effects-driven assessment of multiple stressors using fish populations, *in* Ferenc, S.A., and Foran, J.A., eds., Multiple stressors in ecological risk and impact assessment: approaches to risk estimation: Pensacola, Fla, Society of Environmental Toxicology and Chemistry (SETAC), p. 27–66.

Myers, R., and Foster, A., 2003, Benthic macroinvertebrates of Hells Canyon: Boise, Idaho, Idaho Power Company, Technical Report Appendix E.3.1.8, Hells Canyon Hydroelectric Project New License Application (FERC No. 1971), 76 p., accessed October 2004 at http://www.idahopower.com/riversrec/relicensing/hellscanyon/hellspdfs/techappendices/list_techappend.pdf.

National Council of the Paper Industry for Air and Stream Improvement, Inc., 1989, Effects of biologically treated bleached kraft mill effluent on cold water stream productivity in experimental stream channels—fifth progress report: Research Triangle Park, N.C., National Council of the Paper Industry for Air and Stream Improvement, Inc. (NCASI), Technical Bulletin No. 566, 163 p.

Nebeker, A.V., Cairns, M.A., Onjukka, S.T., and Titus, R.H., 1986a, Effect of age on the sensitivity of *Daphnia magna* to cadmium, copper, and cyanazine: Environmental Toxicology and Chemistry, v. 5, p. 527–530.

Nebeker, A.V., Onjukka, S.T., Cairns, M.A., and Frawczyk, D.F., 1986b, Survival of *Daphnia magna* and *Hyalella azteca* in cadmium-spiked water and sediment: Environmental Toxicology and Chemistry, v. 5, no. 10, p. 933–938.

Nebeker, A.V., Schuytema, G.S., and Ott, S.L., 1994, Effects of cadmium on limb regeneration in the Northwestern salamander *Ambystoma gracile*: Archives of Environmental Contamination and Toxicology, v. 27, p. 318–322.

Nebeker, A.V., Schuytema, G.S., and Ott, S.L., 1995, Effects of cadmium on growth and bioaccumulation in the Northwestern salamander *Ambystoma gracile*: Archives of Environmental Contamination and Toxicology, v. 29, p. 492–499.

Nelson, J.S., Crossman., E.J., Espinosa-Pérez, H., Findley, L.T., Gilbert, C.R., Lea, R.N., and Williams, J.D., 2004, Common and scientific names of fishes from the United States, Canada, and Mexico, (6th ed.): Bethesda, Md., American Fisheries Society Special Publication 29, 386 p.

Newman, M.C., 1991, A statistical bias in the derivation of hardness-dependent metals criteria: Environmental Toxicology and Chemistry, v. 10, no. 10, p. 1295–1297.

Newman, M.C., Ownby, D.R., Mézin, L.C.A., Powell, D.C., Christensen, T.R.L., Lerberg, S.B., and Anderson, B.-A., 2000, Applying species-sensitivity distributions in ecological risk assessment: assumptions of distribution type and sufficient numbers of species: Environmental Toxicology and Chemistry, v. 19, no. 2, p. 508–515.

Niederlehner, B.R., Buikema, A.L., Jr., Pittinger, C.A., and Cairns, J., Jr., 1984, Effects of cadmium on the population growth of a benthic invertebrate *Aelosoma headleyi* (Oligochaeta): Environmental Toxicology and Chemistry, v. 2, no. 3, p. 255–262.

Niederlehner, B.R., Pratt, J.R., Buikema, A.L., Jr., and Cairns, J., Jr., 1985, Laboratory tests evaluating the effects of cadmium on freshwater protozoan communities: Environmental Toxicology and Chemistry, v. 4, p. 155–165.

Niemi, G.J., DeVore, P., Detenbeck, N., Taylor, D., Lima, A., Pastor, J., Yount, J.D., and Naiman, R.J., 1990, Overview of case studies on recovery of aquatic systems from disturbance: Environmental Management, v. 14, no. 5, p. 571–587.

Nimick, D.A., Cleasby, T.E., and McCleskey, R.B., 2005, Seasonality of diel cycles of dissolved trace-metal concentrations in a Rocky Mountain stream: Environmental Geology, v. 47, p. 603–614.

Nimick, D.A., Gammons, C.H., Cleasby, T.E., Madison, J.P., Skaar, D., and Brick, C.M., 2003, Diel cycles in dissolved metal concentrations in streams: Occurrence and possible causes: Water Resources Research, doi:10.1029/2002WR001571,v. 39, no. 9, p. 1247.

Niyogi, S., Couture, P., Pyle, G.G., McDonald, D.G., and Wood, C.M., 2004, Acute cadmium biotic ligand model characteristics of laboratory-reared and wild yellow perch (*Perca flavescens*) relative to rainbow trout (*Oncorhynchus mykiss*): Canadian Journal of Fisheries and Aquatic Sciences, v. 61, no. 6, p. 942–953.

Norberg-King, T.J., 1989, An evaluation of the fathead minnow seven-day subchronic test for estimating chronic toxicity: Environmental Toxicology and Chemistry, v. 8, no. 11, p. 1075–1089.

Norwood, W.P., Borgmann, U., Dixon, D.G., and Wallace, A., 2003, Effects of metal mixtures on aquatic biota: a review of observations and methods: Human and Ecological Risk Assessment, v. 9, no. 4, p. 795–811.

Owens, R.W., and Dittman, D.E., 2003, Shifts in the diets of slimy sculpin (*Cottus cognatus*) and lake whitefish (*Coregonus clupeaformis*) in Lake Ontario following the collapse of the burrowing amphipod *Diporeia*: Aquatic Ecosystem Health & Management, v. 6, no. 3, p. 311–323.

Palawski, D., Hunn, J.B., and Dwyer, F.J., 1985, Sensitivity of young striped bass to organic and inorganic contaminants in fresh and saline waters: Transactions of the American Fisheries Society, v. 114, no. 5, p. 748–753.

Paul, A.J., Post, J.R., and Stelfox, J.D., 2003, Can anglers influence the abundance of native and nonnative salmonids in a stream from the Canadian Rocky Mountains?: North American Journal of Fisheries Management, v. 23, no. 1, p. 109–119.

Perceval, O., Couillard, Y., Pinel-Alloul, B., Giguère, A., and Campbell, P.G.C., 2004, Metal-induced stress in bivalves living along a gradient of Cd contamination: relating sub-cellular metal distribution to population-level responses: Aquatic Toxicology, v. 69, no. 4, p. 327–345.

Pennak, R.W., 1978, Fresh-water Invertebrates of the United States, New York, John Wiley and Sons, 803 p.

Phipps, G.L., and Holcombe, G.W., 1985, A method for aquatic multiple species toxicant testing: acute toxicity of 10 chemicals to 5 vertebrates and 2 invertebrates: Environmental Pollution (Series A:), v. 38, p. 141–147.

Phipps, G.L., Mattson, V.R., and Ankley, G.T., 1995, Relative sensitivity of three freshwater benthic macroinvertebrates to ten contaminants: Archives of Environmental Contamination and Toxicology, v. 28, no. 3, p. 281–286.

Pickering, Q.H., and Gast, M.H., 1972, Acute and chronic toxicity of cadmium to the fathead minnow (*Pimephales promelas*): Journal of the Fisheries Research Board of Canada, v. 29, no. 8, p. 1099–1106.

Playle, R.C., 1998, Modelling metal interactions at fish gills: Science of the Total Environment, v. 219, no. 2-3, p. 147–163.

Playle, R.C., 2004, Using multiple metal–gill binding models and the toxic unit concept to help reconcile multiple-metal toxicity results: Aquatic Toxicology, v. 67, no. 4, p. 359–370.

Playle, R.C., Dixon, D.G., and Burnison, B.K., 1993, Copper and cadmium binding to fish gills: modification by dissolved organic carbon and synthetic ligands: Canadian Journal of Fisheries and Aquatic Sciences, v. 50, p. 2667–2677.

Prothro, M.G., 1993, Office of Water policy and technical guidance on interpretation and implementation of aquatic life metals criteria: Washington, D.C., U.S. Environmental Protection Agency. 52 p., accessed July 2006 at http://dspace.udel.edu:8080/dspace/handle/19716/1556.

Posthuma, L., Traas, T.P., and Suter, G.W., II, 2002, General introduction to species sensitivity distributions, *in* Posthuma, L., Suter, G.W., II, and Traas, T.P., eds., Species Sensitivity Distributions in Ecotoxicology: Boca Raton, Fla., CRC Press, p. 3–32.

Power, M., and McCarty, L.S., 1997, Fallacies in ecological risk assessment practices Environmental Science and Technology, v. 31, no. 8, p. A370–A375

Quinn, T.P., 2005, The behavior and ecology of Pacific salmon and trout: Seattle, Wash., American Fisheries Society and University of Washington, 328 p.

Rainbow, P.S., 2002, Trace metal concentrations in aquatic invertebrates: why and so what?: Environmental Pollution, v. 120, no. 3, p. 497–507.

Ricker, W.E., 1975, Computation and interpretation of biological statistics of fish populations: Ottawa, Fisheries Research Board of Canada Bulletin 191, 382 p.

Riddell, D.J., Culp, J.M., and Baird, D.J., 2005a, Behavioral responses to sublethal cadmium exposure within an experimental aquatic food web: Environmental Toxicology and Chemistry, v. 24, no. 2, p. 431–441.

Riddell, D.J., Culp, J.M., and Baird, D.J., 2005b, Sublethal effects of cadmium on prey choice and capture efficiency in juvenile brook trout (*Salvelinus fontinalis*): Environmental Toxicology and Chemistry, v. 24, no. 7, p. 1751–1758.

Rombough, P.J., and Garside, E.T., 1982, Cadmium toxicity and accumulation in eggs and alevins of Atlantic salmon *Salmo salar*: Canadian Journal of Zoology, v. 60, no. 8, p. 2006–2014.

Rose, K.A., 2000, Why are quantitative relationships between environmental quality and fish populations so elusive?: Ecological Applications, v. 10, p. 367–385.

Roy, I., and Hare, L., 1999, Relative importance of water and food as cadmium sources to the predatory insect *Sialis velata* (Megaloptera): Canadian Journal of Fisheries and Aquatic Sciences, v. 56, no. 7, p. 1143–1149

Ruud, D.F., 1996, A comparison of the macroinvertebrate communities of a trace elements enriched lake and an uncontaminated lake in north Idaho: the effects of mine waste contamination in Coeur d'Alene Lake: Cheney, Wash., Eastern Washington University, M.S. thesis, 65 p.

Santore, R.C., Paquin, P.R., Di Toro, D.M., Allen, H.E., and Meyer, J.S., 2001, Biotic ligand model of the acute toxicity of metals. 2. Application to acute copper toxicity in freshwater fish and *Daphnia*: Environmental Toxicology and Chemistry, v. 20, no. 10, p. 2397–2402.

Sappington, L.C., Mayer, F.L., Dwyer, F.J., Buckler, D.R., Jones, J.R., and Ellersieck, M.R., 2001, Contaminant sensitivity of threatened and endangered fishes compared to standard surrogate species: Environmental Toxicology and Chemistry, v. 20, no. 12, p. 2869–2876.

Sauter, S., Buxton, K.S., Macek, K.J., and Petrocelli, S.R., 1976, Effects of exposure to heavy metals on selected freshwater fish: toxicity of copper, cadmium, chromium and lead to eggs and fry of seven fish species: Duluth, Minn., U.S. Environmental Protection Agency, EPA-600/3-76-105, 85 p.

Scherer, E., McNicol, R.E., and Evans, R.E., 1997, Impairment of lake trout foraging by chronic exposure to cadmium: a black-box experiment: Aquatic Toxicology, v. 37, no. 1, p. 1–7.

Schindler, D.W., 1987, Detecting ecosystem responses to anthropogenic stress: Canadian Journal of Fisheries and Aquatic Sciences, v. 44 (suppl. 1), p. 6–25.

Schindler, D.W., 1988, Experimental studies of chemical stressors on whole lake ecosystems: Verhandlungen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie, v. 23, p. 11–41.

Schindler, D.W., Mills, K.H., Malley, D.F., Findlay, D.L., Shearer, J.A., Davies, I.J., Turner, M.A., Linsey, G.A., and Cruikshank, D.R., 1985, Long-term ecosystem stress—the effects of years of experimental acidification on a small lake: Science, v. 228, no. 6, p. 1395–1401.

Schreck, C.B., and Lorz, H.W., 1978, Stress response of coho salmon (*Oncorhynchus kisutch*) elicited by cadmium and copper and potential use of cortisol as an indicator of stress: Journal of the Fisheries Research Board of Canada, v. 35, p. 1124–1129.

Schwartz, M.L., Curtis, P.J., and Playle, R.C., 2004, Influence of natural organic matter source on acute copper, lead, and cadmium toxicity to rainbow trout (*Oncorhynchus mykiss*): Environmental Toxicology and Chemistry, v. 23, no. 12, p. 2889–2899.

Schubauer-Berigan, M.K., Dierkes, J.R., Monson, P.D., and Ankley, G.T., 1993, pH-dependent toxicity of Cd, Cu, Ni, Pb, and Zn to *Ceriodaphnia dubia, Pimephales promelas, Hyalella azteca*, and *Lumbriculus variegatus*: Environmental Toxicology and Chemistry, v. 12, no. 7, p. 1261–1266.

Scott, G.R., and Sloman, K.A., 2004, The effects of environmental pollutants on complex fish behaviour: integrating behavioural and physiological indicators of toxicity: Aquatic Toxicology, v. 68, no. 4, p. 369–392.

Scott, G.R., Sloman, K.A., Rouleau, C., and Wood, C.M., 2003, Cadmium disrupts behavioural and physiological responses to alarm substance in juvenile rainbow trout (*Oncorhynchus mykiss*): Journal of Experimental Biology, v. 206, no. 11, p. 1779–1790.

Seim, W.K., Curtis, L.R., Glenn, S.W., and Chapman, G.A., 1984, Growth and survival of developing steelhead trout (*Salmo gairdneri*) continuously or intermittently exposed to copper: Canadian Journal of Fisheries and Aquatic Sciences, v. 41, no. 3, p. 433–438,

Selck, H., Riemann, B., Christoffersen, K., Forbes, V.E., Gustavson, K., Hansen, B.W., Jacobsen, J.A., K, K.O., and Petersen, S., 2002, Comparing sensitivity of ecotoxicological effect endpoints between laboratory and field: Ecotoxicology and Environmental Safety, v. 52, no. 2, p. 97–112.

Shaw, J.R., Dempsey, T.D., Chen, C.Y., Hamilton, J.W., and Folt, C.L., 2006, Comparative toxicity of cadmium, zinc, and mixtures of cadmium and zinc to Daphnids: Environmental Toxicology and Chemistry, v. 25, no. 1, p. 182–189.

Shepard, B.K., 1998, Quantification of ecological risks to aquatic biota from bioaccumulated chemicals, National Sediment Bioaccumulation Conference Proceedings: Washington, D.C., U.S. Environmental Protection Agency, Office of Water, EPA 823-R-98-002, p. 2-31 – 2-52, accessed December 2005 at http://www.epa.gov/OST/cs/confprod.html.

Sinley, J.R., Goettl, J.P.J., and Davies, P.H., 1974, The effects of zinc on rainbow trout (*Salmo gairdneri*) in hard and soft water: Bulletin of Environmental Contamination and Toxicology, v. 12, no. 2, p. 193–201.

Sloman, K.A., Baker, D.W., Ho, C.G., McDonald, D.G., and Wood, C.M., 2003a, The effects of trace metal exposure on agonistic encounters in juvenile rainbow trout, *Oncorhynchus mykiss*: Aquatic Toxicology, v. 63, no. 2, p. 187–196.

Sloman, K.A., Lepage, O., Rogers, J.T., Wood, C.M., and Winberg, S., 2005, Socially-mediated differences in brain monoamines in rainbow trout: effects of trace metal contaminants: Aquatic Toxicology, v. 71, no. 3, p. 237–247.

Sloman, K.A., Wood, C.M., and McDonald, D.G., 2003b, Cadmium affects the social behaviour of rainbow trout, *Oncorhynchus mykiss*: Aquatic Toxicology, v. 65, no. 2, p. 171–185.

Smith, E.P., and Cairns, J., Jr., 1993, Extrapolation methods for setting ecological standards for water quality: statistical and ecological concerns: Ecotoxicology, v. 2, no. 3, p. 203–219.

Smolders, R., Baillieul, M., and Blust, R., 2005, Relationship between the energy status of *Daphnia magna* and its sensitivity to environmental stress: Aquatic Toxicology, v. 73, no. 2, p. 155–170.

Snell, T.W., and Carmona, M.J., 1995, Comparative toxicant sensitivity of sexual and asexual reproduction the rotifer *Brachionus calyciflorus*: Environmental Toxicology and Chemistry, v. 14, no. 3, p. 415–420.

Sorensen, E.M.B., 1991, Metal poisoning in fish: Boca Raton, Fla. CRC Press, p. 175–234.

Spehar, R.L., 1976, Cadmium and zinc toxicity to flagfish, *Jordanella floridae*: Journal of the Fisheries Research Board of Canada, v. 33, p. 1939–1945.

Spehar, R.L., Anderson, R.L., and Fiandt, J.T., 1978a, Toxicity and bioaccumulation of cadmium and lead in aquatic invertebrates: Environmental Pollution, v. 15, p. 195–208.

Spehar, R.L., and Carlson, A.R., 1984, Derivation of site-specific water quality criteria for cadmium in the St. Louis River basin, Duluth, Minnesota.: Environmental Toxicology and Chemistry, v. 3, no 4, p. 651–665.

Spehar, R.L., and Fiandt, J.T., 1986, Acute and chronic effects of water quality criteria-based metal mixtures on three aquatic species: Environmental Toxicology and Chemistry, v. 5, no. 10, p. 917–931.

Spehar, R.L., Leonard, E.N., and DeFoe, D.L., 1978b, Chronic effects of cadmium and zinc mixtures on flagfish (*Jordanella floridae*): Transactions of the American Fisheries Society, v. 107, no. 2, p. 354–360.

Spencer, M., and Ferson, S., 1997a, RAMAS Ecotoxicology, version 1.0a: Setauket, New York, Applied Biomathematics,

Spencer, M., and Ferson, S., 1997b, RAMAS Ecotoxicology user manual: ecological risk assessment for structured populations: Setauket, New York, Applied Biomathematics. 81 p.

Sprague, J.B., 1985, Factors that modify toxicity, *in* Rand, G.M., and Petrocelli, S.R., eds., Fundamentals of aquatic toxicology – methods and applications: New York, Hemisphere Publishing, p. 124–163.

Spromberg, J.A., and Meador, J.P., 2005, Relating results of chronic toxicity responses to population-level effects: modeling effects on wild Chinook salmon populations: Integrated Environmental Assessment and Management, v. 1, no. 1, p. 9–21.

Stackhouse, R.A., and Benson, W.H., 1988, The influence of humic acid on the toxicity and bioavailability of selected trace metals: Aquatic Toxicology, v. 13, no. 2, p. 99–107.

Stanley, J.K., Brooks, B.W., and La Point, T.W., 2005, A comparison of chronic cadmium effects on *Hyalella azteca* in effluent-dominated stream mesocosms to similar laboratory exposures in effluent and reconstituted hard water: Environmental Toxicology and Chemistry, v. 24, no. 4, p. 902–908.

Stark, J.D., 2005, How closely do acute lethal concentration estimates predict effects of toxicants on populations?: Integrated Environmental Assessment and Management, v. 1, no. 2, p. 109–113.

Stephan, C.E., 1995, Derivation of conversion factors for the calculation of dissolved freshwater aquatic life criteria for metals: Duluth, Minn., U.S. Environmental Protection Agency, Environmental Research Laboratory, variously paged.

Stephan, C.E., 2002, Use of species sensitivity distributions in the derivation of water quality criteria for aquatic life by the U.S. Environmental Protection Agency, *in* Posthuma, L., Suter, G.W., II, and Traas, T.P., eds., Species sensitivity distributions in ecotoxicology: Boca Raton, Fla., CRC Press, p. 211–220.

Stephan, C.E., Mount, D.I., Hansen, D.J., Gentile, J.H., Chapman, G.A., and Brungs, W.A., 1985, Guidelines for deriving numerical national water quality criteria for the protection of aquatic organisms and their uses: Duluth, Narragansett, and Corvallis, U.S. Environmental Protection Agency, EPA 822-R-85-100, NTIS PB85 227049, 98 p., accessed January 2005 at http://epa.gov/waterscience/criteria/aqlife.html.

Stephan, C.E., Peltier, W.H., Hansen, D.J., Delos, C.G., and Chapman, G.A., 1994a, Appendix C: Guidance concerning the use of "clean techniques" and QA/QC when measuring trace metals, Interim guidance on determination and use of water-effect ratios for metals: Washington, D.C., U.S. Environmental Protection Agency, p. 98–108.

Stephan, C.E., Peltier, W.H., Hansen, D.J., Delos, C.G., and Chapman, G.A., 1994b, Interim guidance on determination and use of water-effect ratios for metals: Washington, D.C., U.S. Environmental Protection Agency, EPA-823-B-94-001, 154 p.

Stephenson, M., and Mackie, G.L., 1988, Multivariate analysis of correlations between environmental parameters and cadmium concentrations in *Hyalella azteca* (Crustacea: Amphipoda) from central Ontario lakes: Canadian Journal of Fisheries and Aquatic Sciences, v. 45, no. 10, p. 1705–1710.

Stephenson, M., and Turner, M.A., 1993, A field study of cadmium dynamics in periphyton and in *Hyalella azteca* (crustacea: amphipoda): Water, Air, and Soil Pollution, v. 68, no. 3-4, p. 341–361.

Strong, D.R., Jr., 1972, Life history variation among populations of an amphipod (*Hyalella Azteca*): Ecology, v. 53, no. 6, p. 1103–1111.

Stubblefield, W.A., 1990, An evaluation of the acute toxicity of cadmium chloride ($CdCl_2$) to brown trout (*Salmo trutta*), rainbow trout (*Oncorhynchus mykiss*), and mountain whitefish (*Prosopium williamsoni*): Fort Collins, Colo. and Laramie, Wyo., ENSR Consulting and Engineering and University of Wyoming Red Buttes Environmental Biology Laboratory, variously paged.

Stubblefield, W.A., Steadman, B.L., La Point, T.W., and Bergman, H.L., 1999, Acclimation-induced changes in the toxicity of zinc and cadmium to rainbow trout: Environmental Toxicology and Chemistry, v. 18, no. 12, p. 2875–2881.

Stuhlbacher, A., Bradley, M.C., Naylor, C., and Calow, P., 1992, Induction of cadmium tolerance in two clones of *Daphnia magna* Straus: Comparative Biochemistry and Physiology Part C, Comparative Pharmacology, v. 101, no. 3, p. 571–577.

Stuhlbacher, A., Bradley, M.C., Naylor, C., and Calow, P., 1993, Variation in the development of cadmium resistance in *Daphnia magna* Straus; effect of temperature, nutrition, age and genotype: Environmental Pollution, v. 80, no. 2, p. 153–158.

Suedel, B.C., Rodgers, J.H., Jr., and Deaver, E., 1997, Experimental factors that may affect toxicity of cadmium to freshwater organisms: Archives of Environmental Contamination and Toxicology, v. 33, no. 2, p. 188–193.

Sullivan, J.F., Atchison, G.J., Kolar, D.J., and McIntosh, A.W., 1978, Changes in the predator-prey behavior of fathead minnows (*Pimephales Promelas*) and largemouth bass (*Micropterus Salmoides*) caused by cadmium: Journal of the Fisheries Research Board of Canada, v. 35, no. 4, p. 446–451.

Suter, G.W., II, Norton, S.B., and Cormier, S.M., 2002a, A method for inferring the causes of observed impairments in aquatic ecosystems: Environmental Toxicology and Chemistry, v. 21, no. 6, p. 1101–1111.

Suter, G.W., II, Rosen, A.E., Linder, E., and Parkhurst, D.F., 1987, Endpoints for responses of fish to chronic toxic exposures: Environmental Toxicology and Chemistry, v. 6, p. 793–809.

Suter, G.W., II, Traas, T.P., and Posthuma, L., 2002b, Issues and practices in the derivation and use of species sensitivity distributions, *in* Posthuma, L., Suter, G.W., II, and Traas, T.P., eds., Species Sensitivity Distributions in Ecotoxicology: Boca Raton, Fla., CRC Press, p. 437–474.

Suttle, K.B., Power, M.E., Levine, J.M., and McNeely, C., 2004, How fine sediment in riverbeds impairs growth and survival of juvenile salmonids: Ecological Applications, v. 14, no. 4, p. 969–974.

Szebedinszky, C., McGeer, J.C., McDonald, D.G., and Wood, C.M., 2001, Effects of chronic Cd exposure via the diet or water on internal organ-specific distribution and subsequent gill Cd uptake kinetics in juvenile rainbow trout (*Oncorhynchus mykiss*): Environmental Toxicology and Chemistry, v. 20, no. 3, p. 597–607.

Thomas, D.G., Cryer, A., Solbé, J.F.d.L.G., and Kay, J., 1983, A comparison of the accumulation and protein binding of environmental cadmium in the gills, kidney and liver of rainbow trout (*Salmo gairdneri*): Comparative Biochemistry and Physiology Part C, Comparative Pharmacology, v. 76C, no. 2, p. 241–246.

Thomson Scientific, 2005, Web of Science® [database], accessed January–May 2005, at http://scientific.thomson.com/products/wos/.

Thorp, J.H., and Covich., A.P., eds., 2001, Ecology and classification of North American freshwater invertebrates: San Diego, Academic Press, 1056 p.

U.S. Environmental Protection Agency, 1984, Ambient water quality criteria for cadmium: Duluth, Minn., U.S. Environmental Protection Agency, EPA 440/5-84-032, 133 p., accessed February 2005 at http://epa.gov/waterscience/criteria/aqlife.html.

U.S. Environmental Protection Agency, 1991, Trimmed Spearman-Karber (TSK) Program, version 1.5: Cincinnati Ecological Monitoring Research Division, Environmental Monitoring Systems Laboratory, U.S. Environmental Protection Agency.

U.S. Environmental Protection Agency, 1992, EPA Probit Analysis Program, version 1.5: Cincinnati Ecological Monitoring Research Division, Environmental Monitoring Systems Laboratory, U.S. Environmental Protection Agency.

U.S. Environmental Protection Agency, 1994, Water quality standards handbook: Washington, D.C, U.S. Environmental Protection Agency, EPA-823-B-94-005a, accessed September 2004 at http://www.epa.gov/ost/standards/handbook/.

U.S. Environmental Protection Agency, 2001, update of the ambient water quality criteria for cadmium: Washington, D.C., U.S. Environmental Protection Agency, EPA/822/R-01-001. 266 p., accessed September 2004 at http://epa.gov/waterscience/criteria/aqlife.html.

U.S. Environmental Protection Agency, 2002a, ECOTOX User Guide: ECOTOXicology Database System. Version 3.0.: U.S. Environmental Protection Agency, National Health and Environmental Research Laboratory, accessed December 2004, at http://www.epa.gov/ecotox/.

U.S. Environmental Protection Agency, 2002b, National recommended water quality criteria: 2002: Washington, D.C., U.S. Environmental Protection Agency, EPA-822-R-02-047. 36 p., accessed September 2005 at http://epa.gov/waterscience/criteria/wqcriteria.html.

U.S. Environmental Protection Agency, 2002c, Short-term methods for estimating the chronic toxicity of effluents and receiving waters to freshwater organisms, 4th ed: Cincinnati, Ohio, U.S. Environmental Protection Agency, EPA-821-R-02-013. 350 p., accessed June 2004 at http://www.epa.gov/waterscience/WET/.

U.S. Environmental Protection Agency, 2003a, 2003 update of ambient water quality criteria for copper: Washington, D.C., U.S. Environmental Protection Agency, Office of Water, Office of Science and Technology, EPA 822-R-03-026. 86 p., accessed January 2004 at http://www.epa.gov/waterscience/criteria/copper/.

U.S. Environmental Protection Agency, 2003b, Draft methodology for conducting biological evaluations of aquatic life criteria: methods manual: Washington, D.C., U.S. Environmental Protection Agency, Office of Science and Technology, EPA-823-B-03-002. 85 p.

U.S. Geological Survey, 2005, Biological Transactional Database (BioTDB) for NAWQA ecological data: U.S. Geological Survey, accessed December 2005 [as of July 2006, data requests from the public can be submitted electronically to gs.nawqa.data@usgs.gov]

van der Hoeven, N., Noppert, F., and Leopold, A., 1997, How to measure no effect. Part I: towards a new measure of chronic toxicity in ecotoxicology. Introduction and workshop results: Environmetrics, v. 8, no. 3, p. 241–248.

Versteeg, D.J., Belanger, S.E., and Carr, G.J., 1999, Understanding single-species and model ecosystem sensitivity –data-based comparison: Environmental Toxicology and Chemistry, v. 18, no. 6, p. 1329–1346.

Wang, F., Goulet, R.R., and Chapman, P.M., 2004, Testing sediment biological effects with the freshwater amphipod *Hyalella azteca:* the gap between laboratory and nature: Chemosphere, v. 57, no. 11, p. 1713–1724

Ward, T.J., and Robinson, W.E., 2005, Evolution of cadmium resistance in *Daphnia magna*: Environmental Toxicology and Chemistry, v. 24, no. 9, p. 2341–2349.

Weast, R.C., ed., 1990, CRC handbook of chemistry and physics (70th ed.): Boca Raton, Fla., CRC Press, 2502 p.

Welsh, P.G., Chapman, G.A., Hansen, J.A., and Lipton, J., 2001, Importance of ionic composition of reconstituted laboratory test water in interpreting metal toxicity test results, *in* Greenberg, B.M., Hull, R.N., Roberts, M.H., and Gensemer, R.W., eds., Environmental toxicology and risk assessment: science, policy, and standardization—implications for environmental decisions: Tenth Volume ASTM STP 1403: West Conshohoken, Pa., American Society for Testing and Materials (ASTM), p. 3–15.

Welsh, P.G., Lipton, J., Podrabsky, T.L., and Chapman, G.A., 2000, Relative importance of calcium and magnesium in hardness-based modification of copper toxicity: Environmental Toxicology and Chemistry, v. 19, no. 6, p. 1624–1631.

Wigginton, A.J., 2005, Some effects of cadmium on select crayfish in the family Cambaridae: Lexington, Ky., University of Kentucky, Ph.D. Dissertation, 196 p.

Wilde, F.D., Radtke, D.B., Gibs, J., and Iwatsubo, R.T., eds., 2004, Processing of water samples (version 2.1): U.S. Geological Survey Techniques of Water-Resources Investigations, book 9, chap. A5, , accessed July 2006 at http://pubs.water.usgs.gov/twri9A5/.

Wilhelm, F.M., Parker, B.R., Schindler, D.W., and Donald, D.B., 1999, Seasonal food habits of bull trout from a small alpine lake in the Canadian Rocky Mountains: Transactions of the American Fisheries Society, v. 128, no. 6, p. 1176–1192.

Williams, K.A., Green, D.W.J., Pascoe, D., and Gower, D., 1986, The acute toxicity of cadmium to different larval stages of *Chironomus riparius* (Diptera: Chironomidae) and its ecological significance for pollution regulation: Oecologia, v. 70, p. 362–366.

Willson, M.F., 1997, Variation in salmonid life histories: patterns and perspectives: Portland, Oreg., U.S. Forest Service, Pacific Northwest Research Station, PNW-RP-498, 50 p.

Windward Environmental, 2002, Development of site-specific water quality criteria for the South Fork Coeur d'Alene River, Idaho: Derivation of acute and chronic criteria for lead and zinc: Seattle, Wash., Prepared for the Idaho Department of Environmental Quality. Windward Environmental. 32 p., accessed March 2005 at http://www.deq.state.id.us/water/data_reports/surface_water/monitoring/site_specific_criteria.cfm.

Winner, R.W., 1984, The toxicity and bioaccumulation of cadmium and copper as affected by humic acid: Aquatic Toxicology, v. 5, p. 267–274.

Winner, R.W., 1986, Interactive effects of water hardness and humic acid on the chronic toxicity of cadmium to *Daphnia pulex*: Aquatic Toxicology, v. 8, p. 281–293.

Winner, R.W., 1988, Evaluation of the relative sensitivities of 7-d *Daphnia magna* and *Ceriodaphnia dubia* toxicity tests for cadmium and sodium pentachlorophenate: Environmental Toxicology and Chemistry, v. 7, no. 2, p. 153–159.

Wood, C.M., Adams, W.J., Ankley, G.T., DiBona, D.R., Luoma, S.N., Playle, R.C., Stubblefield, W.A., Bergman, H.L., Erickson, R.J., Mattice, J.S., and Schlekat, C.E., 1997, Environmental toxicology of metals, *in* Bergman, H.L., and Dorward-King, E.J., eds., Reassessment of metals criteria for aquatic life protection: priorities for research and implementation SETAC Pellston Workshop on Reassessment of Metals Criteria for Aquatic Life Protection: Pensacola, Fla., Society of Environmental Toxicology and Chemistry (SETAC), p. 31–56.

Woodward, D.F., Brumbaugh, W.G., Lelonay, A.J., Little, E.E., and Smith, C.E., 1994, Effects on rainbow trout fry of a metals-contaminated diet of benthic invertebrates from the Clark Fork River, Montana: Transactions of the American Fisheries Society, v. 123, no. 1, p. 51–62.

Zar, J.H., 1984, Biostatistical analysis (2nd ed.): Englewood Cliffs, N.J., Prentice Hall, 717 p.

Zaroban, D.W., Mulvey, M.P., Maret, T.R., Hughes, R.M., and Merritt, G.D., 1999, A classification of species attributes for Pacific Northwest freshwater fishes: Northwest Science, v. 78, p. 81–93.

Zhao, Y., and Newman, M.C., 2004, Shortcomings of the laboratory-derived median lethal concentration for predicting mortality in field populations: exposure duration and latent mortality: Environmental Toxicology and Chemistry, v. 23, no. 9, p. 2147–2153.

This page left intentionally blank

Table 15. Acute toxicity of cadmium to freshwater animals.

[**Method:** D, “dissolved” test concentrations measured after being filtered through a 0.45-µm filter; F, flow-through test exposures; M, test concentrations measured during exposures; R, renewed test exposures; S, static test exposures; T, “total” metal concentrations determined from unfiltered samples; U, unmeasured test concentrations. **LC_{50}:** Concentration killing 50 percent of test organisms. The term “LC50” as used here is equivalent to the term “EC_{50}” as used by some authors to report results of acute tests where the only endpoints measured were mortality or immobilization, which was considered effective mortality. *Italicized LC_{50}* values may have been obtained with resistant life stages and were not used in SMAV calculations. **SMAV:** Species mean acute value. **GMAV:** Genus mean acute value. **Data different than EPA: “**No” indicates that the test descriptions and values listed here for a test are identical to those listed in table 1a of Environmental Protection Agency (2001). “Yes” indicates that the values listed here either were not included in table 1a of U.S. Environmental Protection Agency (2001) or some aspect of the test descriptions or summary results in this table differs from those in table 1a of U.S. Environmental Protection Agency (2001). **Abbreviations:** EPA, U.S. Environmental Protection Agency; ACR, acute to chronic ratio; $CaCO_3$, calcium carbonate; Cd, cadmium; DOC, dissolved organic carbon; LOEC, lowest-observed-effect concentration; $MgSO_4$, magnesium sulfate; MN, Minnesota; MO, Missouri; mo, month; yr, year; ºC, degrees Celsius; g, gram; hrs, hours; mg, milligram; mg/L, milligram per liter; mm, millimeter; ng, not given in source; µg/L, microgram per liter; µm, micrometer. **Symbols:** <, less than; ≤, less than or equal to; –, missing or inapplicable values; ~, approximately; ±, plus or minus]

Species	Common names	Size or age at test initiation	Method	Hardness (mg/L as $CaCO_3$)	LC_{50} (µg/L)	Hardness adjusted LC50 (µg/L)	SMAV (µg/L)	GMAV (µg/L)	Reference	Code/ Notes	Data different than EPA (2001)?
Actinonaias pectorosa	Pheasant shell (mussel)	Juvenile	S,M,T	82	46	30.41	–	–	U.S. Environmental Protection Agency, 2001, citing Keller, unpublished	1	No
		Juvenile	S,M,T	84	69	44.7	36.87	36.87		1	No
Aelosoma headleyi	Polychaete worm	Non-reproductive	S,M,T	62	1,200	1,002	–	–	Niederlehner and others, 1984	–	Yes
		Non-reproductive	S,M,T	168	4,980	1,806	1,345	1,345		–	Yes
Ambystoma gracile	Northwestern salamander	3-mo old larva	F,M,T	45	468.4	511.57	511.57	511.57	Nebeker and others, 1995	–	No
Aplexa hypnorum	Physid snail	Adult	F,M,T	45	93	101.01	101.01	101.01	Holcombe and others, 1984	2	No
		Adult	F,M,T	45	93	101.01	–	–	Phipps and Holcombe, 1985	2	Yes
Arctopsyche sp.	Caddisfly	Field collected	R,M,D	30	467	716.06	716.06	716.06	Windward, 2002	–	Yes
Asellus sp.	Isopod	ng	F,M,T	220	2,129	616.26	616.3	616.26	U.S. Environmental Protection Agency, 2001	3	No
Baetis tricaudatus	Mayfly	Field collected	R,M,T	156	1,611	621.74	–	–	Irving and others, 2003	4	Yes
		Field collected	R,M,D	21	73	150.86	306.3	306.26	Windward, 2002	5	Yes
Branchiura sowerbyi	Tubificid worm	Probably adult, field collected	R,M,T	5.3	240	1,569.61	1,570	1,569.61	Chapman and others, 1982	–	No
Carassius auratus	Goldfish	8.8 g	F,M,T	44	748	832.45	832.5	832.5	U.S. Environmental Protection Agency, 2001	–	No
Catostomus commersoni	White sucker	ng	F,M,T	18	1,110	2,609.69	2,610	2,610	Environmental Protection Agency, 2001	–	No
Ceriodaphnia dubia	Cladoceran, water flea	≤24hrs	S,M,D	17	63.1	155.62	–	–	Suedel and others, 1997	6	Yes
		<24 hrs	R,M,T	100	27.3	15.29	–	–	Spehar and Fiandt, 1986	7	Yes
		<24 hrs	S,M,T	290	120	27.57	–	–	Schubauer-Berigan and others, 1993	8	Yes
		<24 hrs	R,M,T	80	54.5	36.78	–	–	Diamond and others, 1997	–	No
		<24 hrs	R,M,T	170	38.3	13.76	–	–	Brooks and others, 2004	–	Yes
		<24 hr	S,M,T	40	31	37.94	32.86	32.86	Shaw and others, 2006	4	Yes
Chironomus riparius	Midge	2nd instar	R,M,T	105	13,000	6,988	6,988	–	Williams and others, 1986	–	Yes
Chironomus tentans	Midge	2nd instar, 10–12 days	S,M,T	17	8,000	19,730	19,730	11,742	Suedel and others, 1997	–	Yes

Table 15. Acute toxicity of cadmium to freshwater animals.—Continued

Species	Common names	Size or age at test initiation	Method	Hardness (mg/L as $CaCO_3$)	LC50 (µg/L)	Hardness adjusted LC50 (µg/L)	SMAV (µg/L)	GMAV (µg/L)	Reference	Code/ Notes	Data different than EPA (2001)?
Coregonus clupeaformis	Lake whitefish	Yearlings, 140 mm, 22 g	F,M,T	81	530	354	354	354	McNicol, 1997	34	Yes
Cottus bairdi	Mottled sculpin	MO strain, newly hatched	F,M,T	101	2.9	1.62	–	–	Besser and others, 2006	–	Yes
		MO strain, swimup	F,M,T	101	5.2	2.91	–	–		–	Yes
		MN strain, swimup	F,M,T	101	3.6	2.02	–	–		–	Yes
		MN strain, 0.03 g, 3-week swimup	F,M,T	101	8	4.48	2.61	–		9	Yes
		MN strain, 0.1 g, 7-week	F,M,T	101	16.5	*9.24*	–	–		–	Yes
		MN strain, 0.26 g, YOY	F,M,T	101	*12.9*	23	–	–		–	Yes
		MN strain, yearling	F,M,T	101	176	*98.54*	–	–		–	Yes
Cottus confusus	Shorthead sculpin	Field collected, 30–60 mm TL, ~age 1-2 yr.	R,M,D	21	1.29	2.67	2.67	2.67	Windward, 2002	–	Yes
Cyprinella lutrensis	Red shiner	ng	S,M,T	86	6,620	4,225	4,225	4,225	U.S. Environmental Protection Agency, 2001	–	No
Daphnia ambigua	Cladoceran, water flea	<24 hrs	S,M,T	40	10	12.2	12.2	–	Shaw and others, 2006	4	Yes

Table 15. Acute toxicity of cadmium to freshwater animals.—Continued

Species	Common names	Size or age at test initiation	Method	Hardness (mg/L as $CaCO_3$)	LC50 (µg/L)	Hardness adjusted LC50 (µg/L)	SMAV (µg/L)	GMAV (µg/L)	Reference	Code/ Notes	Data different than EPA (2001)?
Daphnia magna	Cladoceran, water flea	Genotype A	S,M,T	170	3.6	1.29	–	–	Baird and others, 1991	–	No
		Genotype A-1	S,M,T	170	9.0	3.23	–	–		–	No
		Genotype A-2	S,M,T	170	9.0	3.23	–	–		–	No
		Genotype B	S,M,T	170	4.5	1.62	–	–		–	No
		Genotype E	S,M,T	170	27.1	9.73	–	–		–	No
		Genotype S-1	S,M,T	170	115.9	41.6	–	–		–	No
		<24 hrs, clone F	S,M,T	170	3.3	1.2	–	–	Barata and Baird, 2000	–	Yes
		<24 hrs, genotype A	S,M,T	170	3.6	1.29	–	–	Barata and others, 1998	–	No
		<24 hrs, genotype A-1	S,M,T	170	9	3.23	–	–		–	No
		<24 hrs, genotype A-2	S,M,T	170	9	3.23	–	–		–	No
		<24 hrs, genotype B	S,M,T	170	4.5	1.62	–	–		–	No
		<24 hrs, genotype E	S,M,T	170	27.1	9.73	–	–		–	No
		<24 hrs, genotype S-1	S,M,T	170	115.9	41.63	–	–		–	No
		<24 hrs, clone S-1	S,M,T	170	129.4	46.47	–	–		–	No
		Clone S-1	S,M,T	46	112	119.9	–	–		–	No

Table 15. Acute toxicity of cadmium to freshwater animals.—Continued

Species	Common names	Size or age at test initiation	Method	Hardness (mg/L as $CaCO_3$)	LC50 (µg/L)	Hardness adjusted LC50 (µg/L)	SMAV (µg/L)	GMAV (µg/L)	Reference	Code/ Notes	Data different than EPA (2001)?
Daphnia magna—Continued	Cladoceran, water flea—Continued	Clone S-1	S,M,T	91	106	64.40	–	–	Barata and others, 1998	–	No
		Clone F	S,M,T	179	233	80.15	–	–		–	No
		<24 hrs	S,M,T	179	23.6	8.12	–	–		–	No
		<24 hrs	R,M,T	105	30	16.13	–	–	Canton and Sloof, 1982	–	No
		<24 hrs	R,M,T	209	30	9.06	–	–		–	No
		<24 hrs	S,M,T	51	9.9	9.74	–	–	Chapman and others, 1980	h	No
		<24 hrs	S,M,T	104	33	17.88	–	–		h	No
		<24 hrs	S,M,T	105	34	18.28	–	–		h	No
		<24 hrs	S,M,T	197	63	20	–	–		h	No
		<24 hrs	S,M,T	209	49	14.81	–	–		h	No
		<24 hrs	F,M,T	130	58	26.07	–	–	Attar and Maly, 1982	–	No
		<24 hrs	S,M,T	170	37	13.29	–	–	Lewis and Weber, 1985	–	Yes
		<24 hrs	S,M,T	170	6.1	2.19	–	–		–	Yes
		<24 hrs	S,M,T	170	43	15.44	–	–		–	Yes
		<24 hrs	S,M,T	170	31	11.13	–	–		–	Yes
		<24 hrs	S,M,T	170	18	6.46	–	–		–	Yes
		<24 hrs	S,M,T	170	12	4.31	–	–		–	Yes
		<24 hrs	S,M,T	170	24	8.62	–	–		–	Yes
		<4 hrs	S,M,T	76	59	41.56	–	–	Nebeker and others, 1986a	–	Yes
		<4 hrs	S,M,T	74	84	60.51	–	–		–	Yes
		<4 hrs	S,M,T	41	99	116.88	–	–		–	Yes
		<4 hrs	S,M,T	38	164	206.34	–	–		–	Yes
		<4 hrs	S,M,T	76	71	50.01	–	–		–	Yes
		<4 hrs	S,M,T	74	178	128.22	–	–		–	Yes
		<4-hrs	S,M,T	74	116	83.56	–	–		–	Yes
		<4-hrs	S,M,T	71	101	75.32	–	–		–	Yes
		1 day	S,M,T	71	4	2.98	–	–		–	Yes
		1 day	S,M,T	41	8	9.45	–	–		–	Yes
		1 day	S,M,T	38	16	20.13	–	–		–	Yes
		1 day	S,M,T	74	146	105.17	–	–		–	Yes
		2 days	S,M,T	76	55	38.74	–	–		–	Yes
		2 days	S,M,T	74	306	220.42	–	–		–	Yes
		2 days	S,M,T	41	98	115.70	–	–		–	Yes
		2 days	S,M,T	38	307	386.25	–	–		–	Yes
		2 days	S,M,T	76	37	26.06	–	–		–	Yes
		2 days	S,M,T	74	94	67.71	–	–		–	Yes
		2 days	S,M,T	74	277	199.53	–	–		–	Yes
		2 days	S,M,T	71	135	100.67	–	–		–	Yes

Table 15. Acute toxicity of cadmium to freshwater animals.—Continued

Species	Common names	Size or age at test initiation	Method	Hardness (mg/L as $CaCO_3$)	LC50 (µg/L)	Hardness adjusted LC50 (µg/L)	SMAV (µg/L)	GMAV (µg/L)	Reference	Code/ Notes	Data different than EPA (2001)?
Daphnia magna—Continued	Cladoceran, water flea—Continued	5 days	S,M,T	76	17	11.98	–	–	Nebeker and others, 1986a	–	Yes
		5 days	S,M,T	74	40	28.81	–	–		–	Yes
		5 days	S,M,T	41	30	35.42	–	–		–	Yes
		5 days	S,M,T	38	131	164.82	–	–		–	Yes
		5 days	S,M,T	76	25	17.61	–	–		–	Yes
		5 days	S,M,T	74	36	25.93	–	–		–	Yes
		5 days	S,M,T	71	18	13.42	–	–		–	Yes
		5 days	S,M,T	34	36	49.71	–	–	Nebeker and others, 1986b	–	Yes
		5 days	S,M,T	34	33	45.57	–	–		–	Yes
		5 days	S,M,T	34	24	33.14	–	–		–	Yes
		5 days	S,M,T	34	40	55.23	–	–		–	Yes
		<24 hrs	S,M,T	40	101	121.9	–	–	Shaw and others, 2006	4	Yes
		<24 hrs, clone F	S,M,T	170	24.5	8.80	–	–	Stuhlbacher and others, 1992	–	No
		<24 hrs, clone S-1	S,M,T	170	129.4	46.47	–	–		–	No
		6 days, clone F	S,M,T	170	49.1	17.63	–	–		–	No
		<24 hrs old, clone F	S,M,T	170	25.4	9.12	–	–	Stuhlbacher and others, 1993	–	No
		3 days, clone S-1	S,M,T	170	228.8	82.17	–	–		–	No
		6 days, clone S-1	S,M,T	170	250	89.79	–	–		–	No
		≤24 hrs	S,M,T	17	26.4	65.11	–	–	Suedel and others, 1997	–	Yes
		≤24 hrs	R,M,T	170	26	9.34	–	–	Ward and Robinson, 2005	–	Yes
		≤24 hrs	R,M,T	170	34	12.21	–	–		–	Yes
		≤24 hrs	R,M,T	170	39	14.01	–	–		–	Yes
		≤24 hrs	R,M,T	170	48	17.24	–	–		–	Yes
		≤24 hrs	R,M,T	170	55	19.75	–	–		–	Yes
		≤24 hrs	R,M,T	170	63	22.63	–	–		–	Yes
		≤24 hrs	R,M,T	170	100	35.92	–	–		–	Yes
		≤24 hrs	R,M,T	170	120	43.10	22.29	–		gt	Yes
Daphnia pulex	Cladoceran, water flea	<24 hrs	S,M,T	53	70.1	66.76	–	–	Stackhouse and Benson, 1988	–	No
		≤24 hrs	R,M,T	58	115	101.57	–	–	Ingersoll and Winner, 1982	10	Yes
		≤24 hrs	R,M,T	85	130	83.39	–	–	Lewis and Weber, 1985	37	Yes
		≤24 hrs	R,M,T	85	120	76.98	–	–		37	Yes
		≤24 hrs	R,M,T	85	170	109.05	–	–		37	Yes
		≤24 hrs	R,M,T	85	130	83.39	–	–		37	Yes

Table 15. Acute toxicity of cadmium to freshwater animals.—Continued

Species	Common names	Size or age at test initiation	Method	Hardness (mg/L as $CaCO_3$)	LC50 (µg/L)	Hardness adjusted LC50 (µg/L)	SMAV (µg/L)	GMAV (µg/L)	Reference	Code/ Notes	Data different than EPA (2001)?
Daphnia pulex—Continued	Cladoceran, water flea—Continued	≤24 hrs	R,M,T	85	190	121.88	–	–	Lewis and Weber, 1985	37	Yes
		≤24 hrs	R,M,T	85	160	102.63	–	–		37	Yes
		≤24 hrs	R,M,T	85	150	96.22	–	–		–	Yes
		≤24 hrs	R,M,T	85	130	83.39	–	–		–	Yes
		≤24 hrs	R,M,T	85	150	96.22	–	–		–	Yes
		≤24 hrs	R,M,T	85	100	64.15	–	–		–	Yes
		≤24 hrs	R,M,T	85	180	115.46	–	–		–	Yes
		≤24 hrs	R,M,T	85	130	83.39	–	–		–	Yes
		≤24 hrs	R,M,T	40	45	54.19	87.19	28.72	Shaw and others, 2006	–	Yes
Dendrocoelum lacteum	Planarian	ng	R,M,T	87	24,702	15,540	15,540	15,540	U.S. Environmental Protection Agency, 2001	–	No
Diporeia spp.	Amphipod	Juveniles, 3.3 ± 1.3 mg	S,M,T	160	60	22.67	22.67	22.67	Gossiaux and others, 1992	4, 36	Yes
Etheostoma fonticola	Fountain darter	4–6 days	R,M,T	270	15.7	3.83	–	–	Castillo and Longley, 2001	–	Yes
		4–6 days	R,M,T	261	14.23	3.57	–	–		–	Yes
		4–6 days	R,M,T	285	13.27	3.09	–	–		–	Yes
		4–6 days	R,M,T	270	11.77	2.87	3.32	3.32		–	Yes
Gambusia affinis	Mosquitofish	ng	F,M,T	11	900	3,195.02	–	–	U.S. Environmental Protection Agency, 2001	–	No
		ng	S,M,T	11	2,200	7,810.05	4,995.32	4,995.32		–	No
Gammarus pseudolimnaeus	Amphipod	ng	S,M,T	44	68.30	76.74	76.74	76.74	Spehar and Carlson, 1984	–	No
Gasterosteus aculeatus	Threespine stickleback	ng	R,M,T	107	23,300	12,327	12327	12327	U.S. Environmental Protection Agency, 2001	–	No
Glossiphonia complanata	Leech	ng	R,M,T	123	480	226.01	226.01	226.01	U.S. Environmental Protection Agency, 2001	12	No
Gyraulus sp.	Snail	larvae	R,M,D	21	73	150.86	150.86	150.86	Windward, 2002	gt	Yes
Hyalella azteca	Amphipod	Larger juveniles and young adults	S,M,T	34	8	11.05	–	–	Nebeker and others, 1986b	–	Yes
		7–10 days	S,M,T	48	3.8	3.93	–	–	Jackson and others, 2000	h, 32	Yes
		7–10 days	S,M,T	118	12.1	5.90	–	–		h, 32	Yes
		2–3 weeks	S,M,D	17	2.8	6.91	–	–	Suedel and others, 1997	h, 6, 13	Yes
		0–2 days	S,M,T	90	13	7.95	–	–	Collyard and others, 1994	h,14	Yes
		2–4 days	S,M,T	90	7.5	4.59	–	–		h,14	Yes
		4–6 days	S,M,T	90	9.5	5.81	–	–		h,14	Yes
		10–12 days	S,M,T	90	7	4.28	–	–		h,14	Yes
		16–18 days	S,M,T	90	11.5	7.03	–	–		h,14	Yes
		24–26 days	S,M,T	90	14	8.56	–	–		h,14	Yes
		7–14 days	S,M,T	290	5	1.15	5.01	5.01	Schubauer-Berigan and others, 1993	8	Yes
Hydra oligactis	Hydra	Monecious species	S,M,T	210	320	96.3	–	–	Karntanut and Pascoe, 2002	15	Yes

Table 15. Acute toxicity of cadmium to freshwater animals.—Continued

Species	Common names	Size or age at test initiation	Method	Hardness (mg/L as $CaCO_3$)	LC50 (µg/L)	Hardness adjusted LC50 (µg/L)	SMAV (µg/L)	GMAV (µg/L)	Reference	Code/ Notes	Data different than EPA (2001)?
Hydra viridissima	Hydra	Monecious species	S,M,T	210	210	63.2	–	–	Karntanut and Pascoe, 2002	15	Yes
Hydra vulgaris	Hydra	Zurich strain, male clone	S,M,T	204	310	95.6	–	–	Karntanut and Pascoe, 2002	15	Yes
		Zurich strain, male clone	S,M,T	210	520	156.5	–	–		15	Yes
		Dioecious strain	S,M,T	210	160	48.15	–	–		15	Yes
Ictalurus punctatus	Channel catfish	7.4 g	F,M,T	44	4,480	4,985.77	4,985.77	4,985.77	Phipps and Holcombe, 1985	–	No
Jordanella floridae	American flagfish	4-5 weeks	F,M,T	44	2,500	2,782.24	2,782.24	2,782.24	Spehar, 1976	–	No
Lampsilis straminea claibornensis	Southern fatmucket (mussel)	Juvenile	S,M,T	40	38	45.80	45.80	45.80	U.S. Environmental Protection Agency, 2001	–	No
Lampsilis teres	Yellow sandshell (mussel)	Juvenile	S,M,T	40	11	13.26	–	–	U.S. Environmental	–	No
		Juvenile	S,M,T	40	33	39.77	22.96	22.96	Protection Agency, 2001	–	No
Lepomis cyanellus	Green sunfish	Juvenile	S,M,T	86	11,520	7,353.46	–	–	U.S. Environmental	h	No
		Life stage unknown	S,M,T	335	20,500	4,173.80	5,540.03	5,540.03	Protection Agency, 2001	h	No
Lepomis macrochirus	Bluegill	40 g, 13 cm adults	S,M,T	18	2,300	5,407.47	–	–	Bishop and McIntosh, 1981	h, 33	No
		40 g, 13 cm adults	S,M,T	18	2,300	5,407.47	–	–		h, 33	No
		Life stage unknown	F,M,T	207	21,100	6,426.94	–	–	U.S. Environmental Protection Agency, 2001	h	No
		1.0 g juvenile	F,M,T	44	6,470	7,146.11	5,896.91	5,709.35	Phipps and Holcombe, 1985	h	No
Limnodrilus hoffmeisteri	Tubificid worm	Probably adult, field collected	R,M,T	5.3	170	1,111.81	–	–	Chapman and others, 1982	h	No
		ng	S,M,T	152	2,400	946.60	1,025.88	1,025.88	Environmental Protection Agency, 2001	h	No
Lirceus alabamae	Isopod	ng	F,M,T	152	150	59.16	59.16	59.16		–	No
Lumbriculus variegatus	Aquatic earthworm	Mixed age adults	S,M,T	290	780	179.18	179.18	179.18	Schubauer-Berigan and others, 1993	–	No
Lymnaea stagnalis	Swamp lymnaea, pond snail	4-week juvenile	S,M,T	250	742	192.99	192.99	192.99	Coeurdassier and others, 2004	–	Yes
Oncorhynchus clarki lewisi	Westslope cutthroat trout	Swimup fry, Sandpoint strain	R,M,D	21	0.35	0.72	–	–	EVS Environment Consultants, 1996; Windward, 2002	16	Yes
		Field collected YOY, 20-50 mm TL	R,M,D	21	0.93	1.92	–	–	Windward, 2002	–	Yes

Table 15. Acute toxicity of cadmium to freshwater animals.—Continued

Species	Common names	Size or age at test initiation	Method	Hardness (mg/L as $CaCO_3$)	LC50 (µg/L)	Hardness adjusted LC50 (µg/L)	SMAV (µg/L)	GMAV (µg/L)	Reference	Code/ Notes	Data different than EPA (2001)?
Oncorhynchus clarki lewisi—Continued	Westslope cutthroat trout—Continued	Swimup fry, captive broodstock source, avg. wt.(g)= 0.2	R,M,D	32	1.41	2.05	–	–	Windward, 2002	–	Yes
		Swimup fry, captive broodstock source, avg. wt.(g)=0.18	R,M,D	31	1.18	1.78	1.50	–		–	Yes
Oncorhynchus mykiss	Rainbow trout	Swimup (0.19 g, ~3 weeks)	F,M,T	24	1.30	2.40	–	–	Chapman, 1978b	h	No
		Parr (7 g, 5-mo)	F,M,T	24	1.0	1.85	–	–		h	No
		Alevin, 1-day old	F,M,T	24	27	*49.90*	–	–		gt	No
		Smolt (68 g, 13 mo)	F,M,T	24	2.90	*5.36*	–	–		gt	No
		2.65 g	F,M,T	9	0.5	2.06	–	–	Cusimano and others, 1986	h, 26	No
		Fry, 1.0 g	F,M,T	29	2.09	3.30	–	–	Davies and Brinkman, 1994b	h	Yes
		45–50 mm, 36 g	F,M,T	45	2.64	2.88	–	–	Davies and others, 1993	h	Yes
		Juvenile, 50 mm	F,M,T	50	3.08	3.08	–	–		h, 18	Yes
		Not reported	F,M,T	31	1.75	2.61	–	–	U.S. Environmental Protection Agency, 2001, citing Davies, 1976	–	No
		Swimup fry, Kootenai strain	R,M,D	21	0.40	0.83	–	–	EVS Environment Consultants, 1996; Windward, 2002	h, 17	Yes
		1.0 g, pH 7.5	F,M,T	28	0.38	0.61	–	–	Hansen and others, 1999; Hansen and others, 2002a	h, 19	Yes
		1.1 g, pH 7.5	F,M,T	28	0.47	0.76	–	–		h, 19	Yes
		0.26 g, pH 7.5	F,M,T	29	0.71	1.12	–	–		h, 19	Yes
		0.3 g, pH 6.5	F,M,T	29	1.29	2.03	–	–		h, 20	Yes
		0.66 g, pH 7.5	F,M,T	30	0.51	0.79	–	–		h, 21	Yes
		0.28 g, pH 7.5	F,M,T	83	2.85	1.86	–	–		h, 19	Yes
		Juvenile, ~4.5 g	F,M,T	140	22	9.30	–	–	Hollis and others, 1999	h	Yes
		Juvenile, ~ 12 g	F,M,T	20	0.77	1.66	–	–	Hollis and others, 2000a	h, 22	Yes
		Juvenile, ~ 12 g	F,M,T	20	0.61	1.31	–	–		h, 23	Yes
		Juvenile, ~ 12 g	F,M,T	20	2.07	4.46	–	–		h, 24	Yes
		Juvenile, size not reported	F,M,T	25	2.35	4.19	–	–	Hollis and others, 2000b	h, 25	Yes
		Juvenile, size not reported	F,M,T	47	2.35	2.47	–	–		h	Yes
		Juvenile, size not reported	F,M,T	77	2.15	1.50	–	–		h	Yes

Table 15. Acute toxicity of cadmium to freshwater animals.—Continued

Species	Common names	Size or age at test initiation	Method	Hardness (mg/L as $CaCO_3$)	LC50 (µg/L)	Hardness adjusted LC50 (µg/L)	SMAV (µg/L)	GMAV (µg/L)	Reference	Code/ Notes	Data different than EPA (2001)?
Oncorhynchus mykiss—Continued	Rainbow trout—Continued	Juvenile, size not reported	F,M,T	120	1.15	0.55	–	–	Hollis and others, 2000b	h	Yes
		Juvenile, 8-12 g	F,M,T	120	19	9.13	–	–	Niyogi and others, 2004	h	Yes
		8.8 g	F,M,T	44	3	3.31	–	–	Phipps and Holcombe, 1985	h	No
		0.5 g	S,M,T	44	2.30	2.58	–	–	Spehar and Carlson, 1984	h	No
		Juvenile, 18 g	F,M,T	50	1.88	1.88	–	–	Stubblefield, 1990	h	Yes
		Swimup fry, Mt Lassen strain, avg. wt.(g)= 0.39	R,M,D	7.5	0.48	2.33	–	–	Windward, 2002	h	Yes
		Swimup fry, Mt Lassen strain, avg. wt.(g)= 0.39	R,M,D	14	0.97	2.89	–	–		h	Yes
		Swimup fry, Mt Lassen strain	R,M,D	21	0.84	1.74	–	–		h	Yes
		Swimup fry, Mt Lassen strain, avg. wt.(g)= 0.39	R,M,D	24	1.30	2.40	–	–		h	Yes
		Swimup fry, Mt Lassen strain	R,M,D	26	1.58	2.73	–	–		h	Yes
		Swimup fry, Mt Lassen strain	R,M,D	26	1.61	2.78	–	–		h	Yes
		Swimup fry, Mt Lassen strain, avg. wt.(g)= 0.36	R,M,D	29	0.83	1.33	–	–		h	Yes
		Swimup fry, Mt Lassen strain, avg. wt.(g)= 0.39	R,M,D	30	0.99	1.51	–	–		h	Yes
		Swimup fry, Mt Lassen strain, avg. wt.(g)= 0.31	R,M,D	32	0.89	1.29	–	–		h	Yes
		Swimup fry, 4–5 weeks	F,M,T	101	3.80	2.13	–	–	Besser and others, 2006	h	Yes
		Swimup fry, 4–5 weeks	F,M,T	101	5.30	2.97	–	–		h	Yes
		Swimup fry, 4–5 weeks	F,M,T	101	5.40	2.97	2.04	–		–	Yes

Table 15. Acute toxicity of cadmium to freshwater animals.—Continued

Species	Common names	Size or age at test initiation	Method	Hardness (mg/L as $CaCO_3$)	LC50 (µg/L)	Hardness adjusted LC50 (µg/L)	SMAV (µg/L)	GMAV (µg/L)	Reference	Code/ Notes	Data different than EPA (2001)?
Oncorhynchus tshawytscha	Chinook salmon	Alevin, 0.05 g, 1-day	F,M,T	24	27	*49.90*	–	–	Chapman, 1978b	gt	No
		Swimup (0.23 g, ~3 weeks)	F,M,T	24	1.8	3.33	–	–			No
		Parr (12 g, 5 months)	F,M,T	24	3.5	*6.47*	–	–		9	No
		Smolt (32 g, 18 months)	F,M,T	24	2.9	*5.36*	–	–		gt, 9	No
		~ 0.8 g	F,M,T	25	1.41	2.52	–	–	Chapman, 1982	27	No
		1.1 g	F,M,T	21	1.1	2.27	2.67	2.00	Finlayson and Verrue, 1982		No
Orconectes immunis	Calico crayfish	1.8 g	F,M,T	44	*10,200*	–	–	–	Phipps and Holcombe, 1985		No
Orconectes juvenilis	Crayfish	3rd to 5th instars, ~ 0.02 g	R,M,T	44	60	66.8	66.8	–	Wigginton, 2005	35	Yes
Orconectes placidus	Placid crayfish	3rd to 5th instars, ~ 0.02 g	R,M,T	55	37	34.2	34.2	47.8		–	Yes
Orconectes virilis	Virile crayfish	ng	F,M,T	26	*6,100*	*10,543*	–	–	U.S. Environmental Protection Agency, 2001	–	No
Perca flavescens	Yellow perch	8–12 g	F,M,T	120	8,140	3,912.74	3,912.74	3,913	Niyogi and others, 2004	–	Yes
Perlodidae	Stonefly	Field collected	R,M,D	30	5,130	7,865.94	7,866	7,866	EVS Environment Consultants, 1996	–	Yes
Physa gyrina	Physid snail	Immature	F,M,T	200	410	128.53	128.53	128.5	U.S. Environmental Protection Agency, 2001	28	No
Pimephales promelas	Fathead minnow	4–6 days	R,M,T	266	17	4.20	–	–	Castillo and Longley, 2001	h	Yes
		4–6 days	R,M,T	278	15.43	3.67	–	–		h	Yes
		2–4 days	S,M,D	17	4.8	11.84	–	–	Suedel and others, 1997	h	Yes
		Fry	S,M,T	40	21	25.31	–	–	U.S. Environmental Protection Agency, 2001, citing Spehar, 1982	h	No
		Fry	S,M,T	48	11.7	12.11	–	–		h	No
		Fry	S,M,T	39	19.3	23.76	–	–		h	No
		Fry	S,M,T	45	42.5	46.42	–	–		h	No
		Fry	S,M,T	47	54.2	57.08	–	–		h	No
		Fry	S,M,T	44	29	32.27	–	–		h	No
		<24 hr	S,M,T	290	65	14.93	–	–	Schubauer-Berigan and others, 1993	h, 8	Yes
		0.15 g, 30-days	S,M,T	45	13.2	14.42	16.49	16.49	Spehar and Fiandt, 1986	h	No
Poecilia reticulata	Guppy	ng	R,M,T	105	3,800	2,042.51	–	–	U.S. Environmental Protection Agency, 2001	–	No
		ng	R,M,T	209	11,100	3,353.90	2,617.33	2,617.33	U.S. Environmental Protection Agency, 2001	–	No
Potamopyrgus antipodarum	New Zealand mud snail	Juvenile, 3–4 mm	R,M,T	148	560	225.9	225.9	225.9	Jensen and Forbes, 2001	–	Yes

Table 15. Acute toxicity of cadmium to freshwater animals.—Continued

Species	Common names	Size or age at test initiation	Method	Hardness (mg/L as $CaCO_3$)	LC50 (µg/L)	Hardness adjusted LC50 (µg/L)	SMAV (µg/L)	GMAV (µg/L)	Reference	Code/ Notes	Data different than EPA (2001)?
Procambarus clarkii	Red swamp crayfish	3rd to 5th instars, ~ 0.02 g	R,M,T	42	624	722	722	722	Wigginton, 2005	–	Yes
		Juvenile	S,M,T	30	1,040	*1,594.65*	–	–	U.S. Environmental Protection Agency, 2001	–	No
Prosopium williamsoni	Mountain whitefish	Field collected, 209 g	F,M,T	50	8.29	8.29	8.29	8.29	Stubblefield, 1990	gt	Yes
Ptychocheilus oregonensis	Northern pikeminnow	Juvenile	F,M,T	25	1,092	1,950.34	1,950.34	1,950.34	U.S. Environmental Protection Agency, 2001	–	No
Quistadrilus multisetosus	Tubificid worm	Probably adult, field collected	R,M,T	5.3	320	2,092.81	2,092.81	2,092.81	Chapman and others, 1982	–	No
Rhithrogena sp.	Mayfly	larvae	R,M,D	21	50	103.33	103.33	103.33	Windward, 2002	gt	Yes
Rhyacodrilus montana	Tubificid worm	ng	S,M,T	5.3	630	4,120.22	4,120.22	4,120.22	Chapman and others, 1982	–	No
Salmo trutta	Brown trout	Fingerling, 22 g	F,M,T	50	2.85	2.85	–	–	Stubblefield, 1990	h	Yes
		0.48 g	F,M,T	38	2.37	3.01	–	–	Davies and Brinkman, 1994c	h	Yes
		34 days post swimup	F,M,T	29	1.23	1.94	–	–	Brinkman and Hansen, 2004a	h	Yes
		34 days post swimup	F,M,T	68	3.9	3.02	–	–		h	Yes
		34 days post swimup	F,M,T	151	10.1	4.01	–	–		h	Yes
		1-2 g	S,M,T	44	1.4	1.57	2.61	2.61	Spehar and Carlson, 1984	h	No
Salvelinus confluentus	Bull trout	0.07 g, pH 7.5	F,M,T	29	.91	1.43	–	–	Hansen and others, 1999; Hansen and others, 2002a	h, 19	Yes
		0.07 g, pH 7.5	F,M,T	83	6.06	3.96	–	–		h, 19	Yes
		0.08 g, pH 6.5	F,M,T	29	2.89	4.54	–	–		h, 20	Yes
		0.2 g, pH 7.5	F,M,T	30	1.0	1.54	–	–		h, 21	Yes
		0.22 g, pH 7.5	F,M,T	28	.90	1.44	–	–		h, 19	Yes
		0.22 g, pH 7.5	F,M,T	28	.99	1.61	2.13	2.13		h, 19	Yes
Simocephalus serrulatus	Cladoceran	ng	S,M,T	11	7	24.85	–	–	U.S. Environmental Protection Agency, 2001	–	No
		ng	S,M,T	44	24.5	27.53	26.15	26.15	Spehar and Carlson, 1984	–	No
Spirosperma sp. (“*ferox*”)	Tubificid worm	Probably adult, field collected	R,M,T	5.3	350	2,943.02	2,943.02	2,943.02	Chapman and others, 1982	29	No
Spirosperma nikolskyi	Tubificid worm	Probably adult, field collected	R,M,T	5.3	450	2,943.02	2,943.02	2,943.02	Chapman and others, 1982	–	No
Stylodrilus heringlianus	Tubificid worm	Probably adult, field collected	R,M,T	5.3	550	3,597.02	3,597.02	3,597.02	Chapman and others, 1982	–	No
Tubifex tubifex	Tubificid worm	ng	S,M,T	128	3,200	1,457.31	–	–	U.S. Environmental Protection Agency, 2001	h	No
		ng	S,M,T	128	1,700	7,74.20	–	–		h	No
		Probably adult, field collected	R,M,T	5.3	320	2,092.81	1,331.61	1,331.61	Chapman and others, 1982	h	No

Table 15. Acute toxicity of cadmium to freshwater animals.—Continued

Species	Common names	Size or age at test initiation	Method	Hardness (mg/L as $CaCO_3$)	LC50 (µg/L)	Hardness adjusted LC50 (µg/L)	SMAV (µg/L)	GMAV (µg/L)	Reference	Code/ Notes	Data different than EPA (2001)?
Utterbackia imbecilis	Paper pondshell (mussel)	ng	S,M,T	90	114.7	70.14	–	–	U.S. Environmental Protection Agency, 2001, citing Keller, unpublished	–	No
		ng	S,M,T	90	111.8	68.37	–	–		–	No
		Juvenile	S,M,T	92	81.9	49.17	–	–		–	No
		Juvenile	S,M,T	86	93	59.07	–	–		–	No
		Juvenile	S,M,T	44	9	10.02	–	–	Keller and Zam, 1991	–	Yes
		Juvenile	S,M,T	90	107	65.43	46.49	46.49		–	No
Varichaetadrilus pacificus	Tubificid worm	Probably adult, field collected	R,M,T	5.3	380	2,485.21	2,485.21	2,485.21	Chapman and others, 1982	30	No
Villosa vibex	Southern rainbow (mussel)	Juvenile	S,M,T	40	30	36.16	–	–	U.S. Environmental Protection Agency, 2001, citing Keller, unpublished	h, 31	No
		Juvenile	S,M,T	186	125	41.64	38.80	38.80		h, 32	No

Code/Notes: **Alpha notes:** gt, "greater than" value; h, data used in hardness-toxicity regressions; lt, "less than" value.

Numbered notes:

1. Listed as "Actinonaia pecorosa" in source.
2. Apparently identical data that were described in more than one publication are assumed to reflect the results of a single test and are only used once here.
3. Listed as "Asellus bicrenata" in source. No record found in ITIS (2002).
4. Hardness value obtained by written communication from author.
5. Originally reported as >73 µg/L, but because mortality at 73 µg/L (the highest concentration tested) was 45 percent greater than control mortality, this concentration is considered a reasonable estimate of the 50 percent lethality level.
6. Organisms were fed but data considered usable because in comparison to other values for this species, results did not appear to be insensitive suggesting feeding did not appear to be an important factor in reducing toxicity. Because aeration was not needed to maintain dissolved oxygen levels, organic loading from feeding was not very high (B. Suedell, oral commun.).
7. Total organic carbon concentration in the test water was 7.1 mg/L. Stephan and others (1985) advise against using the results of tests in which organic carbon exceeded 5 mg/L in the test water unless a relationship is developed or unless data show that organic carbon does not affect toxicity. In comparison to other values for this genus in waters with lower DOC, the responses from this test were about as sensitive. Further, in contrast to copper, DOC has been shown to play a minor role in mitigating cadmium toxicity to fish because the binding affinity of cadmium to the gill surface is higher than the binding affinity of cadmium to dissolved organic matter (Playle, 2004; Playle and others, 1993). Thus, it appears that organic carbon in ambient waters at usual concentrations is not an important influence on cadmium toxicity.
8. Only values with natural pH of 8.3 used. Study manipulated pH while holding hardness and alkalinity constant. Only values were used from treatments in which pH was not manipulated in a manner conflicting with the assumption that hardness and pH are correlated.
9. Only newly hatched and swimup stage data were used in SMAV calculation because older fish may be more resistant.
10. Value for 72-hour LC50, 72-hour LC_{50} lower than 48 hours suggesting ACRs based on 48-hour LC_{50}s biased high.
12. Listed as "Glossiponia complanta" in source.
13. Organisms were fed; however, investigations of the influence of feeding on the responses of *Hyalella azteca* in sediment toxicity testing suggest that feeding was unlikely to have biased the results (Ankley and others, 1993).
14. Value estimated from authors' graph.
15. Sublethal effects, tenacle clubbing and slight contraction of body, noted at lower concentrations than LC_{50} (LOEC 110 µg/L). Zurich strain, male clone. "*Hydra vulgaris*" commonly referenced in literature, but no record for *H. vulgaris* found in ITIS (2002).

Table 15. Acute toxicity of cadmium to freshwater animals.—Continued

Code/Notes:

16. 86 percent dead at 0.5 µg/L (lowest concentration tested), LC_{50} calculated from original data using U.S. Environmental Protection Agency's (1991) trimmed Spearman-Karber method.
17. 72 percent dead at 0.5 µg/L (lowest concentration tested), LC50 calculated from original data using EPA's trimmed Spearman-Karber method.
18. Hardness adjusted from 45 mg/L (ambient) to 50 mg/L with $MgSO_4$. Because adjustment was minimal, the use of Mg-only hardness adjustment did not disqualify data.
19. Test conducted at pH 7.5. Value listed was calculated by Probit analysis using responses at 96 hours (Hansen and others, 1999, appendix D).
20. Test conducted at pH 6.5. Value listed was calculated by Probit analysis using responses at 96 hours (Hansen and others, 1999, appendix D).
21. Test conducted at pH 7.5 at 8ºC. Value listed was calculated by Probit analysis using responses at 96 hours (Hansen and others, 1999, appendix D).
22. Fish pre-exposed for 30 days at 0.07 µg/L, too low to induce Cd acclimation. Fish weights in acute tests were not given directly, estimated size of fish used in their acute tests were calculated from specific growth rates obtained during preceding chronic tests.
23. Fish pre-exposed for 30 days at 0.11 µg/L, too low to induce Cd acclimation. Fish weights in acute tests were not given directly, estimated size of fish used in their acute tests were calculated from specific growth rates obtained during the preceding chronic tests.
24. Control fish pre-exposed for 30 days at 0.02 µg/L, too low to induce Cd acclimation. Fish weights in acute tests were not given directly, estimated size of fish used in their acute tests were calculated from specific growth rates obtained during preceding chronic tests.
25. Data published in paper had a typographic error (reported as 2.53 µg/L on p. 2729). Lydia Hollis, Golder Associates, Calgary, written commun., 2006.
26. Author reported value as <0.5 µg/L in table; but here value is estimated as 0.5 µg/L (not less than). Basis is interpretation of authors' figure, as well as descriptions in text and figures of the value being near 0.4 µg/L, the nominal value or test exposure with partial kill. Detection limit was 0.5 µg/L.
27. Same cohort as used in chronic test.
28. Adults 3X less sensitive than immature, whereas with Aplexa, adult were more sensitive.
29. Listed as "Spirosperma ferox" in source. No record found in ITIS (2002).
30. "Varichaeta pacifica" in source, changed per ITIS (2002).
31. Listed as "Vilosa vibex" in source.
32. Jackson and others (2000) conducted tests in addition to the two values listed in this table. However, the additional values were not used because they were from tests that varied the ionic composition of dilution waters in proportions that seemed unlikely to occur in natural waters.
33. Bishop and McIntosh (1981) reported the results obtained with these two acute tests (with identical results) under a heading that seemed to indicate they were obtained using fish collected from a cadmium-contaminated lake (their table 2). However, following a careful reading of their text, conclude that row heading was missing and in fact the data were obtained from hatchery control fish; thus the data are considered valid and usable.
34. Young-of-year (YOY) lake whitefish were less sensitive than yearlings, with 100 percent survival of YOY at 529 µg/L Cd after 96 hours.
35. No species record in ITIS (2002).
36. Tested at 15ºC, tests at colder temperature were less sensitive. Additional information on test waters that was not presented in the published paper included: alkalinity 250–270 mg/L, hardness, 150–170 mg/L, and pH 8.1–8.4 (Duane Gossiaux, National Oceanic and Atmospheric Administration, written commun., 2005).
37. Animals were fed, however LC50s were similar between fed and unfed animals, so data were not rejected from tests with fed animals.

Table 16. Chronic toxicity of cadmium to freshwater animals.

[Each row describes one test. **Test type:** ELS, early life stage test; LC, life cycle test; JGS, juvenile growth and survival test; PLC, partial life cycle test (survival, growth, and reproduction less than one complete life cycle). **Method:** D, dissolved test concentrations that were measured after being filtered through a 0.45-µm filter; F, flow-through test exposures; M, test concentrations were measured during exposures; R, renewed test exposures; S, static test exposures; T, total metal concentrations determined from unfiltered samples; U, unmeasured test concentrations. **NOEC:** Highest no observed effect concentrations. **LOEC:** Lowest observed effect concentration. **EC_{10}, EC_{20}:** Concentration causing 10 or 20 percent effect in the most sensitive endpoint. **Statistic for chronic value:** MATC, maximum acceptable threshold concentration. Calculated as the geometric mean of the NOEC and LOEC. **SMCV:** species mean chronic value. **GMCV:** genus mean chronic value. **Data different than EPA:** "No" indicates that the test descriptions and values listed here for a test are identical to those listed in table 2a of U.S. Environmental Protection Agency (2001). "Yes" indicates either that the values listed here either were not included in table 2a of U.S. Environmental Protection Agency (2001), or if they were, some aspect of the test descriptions or summary results in this table differs from those in table 2a of U.S. Environmental Protection Agency (2001). **Abbreviations:** Cd, cadmium; °C, degrees Celsius; DOC, dissolved organic carbon; EPA, U.S. Environmental Protection Agency; g, gram; ITIS, Integrated Taxonomic Information System; nc, not calculable; µg/L, microgram per liter; µm, micrometer; USGS, U.S. Geological Survey. **Symbols**: <, less than; >, greater than; –, missing or inapplicable values; ~, approximately]

Test No.	Species	Common name	Test type	Duration (days)	Method	Lowest endpoint	Hardness (mg/L as $CaCO_3$)	NOEC (µg/L)	LOEC (µg/L)	EC_{10} (µg/L)	EC_{20} (µg/L)	Statistic used for chronic value	Chronic value (CV, µg/L)	Hardness adjusted CV (µg/L)	SMCV	GMCV	Code/ notes	Reference	Data different than EPA (2001)?
1	*Aelosoma headleyi*	Polychaete worm	LC	14	R,M,T	Population	62	17.2	36.9	–	–	MATC	25.2	22.0	–	–	–	Niederlehner and others, 1984	Yes
2			LC	14	R,M,T	Population	168	32	50.2	–	–	MATC	40.1	18.8	–	–	–		Yes
3			LC	14	R,M,T	Population	189	53.6	92	–	–	MATC	70.2	30.6	23.3	23.3			Yes
4	*Ambystoma gracile*	Northwestern salamander	JGS	24	F,M,T	Growth	45	12.8	44.6	–	–	NOEC	23.89	25.5	25.5	25.5	1	Nebeker and others, 1994, 1995	Yes
5	*Aplexa hypnorum*	Snail	LC	26	F,M,T	Reproduction	45.3	2.41	4.41	–	–	MATC	3.26	3.47	–	–	2	Holcombe and others, 1984	Yes
6			LC	26	F,M,T	Reproduction	45.3	1.51	2.5	–	–	MATC	1.94	2.07	2.68	2.68	3		Yes
7	*Bufo americanus*	American toad	PLC	60	R,M,T	Growth	51	<1	5	–	–	MATC	2.24	2.21	2.21	2.21	4	James and Little, 2003	Yes
8	*Catostomus commersoni*	White sucker	ELS	60	F,M,T	Mortality	44	4.2	12	–	–	MATC	7.10	7.68	7.68	7.68	5	Eaton, 1978	No
9	*Ceriodaphnia dubia*	Water flea	LC	7	S,M,D	Reproduction	17	1.0	4.0	–	–	MATC	2.00	3.90	–	–	6	Suedel and others, 1997	Yes
10			LC	7	R,M,T	–	20	10	19	–	–	MATC	13.78	24.3	–	–	7	Jop and others, 1995	No
11			LC	7	R,M,T	–	100	–	–	–	–	MATC	2.20	1.43	–	–	8	Spehar and Fiandt, 1986	Yes
12			LC	10	R,M,T	Mortality	270	10.7	21.9	nc	nc	MATC	15.31	5.38	–	–	9	Castillo and Longley, 2001	Yes
13			LC	8	R,M,T	Reproduction	261	1.6	3	1.8	2.5	MATC	2.19	.79	–	–	10		Yes
14			LC	8	R,M,T	Reproduction	285	5.7	8.5	5.3	6.4	MATC	6.96	2.37	–	–	11		Yes
15			LC	8	R,M,T	Reproduction	271	9.6	12.2	4.5	6.1	MATC	10.82	3.80	–	–	12		Yes
16			LC	10	R,M,T	Mortality	292	2.8	4	1.7	2.2	MATC	3.35	1.12	–	–	13		Yes
17			LC	7	R,M,T	Reproduction	170	1.1	3.4	–	–	MATC	1.93	.91	2.04	–	–	Brooks and others, 2004	Yes
18	*Ceriodaphnia reticulata*	Water flea	LC	9	R,M,T	–	44	3.6	7.5	–	–	MATC	5.20	5.62	5.62	3.39	–	Spehar and Carlson, 1984	Yes
19	*Chironomus tentans*	Midge	LC	60	F,M,T	Hatching	280	5.8	16.4	–	4.0	MATC	9.75	3.35	3.35	–	–	Ingersoll and Kemble, 2001	No

Table 16. Chronic toxicity of cadmium to freshwater animals.—Continued

Test No.	Species	Common name	Test type	Duration (days)	Method	Lowest endpoint	Hardness (mg/L as $CaCO_3$)	NOEC (µg/L)	LOEC (µg/L)	EC_{10} (µg/L)	EC_{20} (µg/L)	Statistic used for chronic value	Chronic value (CV, µg/L)	Hardness adjusted CV (µg/L)	SMCV	GMCV	Code/ notes	Reference	Data different than EPA (2001)?
20	*Cottus bairdi*	Mottled sculpin	JGS	28	F,M,D	Mortality	100	1.4	2.6	–	–	MATC	1.91	1.24	–	–	14	Besser and others, 2006	Yes
21			JGS	28	F,M,D	Mortality	100	1.25	2.74	–	–	MATC	1.85	1.20	1.22	1.22	15		Yes
22	*Daphnia magna*	Water flea	LC	21	R,M,T	Growth	170	.6	2.0	–	–	MATC	1.1	.51	–	–	–	Baird and others, 1990	Yes
23			LC	21	R,M,T	–	53	0.08	0.29	–	0.07	MATC	.15	0.15	–	–	16,h	Chapman and others, 1980; U.S. Environmental Protection Agency, 2001	No
24			LC	21	R,M,T	–	103	.16	.28	–	.23	MATC	.21	.14	–	–	16,h	Chapman and others, 1980; U.S. Environmental Protection Agency, 2001	No
25			LC	21	R,M,T	–	209	.21	.91	–	.33	MATC	.44	.18	–	–	16,h		No
26			LC	20	R,M,T	Mortality	200	.37	.48	–	.37	EC20	.37	.16	–	–	17	Canton and Sloof, 1982	Yes
27			LC	21	R,M,T	Reproduction	52	1.97	3.43	1.56	2.14	MATC	2.60	2.54	–	–	–	Chadwick Ecological Consultants, 2004a	Yes
28			LC	21	R,M,T	Reproduction	99	1.67	3.43	1.89	2.21	MATC	2.39	1.57	–	–	–		Yes
29			LC	21	R,M,T	Reproduction	150	5	10		–	MATC	7.07	3.58	–	–	–	Bodar and others, 1988	No
30			LC	21	R,M,T	Reproduction	130	.22	1.86	1.13	1.65	LOEC	1.65	.91	.54	–	18, 19	Borgmann and others, 1989a	No
31			LC	21	R,M,T	Reproduction	130	.22	1.86	1.65	1.69	LOEC	1.69	.93	–	–	19	Borgmann and others, 1989b	Yes
32	*Daphnia pulex*	Water flea	LC	18	R,M,T	Reproduction	52	14.6	>14.6	1.5	2.17	EC20	2.17	2.12	–	–	20	Chadwick Ecological Consultants, 2004a	Yes
33			LC	–	R,M,T	Reproduction	65	–	–	–	–	MATC	7.49	6.37	–	–	–	U.S. Environmental Protection Agency, 2001	No
34			LC	58	R,M,T	Reproduction	106	5	10	–	–	MATC	7.07	4.44	–	–	h	Ingersoll and Winner, 1982	Yes
35			LC	42	R,M,T	Reproduction	58	2.7	4.8	–	–	MATC	3.60	3.28	–	–	h	Winner, 1986	Yes
36			LC	42	R,M,T	Reproduction	116	5.6	10.8	–	–	MATC	7.78	4.62	–	–	h		Yes
37			LC	42	R,M,T	Reproduction	230	5.2	10.4	–	–	MATC	7.35	2.86	3.71	1.44	h		Yes
38	*Esox lucius*	Northern pike	ELS	60	F,M,T	Mortality	44	4.2	12.9	–	–	MATC	7.36	7.97	7.97	7.97	5	Eaton, 1978	No

Table 16. Chronic toxicity of cadmium to freshwater animals.—Continued

Test No.	Species	Common name	Test type	Duration (days)	Method	Lowest endpoint	Hardness (mg/L as $CaCO_3$)	NOEC (µg/L)	LOEC (µg/L)	EC_{10} (µg/L)	EC_{20} (µg/L)	Statistic used for chronic value	Chronic value (CV, µg/L)	Hardness adjusted CV (µg/L)	SMCV	GMCV	Code/notes	Reference	Data different than EPA (2001)?
39	*Etheostoma fonticola*	Fountain darter	JGS	7	R,M,T	Mortality	270	1.4	2.8	3.1	4.2	MATC	1.98	0.70	–	–	9	Castillo and Longley, 2001	Yes
40			JGS	7	R,M,T	Mortality	261	5.5	11.5	nc	nc	MATC	7.95	2.86	–	–	10	Castillo and Longley, 2001	Yes
41			JGS	7	R,M,T	Mortality	285	5.7	8.5	3.7	4.6	MATC	6.96	2.37	–	–	11		Yes
42			JGS	7	R,M,T	Mortality	270	6.6	9.6	7.1	8.5	MATC	7.96	2.8	–	–	12		Yes
43			JGS	7	R,M,T	Mortality	292	4	5.3	4.1	4.5	MATC	4.60	1.54	1.83	1.83	13		Yes
44	*Gammarus fasciatus*	Amphipod	JGS	42	R,M,T	Mortality	130	1.49	2.23	1.2	1.5	MATC	1.82	1.01	1.01	1.01	21	Borgmann and others, 1989b	Yes
45	*Hyalella azteca*	Amphipod	JGS	14	S,M,D	Mortality	17		0.25	–	–	MATC	.16	.31	–	–	6,h	Suedel and others, 1997	Yes
46			JGS	28	F,M,T	Mortality	280	.51	1.9	1.1	1.40	MATC	.98	.34	–	–	h	Ingersoll and Kemble, 2001	No
47			JGS	28	R,M,T	Mortality	126	.5	1.1	.30	.50	MATC	.74	.42	–	–	h	Chadwick Ecological Consultants, 2004a	Yes
48			JGS	28	R,M,T	Mortality	153	.8	1.3	.40	.70	MATC	1.02	.51	–	–	h		Yes
49			JGS	42	R,M,T	Mortality	130	.18	.34			MATC	.25	.14	–	–	–	Borgmann and others, 1991	Yes
50			JGS	42	R,M,T	Mortality	130	.57	.92	.64	.81	MATC	.72	.40	–	–	h	Borgmann and others, 1989b	Yes
51			JGS	42	R,M,T	Mortality	163	.48	.94	–	–	MATC	.67	.32	.33	.33	22,h	Stanley and others, 2005	Yes
52	*Jordanella floridae*	American flagfish	LC	~100	F,M,T	Reproduction	44	4.1	8.1	4.5	5.3	MATC	5.76	6.24	–	–	–	Spehar, 1976	Yes
53			LC	–	–	–	47.5	3	6.5	–	–	MATC	4.42	4.56	–	–	–	U.S. Environmental Protection Agency, 2001	No
54			LC	–	–	–	47.5	3.4	7.3	–	–	MATC	4.98	5.14	5.27	5.27	–		No
55	*Lepomis macrochirus*	Bluegill	LC	–	–	–	207	31	80	–	–	MATC	49.80	20.6	20.6	–	–	U.S. Environmental Protection Agency, 2001	No
56	*Micropterus dolomieui*	Smallmouth bass	ELS	60	F,M,T	Mortality	44	4.3	12.7	–	–	MATC	7.39	8.00	8.00	–	5	Eaton, 1978	No
57	*Oncorhynchus kisutch*	Coho salmon	ELS	27	F,M,T	Mortality	44	1.3	3.4	–	–	MATC	2.10	2.28	2.28	–	23	Eaton, 1978	No

Table 16. Chronic toxicity of cadmium to freshwater animals.—Continued

Test No.	Species	Common name	Test type	Duration (days)	Method	Lowest endpoint	Hardness (mg/L as $CaCO_3$)	NOEC (µg/L)	LOEC (µg/L)	EC_{10} (µg/L)	EC_{20} (µg/L)	Statistic used for chronic value	Chronic value (CV, µg/L)	Hardness adjusted CV (µg/L)	SMCV	GMCV	Code/ notes	Reference	Data different than EPA (2001)?
58	*Oncorhynchus mykiss*	Rainbow trout	ELS	53	F,M,D	Mortality	20	0.6	1.3	0.89	1.21	MATC	0.88	1.56	–	–	h	C.A. Mebane, D.P. Hennessy, and F.S. Dillon, unpub. data, 2004	Yes
59			ELS	62	F,M,D	Mortality	29	1.0	2.5	1.61	2.21	MATC	1.58	2.22	–	–	h		Yes
60			ELS	665	F,M,T	Reproduction	250	3.39	5.48	–	–	MATC	4.31	1.59	–	–	24,h	Brown and others, 1994	No
61			ELS	100	F,M,T	Mortality	46	1.25	1.74	2	–	MATC	1.47	1.55	–	–	25,h	Davies and others, 1993	Yes
62			JGS	100	F,M,T	Mortality	29	1.02	1.89	–	–	MATC	1.39	1.95	–	–	h	Davies and Brinkman, 1994b	Yes
63			JGS	100	F,M,T	Mortality	282	2.24	7.02	–	–	MATC	3.97	1.36	1.68	–	26,h		Yes
64	*Oncorhynchus tshawytscha*	Chinook salmon	ELS	120	F,M,T	Mortality	25	.96	1.3	1.33	1.83	MATC	1.12	1.72	1.72	1.87	27	Chapman, 1975, 1982	Yes
65	*Oreochromis aurea*	Blue tilapia	LC	–	–	–	145	52	–	–	–	NOEC	52.00	26.9	26.9	26.9	–	U.S. Environmental Protection Agency, 2001	No
66	*Pimephales promelas*	Fathead minnow	JGS	10	S,M,D	Mortality	17	1	2	–	–	MATC	1.41	2.76	–	–	6	Suedel and others, 1997	Yes
67			JGS	14	S,M,D	Mortality	17	2	3	–	–	MATC	2.45	4.78	–	–	–		Yes
68			ELS	32	F,M,T	Mortality	44	–	–	–	–	MATC	10.00	10.8	–	–	–	Spehar and Fiandt, 1986	No
69			ELS	32	F,M,T	Mortality	44	9	18	–	–	MATC	12.73	13.8	–	–	–	Spehar and Carlson, 1984	Yes
70			LC	~300	F,M,T	Mortality	201	37	110	–	–	MATC	63.80	26.8	–	–	–	Pickering and Gast, 1972	No
71			LC	~250	F,M,T	Mortality	201	27	57	–	–	MATC	39.23	16.5	–	–	–		No
72			JGS	7	R,M,T	Mortality	270	10.7	21.9	7.0	12.8	MATC	15.31	5.38	–	–	9	Castillo and Longley, 2001	Yes
73			JGS	7	R,M,T	Mortality	261	11.5	21.3	nc	–	MATC	15.65	5.62	–	–	10		Yes
74			JGS	7	R,M,T	Mortality	285	8.5	11.3	11.3	12.6	MATC	9.80	3.33	–	–	11		Yes
75			JGS	7	R,M,T	Mortality	272	9.6	12.2	8.7	10.8	MATC	10.82	3.79	–	–	12		Yes
76			JGS	7	R,M,T	Mortality	292	5.3	6.9	5.9	7.3	MATC	6.05	2.03	6.35	5.9	13		Yes
77	*Salmo salar*	Atlantic salmon	ELS	92	F,M,T	Alevin biomass	28	.47	.78	–	–	MATC	.61	.87	.87	.87	28	Rombough and Garside, 1982	Yes

Table 16. Chronic toxicity of cadmium to freshwater animals.—Continued

Test No.	Species	Common name	Test type	Duration (days)	Method	Lowest endpoint	Hardness (mg/L as $CaCO_3$)	NOEC (µg/L)	LOEC (µg/L)	EC_{10} (µg/L)	EC_{20} (µg/L)	Statistic used for chronic value	Chronic value (CV, µg/L)	Hardness adjusted CV (µg/L)	SMCV	GMCV	Code/ notes	Reference	Data different than EPA (2001)?
78	*Salmo trutta*	Brown trout	JGS	84	F,M,T	Mortality	37.6	–	0.7	–	–	LOEC	0.70	0.84	–	–	29	Davies and Brinkman, 1994c	Yes
79			JGS	136	F,M,T	Mortality	39.6	1	1.78	–	–	MATC	1.33	1.54	–	–	–	Davies and Brinkman, 1994a	Yes
80			ELS	55	F,M,T	Mortality	31	2.54	4.87	–	–	MATC	3.52	4.73	–	–	h	Brinkman and Hansen, 2004a	Yes
81			JGS	30	F,M,T	Mortality	29	0.74	1.4	–	–	MATC	1.02	1.43	–	–	–		Yes
82			ELS	55	F,M,T	Mortality	71	4.68	8.64	–	–	MATC	6.36	5.12	–	–	h		Yes
83			JGS	30	F,M,T	Mortality	68	1.3	2.58	–	–	MATC	1.83	1.51	–	–	–		Yes
84			ELS	55	F,M,T	Mortality	149	9.62	19.1	–	–	MATC	13.56	6.89	–	–	h		Yes
85			JGS	30	F,M,T	Mortality	151	4.81	8.88	–	–	MATC	6.54	3.29	–	–	–		Yes
86			LC	665	F,M,T	Mortality	250	9.3	29.1	–	–	MATC	16.45	6.07	–	–	h	Brown and others, 1994	No
87			ELS	60	F,M,T	Mortality	44	3.8	11.7			MATC	6.67	7.22	3.04	1.62	5,h	Eaton, 1978	No
88	*Salvelinus confluentus*	Bull trout	JGS	55	F,M,T	Mortality	29	.38	.78	0.49	0.64	MATC	.55	.77	.77	–	–	Hansen and others, 2002b	Yes
89	*Salvelinus fontinalis*	Brook trout	LC	1,100	F,M,T	Mortality	44	1.7	3.4	–	–	NOEC	1.70	1.84	–	–	30	Benoit and others, 1976	Yes
90			ELS	60	F,M,T	Mortality	37	1	3	1.7	2.8	MATC	1.70	2.05	–	–	–	Sauter and others, 1976	No
91			ELS	60	F,M,T	Mortality	188	3	7	2.9	7.9	MATC	4.58	2.02	–	–	–		Yes
92			ELS	60	F,M,T	Mortality	44	1.1	3.8	–	–	MATC	2.00	2.16	2.12	1.28	5	Eaton, 1978	No
93	*Salvelinus namaycush*	Lake trout	ELS	60	F,M,T	Mortality	44	4.4	12.3	–	–	MATC	7.36	7.96	7.96	–	5, 31	Eaton, 1978	No

Code/Notes: **Alpha notes:** h, data used in hardness-toxicity regressions.

Numbered notes:

1 Test considered acceptable for the chronic dataset because exposure durations and measured endpoints were similar to those used for chronic testing with fish (fish and salamanders have somewhat similar life spans), and because the EPA authors stated that the impetus for the testing the effects of long-term Cd exposures to salamanders was to address a data gap in the aquatic life criteria dataset. Unaware of any guidance on test durations/endpoints sufficient to elicit sensitive, chronic responses in salamanders. Leg regeneration was the most sensitive endpoint reported, with significantly slower regrowth at LOEC than in controls.

2 4.4 µg/L treatment considered LOEC: Eggs surviving to test termination was 57 percent of control. Statistical power very low because only 2 replicates (Holcombe and others, 1984, table 1).

3 2.5 µg/L treatment considered LOEC: <50 percent of control hatchability, <50 percent embryo survival, and 30 percent of control egg production (Holcombe and others, 1984, table 1).

4 NOEC was control, <1.0 µg/L. Cd caused 5-6 percent accelerated growth and metamorphosis at both 5 and 54 µg/L treatments, considered an adverse stress response by the authors. Decreased survival only observed at highest concentration tested (540 µg/L). Chronic value is uncertain because only three treatments were used with 10 times difference in concentrations (5, 54, and 540 µg/L) and because effects were observed at the lowest concentration tested. MATC value using less than detection control concentrations is greatly dependent on value assumed, assuming lower values such as 0.05 or 0.5 (i.e., half the detection value from other studies at same lab which used more sensitive methods, or half the detection limit from this study) would result in MATC much lower than lowest observed effect concentration. Because magnitude of observed effects seemed fairly small, such extrapolation does not seem justified by the information presented.

5 No information on magnitude of response given.

Table 16. Chronic toxicity of cadmium to freshwater animals.—Continued

Code/Notes:

6 Tests used static technique, however data considered usable because results appear reasonable in comparison to other tests with this species. Decision to use static testing in this test series was influenced by analyses showing that exposure concentrations, pH, and dissolved oxygen concentrations were acceptably stable. Organic loading from feeding was minimal, as evidenced by the lack of need for aeration (B. Suedell, Entrix, Inc., oral commun., 2005).

7 This chronic value was 31 times higher than the lowest hardness-adjusted chronic value for this species. Value was considered anomalous and was not used in GMCV calculations, even though the test appeared otherwise acceptable.

8 St. Louis river water with DOC >5 mg/L, but low values and a ratio of 1.0 for chronic *C. reticulata* value with reconstituted lab water and St. Louis water value suggest minimal DOC influence. See also table 16, note 7.

9 Test initiated 11/12/1999. Hardness values were not included in the source for the C. dubia test series, but were the same as hardness values from concurrent fountain darter and fathead tests that all used a common batch of dilution water (V. Castillo, Southwest Texas State University, written commun., 2005).

10 Test initiated 12/3/1999.

11 Test initiated 1/28/2000.

12 Test initiated 2/4/2000.

13 Test initiated 2/27/2000.

14 2003 test using Minnesota strain.

15 2004 test using Missouri strain.

16 EC20 values from U.S. Environmental Protection Agency, 2001. Source of U.S. Environmental Protection Agency's (2001) values is assumed to have been an unattributed author communication, because neither the EC20 values nor sufficient information to calculate them, were presented in the attributed source.

17 Effects were observed at lowest test concentration but magnitude of effect not given. The EC20 was considered the best NOEC estimate.

18 LOEC=20 percent reduction in brood size. Considered better estimate of chronic value than listing as "less than" or as the geometric mean of the control (0.22 µg/L NOEC) and LOEC. LOEC was lowest treatment.

19 Similar data that were described in more than one publication are assumed to reflect the results of a single test and are only one value is used.

20 MATC >14.6 µg/L, IC20 used instead as chronic value, assuming a 20 percent reduction in survival is at least biologically significant if not statistically significant.

21 NOEC resulted in 30 percent reduction in growth from controls even though not statistically different from control. Authors considered 55 percent mortality to be the LOEC and 18 percent mortality and 30 percent growth reduction to be the NOEC. Were the 30 percent growth reduction to have been considered the LOEC the MATC would have been 1.17 µg/L (0.92 and 1.49 µg/L limits). Control survival was only 45 percent. Because a concentration response was observed with cadmium, and controls and low treatment concentrations were statistically distinguished from higher treatments by analysis of variance (ANOVA), the responses were considered acceptably reliable and the data usable.

22 Tested in reconstituted hard water. In matched tests conducted in effluent dominated stream mesocosms or laboratory tests using effluent dominated dilution water, cadmium was considerably less toxic.

23 Test initiated with sac fry about 1 week after hatching.

24 Lowest effect was with second generation fry development, which was significantly affected when the parents were exposed to 1.77 µg/L cadmium, but not when exposed to 0.47 µg/L cadmium (control). However, second generation embryo survival for all groups was less than 60 percent, which may have influenced the fry development effect levels. The ability of the first generation adults to reach sexual maturity, with NOEC and LOEC values of 3.39 and 5.48 µg/L cadmium, respectively was considered a more reliable endpoint. See also U.S. Environmental Protection Agency (2001, p. 15)

25 Long term growth and survival test, initiated with 36 g juveniles.

26 Authors considered 2.5 percent mortality LOEC with no statistical comparisons. 2.5 percent mortality is considered here to be a biologically minor effect, and is considered the NOEC concentration. 7.3 µg/L total Cd considered LOEC with 37.5 percent mortality. Test conducted in "aged" hard water (allowed to equilibrate).

27 190 percent increase in mortality over controls (18 vs... 9 percent) with at 1.3 µg/L treatment, which is considered the LOEC.

Table 16. Chronic toxicity of cadmium to freshwater animals.—Continued

Code/Notes:

28 Statistical LOEC of 0.47 µg/L resulted in reduced alevin weight and growth, lowest concentration tested, although authors suggest threshold would be 0.13 µg/L (p. 2012) - *"We observed significant reduction in the growth of salmon alevins in 0.47 µg/L and examination of a plot of final alevin wet weight versus ambient cadmium concentration indicated a response threshold near the control concentration of 0.13 µg/L. This suggests that levels currently assumed to be safe may have some deleterious effect on populations, although probably of such a small magnitude as to be of little consequence."* Based on this statement, conclude statistical LOEC is more like a biological NOEC. Next higher concentration also had reduced survival and statistically significant reduction in biomass (their table 6). To summarize effects for this synthesis, treatment 1 (0.47 µg/L) is considered the NOEC with biomass 99 percent of control, treatment 2 (0.78 µg/L) is considered LOEC with biomass 72 percent of control. Average zinc in control water was 95 µg/L during 1977 test and 80 µg/L during the 1978 test. Because response pattern tracked cadmium concentrations with adequate control survival, and responses were relative to controls, background zinc probably did not compromise tests. One each tests were conducted in the winters of 1977 and 1978. Results from 1977 were much more resistant than results from 1978, only the 1978 results were used for this report. Hardnesses were 19 mg/L (Ca:Mg ratio 2:1) in winter 1977 and 28 mg/L (Ca:Mg ratio 7:1) in winter 1978. The more sensitive results were obtained in 1978 with average temperature of 10°C and pH 7.3 compared with average temperature of 5°C and pH 6.5 in 1977.

29 25 percent mortality at lowest concentration tested

30 NOEC judged to be best estimate of chronic value. MATC considered neither acceptable nor a threshold because the LOEC resulted in complete mortality to spawning males.

31 This chronic value was >10 times higher than the lowest hardness-adjusted chronic value for this species. Value was considered anomalous and was not used in GMCV calculations, even though the test was presumably otherwise acceptable.

Table 17. Other data on effects of cadmium to freshwater organisms.

[Each row describes one test. Underlined values are lower than their corresponding acute or chronic criterion values. **Size or age at test initiation:** CSL, chronic sublethal; ELS, early life stage; PLC, partial life cycle (survival, growth, and reproduction, less than one complete life cycle). **Method:** D, dissolved test concentrations measured after being filtered through a 0.45-µm filter; F, flow-through test exposures; M, test concentrations measured during exposures; R, renewed test exposures; S, static test exposures; T, total metal concentrations determined from unfiltered samples; U, unmeasured test concentrations; ?, unknown. **NOEC:** Highest no observed effect concentrations. **LOEC:** Lowest observed effect concentration. **Summary statistic:** MATC, maximum acceptable threshold concentration. Calculated as the geometric mean of the NOEC and LOEC. LC_{50}, concentration killing 50 percent of test organisms; ILL, incipient lethal level. **Notes:** ALC, aquatic live criteria; Cd, cadmium; Cu, copper; Hg, mercury; $MgSO_4$, magnesium sulfate. **Abbreviations:** °C, degrees Celsius; DOC, dissolved organic carbon; EPA, U.S. Environmental Protection Agency; g, gram; ITIS, Integrated Taxonomic Information System; g, gram; GMCV, genus mean chronic value; nc, not calculable; µg/L, microgram per liter; µm, micrometer; mg/kg, milligram per kilogram; mm, millimeter; Pb, lead; SMAV, species mean acute value; SMCV, species mean chronic value; USGS, U.S. Geological Survey; YOY, young-of-year fish; <, less than; >, greater than; ~, approximately; –, missing or inapplicable values]

Species	Common name	Size or age at test initiation	Exposure duration (days)	Method	Effect	Hardness (mg/L as $CaCO_3$)	NOEC (µg/L)	LOEC (µg/L)	Summary statistic	Summary effect value (µg/L)	Corresponding acute or chronic criterion value (µg/L)	Notes	Reference
Ambystoma gracile	Northwestern salamander	3-mo old larvae	24	F,M,T	Growth	45	48.9	193	MATC	97.15	0.38	Overall growth and survival were less sensitive endpoints than limb regeneration.	Nebeker and others, 1994, 1995
Brachionus calyciflorus	Rotifer	Cysts	2	S,U	Reproduction	90	18	25	MATC	21.2	14.7	Species has a cyclically parthenogenetic life cycle; Sexual reproduction (mictric phase) more affected by contaminants than asexual (amictic phase).	Snell and Carmona, 1995
Brachycentrus sp.	Caddisfly	5-8 mm	28	F,M,T	Behavior	45	27.5	85.5	MATC	48.5	.36	Behavior of caddisflies at 85 and 238 µg/L was abnormal. The affected larvae showed a vigorous curling motion and were usually free swimming. Essentially all control larvae were observed in larval cases and responded slowly to probing.	Spehar and others, 1978a
Bufo americanus	American toad	Tadpoles	103	M,S,T	Mortality and growth	60	–	.5	LOEC	.45	.43	Slightly reduced survival (80 vs.. 88 percent for controls), delayed development of tadpoles, and a non-significant trend of older and smaller metamorphs exposed to Cd exposed to Cd through periphyton (about 10 mg/kg dry weight and water (5 to <0.1 µg/L). Adverse effects were noted in the lowest cadmium exposure which was dosed with a single slug resulting in an initial nominal aqueous concentration of 5 µg/L. Aqueous concentrations dropped to <0.1 µg/L during the course of the test. Tissue-residue LOEC was 16.7 mg/kg dry weight (whole body).	James and others, 2005

Table 17. Other data on effects of cadmium to freshwater organisms.—Continued

Species	Common name	Size or age at test initiation	Exposure duration (days)	Method	Effect	Hardness (mg/L as $CaCO_3$)	NOEC (µg/L)	LOEC (µg/L)	Summary statistic	Summary effect value (µg/L)	Corresponding acute or chronic criterion value (µg/L)	Notes	Reference
Ceriodaphnia dubia	Water flea	Neonate	7	R,U	Reproduction	90	0.5	1.0	MATC	0.71	0.55	Reproductive endpoints more sensitive than survival, and were as low as those generated from longer duration tests. No analytical confirmation of exposure concentrations.	Winner, 1988
Chironomus riparius	Midge	10-12 mm (late instar)	4	F,M,T	Mortality	152	–	–	LC50	300,000	1.84	Late instar insects may be much less sensitive than early instars (1 and 2).	U.S. Environmental Protection Agency, 2001
Coregonus clupeaformis	Lake whitefish	10 month old	.05	F,M,T	Behavior	81	<.2	.2	LOEC	.2	.51	Detected and responded to area of trough with Cd; response could be either preference or avoidance.	McNicol and Scherer, 1991
Coregonus clupeaformis	Lake whitefish	10 month old	.1	F,M,T	Behavior	81	1	5	MATC	2.2	.51	Avoidance under uniform lighting conditions.	McNicol and others, 1999
Coregonus clupeaformis	Lake whitefish	10 month old	.75	F,M,T	Behavior	81	50	125	MATC	79	.51	Cd began to force fish to leave shaded area of the tank, which they were strongly attracted to.	McNicol and others, 1999
Coregonus clupeaformis	Lake whitefish	10 month old	14	F,M,T	Mortality	81	–	–	LC50	361	1.1	Young-of-year fish (YOY) less sensitive than yearlings.	McNicol, 1997; McNicol and others, 1999
Crangonyx pseudogracilis	Amphipod	Adult, 4 mm	4	R,U	Mortality	50	–	–	LC50	1,700	.38	Test value indicates that the order Amphipoda is not uniformly sensitive to Cd. Test not used in acute dataset because of no analytical confirmation of exposure concentrations.	Martin and Holdich, 1986

Table 17. Other data on effects of cadmium to freshwater organisms.—Continued

Species	Common name	Size or age at test initiation	Exposure duration (days)	Method	Effect	Hardness (mg/L as $CaCO_3$)	NOEC (µg/L)	LOEC (µg/L)	Summary statistic	Summary effect value (µg/L)	Corresponding acute or chronic criterion value (µg/L)	Notes	Reference
Daphnia galeata mendota	Water flea	Mixed population	154	R,M,T	Population	136	1	2	MATC	1.4	0.71	Reduction in carrying capacity. Due to compensatory increases, population attributes were less sensitive than most sensitive individual attributes. Study was located too late to include in chronic criterion dataset, although it would have been otherwise usable. Water hardness not reported, hardness values for the same water source ranged from 135-137 mg/L in 2003-05, city of Chicago tapwater (filtered Lake Michigan water), www.cityofchicago.org/water	Marshall, 1978
Daphnia magna	Water flea	≤24hrs old	4	S,M,T	Mortality	17	–	–	LC50	12.7	.31	Data point not used because most other data for *Daphnia magna* were from 48-hour exposures with no feeding. Feeding may reduce cadmium toxicity in 48-hour exposures by about 2 times (Lewis and Weber 1985). However, Suedel and others (1997) data shows that Daphnia were about 2 times more sensitive in 96-hour exposures with feeding than in 48-hour exposures without feeding.	Suedel and others, 1997
Daphnia magna	Water flea	≤24hrs old	2	S,M,T	Mortality	17	–	–	LC50	26.4	.31	–	Suedel and others, 1997
Daphnia magna	Water flea	Neonate	7	R,U	Growth	90	1	2	MATC	1.4	.55	Concluded that a 7-day test was insufficiently sensitive for *D. magna* reproduction endpoints, and recommended C. dubia as a reasonably sensitive short-term chronic test. No analytical confirmation of exposure concentrations.	Winner, 1988
Daphnia magna	Water flea	8-days old	8	R,M,T	Reproduction	170	.2	.4	MATC	.28	.81	Two instars were exposed starting at 8-days.	Barata and Baird, 2000

Table 17. Other data on effects of cadmium to freshwater organisms.—Continued

Species	Common name	Size or age at test initiation	Exposure duration (days)	Method	Effect	Hardness (mg/L as $CaCO_3$)	NOEC (µg/L)	LOEC (µg/L)	Summary statistic	Summary effect value (µg/L)	Corresponding acute or chronic criterion value (µg/L)	Notes	Reference
Ephemerella sp.	Mayfly	5-8 mm	28	F,M,T	Mortality	45	–	3	MATC	1.50	0.36	LC50 ~3 µg/L, adjusted for 31 percent control mortality, higher treatments resulted in 84-94 percent mortality although not consistent with concentration. Almost all control and treatment mortality occurred after day 21. Treatment spacing and lack of concentration-response precludes calculating statistics, LC50/2 is MATC best guess.	Spehar and others, 1978a
Gila elegans	Bonytail chub	Larva	4	S,U	Mortality	199	–	–	LC50	148	2.30	No analytical confirmation of exposure concentrations.	Buhl, 1997
Gila elegans	Bonytail chub	Juvenile	4	S,U	Mortality	199	–	–	LC50	168	2.30	No analytical confirmation of exposure concentrations.	Buhl, 1997
Hyalella azteca	Amphipod	Adults and larger juveniles	4	S,M,T	Mortality	34	–	–	LC50	8	.54	This acute test included adult organisms and also was the highest acute value for the species (~10 times greater than lowest value). While no age-sensitivity relationship is known of for this species, it still seemed possible that the most sensitive life stage was not tested and thus the value was excluded from the SMAV.	Nebeker and others, 1986b
Hyalella azteca	Amphipod	7-14 days old	10	F,M,T	Mortality	45	–	–	LC50	2.8	.68	Hardness assumed from other tests at EPA's Duluth lab using Lake Superior water.	Phipps and others, 1995
Hyalella azteca	Amphipod	Juvenile	10	F,M,T	Mortality	45	–	–	LC50	2.8	.68	Unclear if the reported results reflected two tests with identical results or the same test described twice.	Ankley and others, 1991
Hyalella azteca	Amphipod	1-11 days	7	S,M,T	Mortality	18	–	–	LC50	.15	.20	Cd was the most toxic of the 63 metals tested with *Hyalella* at low hardness. Seven-day test is still an acute test, but was a better predictor of chronic toxicity than were 48- or 96-hour tests. Here the 7-day LC50s are compared to the chronic criterion values.	Borgmann and others, 2005

Table 17. Other data on effects of cadmium to freshwater organisms.—Continued

Species	Common name	Size or age at test initiation	Exposure duration (days)	Method	Effect	Hardness (mg/L as $CaCO_3$)	NOEC (µg/L)	LOEC (µg/L)	Summary statistic	Summary effect value (µg/L)	Corresponding acute or chronic criterion value (µg/L)	Notes	Reference
Hyalella azteca	Amphipod	1-11 days	7	S,M,T	Mortality	124	–	–	LC50	1.6	0.67	Cd was the most toxic of the 63 metals tested with Hyalella at moderate hardness. Seven-day test is still an acute test, but was a better predictor of chronic toxicity than were 48- or 96-hour tests. Here the 7-day LC50s are compared to the chronic criterion values.	Borgmann and others, 2005
Hyalella azteca	Amphipod	2-3 week old juveniles	3	R,M,T	Mortality	8			LC50	1.9	.17	Non-standard duration.	Gust, 2006
Hydropsyche betteni	Caddisfly	5-8 mm	28	F,M,T	Behavior	45	27.5	85.5	MATC	48.49	.36	Behavior of caddisflies at 85 and 238 µg/L was abnormal. The affected larvae showed a vigorous curling motion and were usually free swimming. Essentially all control larvae were observed in larval cases and responded slowly to probing	Spehar and others, 1978a
Lumbriculus variegatus	Oligochaete	Mixed age	10	F,M,T	Mortality	45	–	–	LC50	158	.68	10- and 4-day LC50s were similar.	Ankley and others, 1991
Lymnaea palustris	Pulmonate pond snail	PLC	~100	R,U	Hatching (embryo-genesis inhibition)	250	<40	40	LOEC	40	1.03	Complete inhibition of hatching. Cd concentrations unmeasured, however, in other studies in their lab they always had nominal Cd within 30 percent of measured in 1-week static exposures. (M. Coeurdassier, University of Franche-Comte, Besançon, France, written commun., 2005)	Coeurdassier and others, 2003
Lymnaea stagnali	Pulmonate pond snail	4-week old juvenile	4	S,M,T	Mortality	250	–	–	LC50	742	2.77	Juveniles apparently more sensitive than older animals.	Coeurdassier and others, 2004
Lymnaea stagnali	Pulmonate pond snail	9-week old juvenile	4	S,M,T	Mortality	250	–	–	LC50	1,515	2.77	Juveniles apparently more sensitive than older animals.	Coeurdassier and others, 2004
Lymnaea stagnali	Pulmonate pond snail	20-week old adult	4	S,M,T	Mortality	250	–	–	LC50	1,585	2.77	Juveniles apparently more sensitive than older animals.	Coeurdassier and others, 2004
Lymnaea stagnali	Pulmonate pond snail	PLC	49	S,U	Hatching	250	<25	25	–	25	1.03	No analytical confirmation of exposure concentrations.	Gomot, 1998
Moina macrocopa	Planktonic cladoceran	24 hrs old	4	S,U	Mortality	396			LC50	680	4.03	No analytical confirmation of exposure concentrations.	García and others, 2004

Table 17. Other data on effects of cadmium to freshwater organisms.—Continued

Species	Common name	Size or age at test initiation	Exposure duration (days)	Method	Effect	Hardness (mg/L as $CaCO_3$)	NOEC (µg/L)	LOEC (µg/L)	Summary statistic	Summary effect value (µg/L)	Corresponding acute or chronic criterion value (µg/L)	Notes	Reference
Moina macrocopa	Planktonic cladoceran	24 hrs old	15	R,U	Population growth rate	396	5	10	MATC	7.07	1.36	No analytical confirmation of exposure concentrations.	García and others, 2004
Moina macrocopa	Planktonic cladoceran		20	R,U	Reduced survival	82	.2	0.4	MATC	.28	.52	No analytical confirmation of exposure concentrations.	Hatakeyama and Yasuno, 1981
Moina triserialis	Planktonic cladoceran	24 hrs old	4	S,U	Mortality	396	–	–	LC50	420	4.03	No analytical confirmation of exposure concentrations.	García and others, 2004
Moina triserialis	Planktonic cladoceran	24 hrs old	31	R,U	Population growth rate	396	11	21	MATC	15.2	1.36	No analytical confirmation of exposure concentrations.	García and others, 2004
Morone saxatilis	Striped bass	63-days old	4	S,U	Mortality	40	–	–	LC50	4	.62	No analytical confirmation of exposure concentrations.	Palawski and others, 1985
Morone saxatilis	Striped bass	63-days old	4	S,U	Mortality	285	–	–	LC50	10	3.08	No analytical confirmation of exposure concentrations.	Palawski and others, 1985
Oncorhynchus kisutch	Coho salmon	Embryo (ELS)	82	F,M,T	Mortality	45.3	4.1	12.5	MATC	7.16	.36	Exposed as eyed eggs 20 days before hatching which may have induced acclimation. Fish were 4 times more resistant than companion test with coho salmon which introduced fish 1-week after hatching. The two tests used fish from different stocks, so whether the differences are due to acclimation or stock differences is unknown. Since the differences could have been from acclimation influences, the more resistant results were not used in criteria calculations.	Eaton, 1978
Oncorhynchus kisutch	Coho salmon	Adult	4	F,M,T	Mortality	22	–	–	LC50	17.5	.38	Only the most sensitive life stages (swim-up fry and parr) used directly to estimate acute values. More resistant younger and older life stages not used in acute criteria calculations.	Chapman, 1975

Table 17. Other data on effects of cadmium to freshwater organisms.—Continued

Species	Common name	Size or age at test initiation	Exposure duration (days)	Method	Effect	Hardness (mg/L as $CaCO_3$)	NOEC (µg/L)	LOEC (µg/L)	Summary statistic	Summary effect value (µg/L)	Corresponding acute or chronic criterion value (µg/L)	Notes	Reference
Oncorhynchus kisutch	Coho salmon	2-weeks old	4	R,M,T	Mortality	22	–	–	LC50	1,000	0.38	LC50 "greater than" value. Cu, Pb, and Hg reported above chronic ALC values in dilution water, considered data artifacts due to use of pre-"clean" methods and unlikely to contribute to test mortalities. Control mortalities not reported, assumed to be zero.	Chapman, 1975
Oncorhynchus kisutch	Coho salmon	4-weeks old	4	R,M,T	Mortality	22	–	–	LC50	3.66	.38	LC50 calculated by trimmed Spearman-Karber method ("TSK," U.S. Environmental Protection Agency, 1992) using nominal concentrations (exposures were reported to have been measured but data were not presented). Control mortalities not reported, assumed to be zero. Cu, Pb, and Hg reported above chronic ALC values in dilution water, considered data artifacts due to use of pre-"clean" methods. Cu, Pb, and Hg reported above chronic ALC values in dilution water, considered data artifacts due to use of pre-"clean" methods, unlikely to contribute to test mortalities.	Chapman, 1975
Oncorhynchus kisutch	Coho salmon	5-weeks old	4	R,M,T	Mortality	22	–	–	LC50	2.76	.38	LC50 calculated using Probit regression (U.S. Environmental Protection Agency, 1992). Exposures reported to be measured but data not presented. Cu, Pb, and Hg reported above chronic ALC values in dilution water, considered data artifacts due to use of pre-"clean" methods. Control mortalities not reported, assumed to be zero.	Chapman, 1975

Table 17. Other data on effects of cadmium to freshwater organisms.—Continued

Species	Common name	Size or age at test initiation	Exposure duration (days)	Method	Effect	Hardness (mg/L as $CaCO_3$)	NOEC (µg/L)	LOEC (µg/L)	Summary statistic	Summary effect value (µg/L)	Corresponding acute or chronic criterion value (µg/L)	Notes	Reference
Oncorhynchus kisutch	Coho salmon	6-weeks old	4	R,M,T	Mortality	22	–	–	LC50	1.73	0.38	LC50 calculated by TSK (U.S. Environmental Protection Agency, 1991) using nominal concentrations (exposures were reported to have been measured but data were not presented). Control mortalities not reported, assumed to be zero. Cu, Pb, and Hg reported above chronic ALC values in dilution water, considered data artifacts due to use of pre-“clean” methods.	Chapman, 1975
Oncorhynchus kisutch	Coho salmon	7-weeks old	4	R,M,T	Mortality	22	–	–	LC50	1.40	.38	LC50 calculated using Probit regression (U.S. Environmental Protection Agency, 1992). Exposures reported to be measured but data not presented. Control mortalities not reported, assumed to be zero. Cu, Pb, and Hg reported above chronic ALC values in dilution water, considered data artifacts due to use of pre-“clean” methods.	Chapman, 1975
Oncorhynchus kisutch	Coho salmon	21-weeks old	4	F,M,T	Mortality	24	–	–	LC50	2.7	.41	Life stage apparently more resistant than other life stages. Cu, Pb, and Hg reported above chronic ALC values in dilution water, considered data artifacts due to use of pre-“clean” methods.	Chapman, 1975
Oncorhynchus kisutch	Coho salmon	Adult	4	F,M,T	Mortality	24	–	–	LC50	17.5	.41	Life stage apparently more resistant than other life stages. Cu, Pb, and Hg reported above chronic ALC values in dilution water, considered data artifacts due to use of pre-“clean” methods.	Chapman, 1975
Oncorhynchus kisutch	Coho salmon	Yearling	70	F,M,T	Seawater adaptation	90	4.5		NOEC	4.5	.55	No effect on capacity to adapt to seawater, cortisol levels (a measure of stress) or mortality.	Schreck and Lorz, 1978
Oncorhynchus mykiss	Rainbow trout	CSL with fry	100	F,M,T	Mortality	217	–	–	MATC	3.58	.94	Atypical dilution water. Hardness amended with $MgSO_4$ only resulting in very low Ca:Mg ratio.	Davies and others, 1993

Table 17. Other data on effects of cadmium to freshwater organisms.—Continued

Species	Common name	Size or age at test initiation	Exposure duration (days)	Method	Effect	Hardness (mg/L as $CaCO_3$)	NOEC (µg/L)	LOEC (µg/L)	Summary statistic	Summary effect value (µg/L)	Corresponding acute or chronic criterion value (µg/L)	Notes	Reference
Oncorhynchus mykiss	Rainbow trout	CSL with fry	100	F,M,T	Mortality	414	–	–	MATC	3.64	1.40	Atypical dilution water. Hardness amended with $MgSO_4$ only resulting in very low Ca:Mg ratio.	Davies and others, 1993
Oncorhynchus mykiss	Rainbow trout	CSL with fry	100	F,M,T	Mortality	300	8.2	14.2	MATC	10.79	1.15	Atypical dilution water. "Unaged" dilution water where highly alkaline water was mixed quickly without allowing equilibrium with Cd.	Davies and Brinkman, 1994b
Oncorhynchus mykiss	Rainbow trout	0.19 g fry	14	F,M,T	Mortality	280	–	–	ILL	20	1.10	Test at 7.5˚C, non-Cd acclimated controls.	Stubblefield and others, 1999
Oncorhynchus mykiss	Rainbow trout	Adult	14	F,M,T	Mortality	280	–	–	ILL	6.1	1.10	Test at 7.5˚C, non-Cd acclimated controls.	Stubblefield and others, 1999
Oncorhynchus mykiss	Rainbow trout	Adult	2	S,M,T	Behavior	120	4	–	NOEC	4	.65	No change in dominance hierarchies. Two treatments (4 and 7 µg/L).	Sloman and others, 2005
Oncorhynchus mykiss	Rainbow trout	Adult	–	S,M,T	Behavior	120	–	2	LOEC	2	.65	Decreased competitive ability and increased growth in Cd exposed fish. Single exposure concentration.	Sloman and others, 2003b
Oncorhynchus mykiss	Rainbow trout	0.58 g fry	1	S,M,T	Behavior	120		3.17	LOEC	3.17	.65	Cd exposed fish were less aggressive and subordinate to non-Cd exposed fish.	Sloman and others, 2003a
Oncorhynchus mykiss	Rainbow trout	2.5 g fry	7	F,M,T	Behavior	120	.56	2.33	MATC	1.14	.65	Normal antipredator behaviors exhibited in response to chemical alarm substance were lost following Cd exposure.	Scott and others, 2003
Oncorhynchus mykiss	Rainbow trout	Swimup fry	4	R,U	Neurotoxic	163	–	5	LOEC	5	.79	No effect on swimming behavior or brain neurochemical activity (cholinesterase (ChE) activity, muscarinic cholinergic receptor (MChR) number, and MChR affinity).	Beauvais and others, 2001
Oncorhynchus mykiss	Rainbow trout	Fingerling, 18 g	4	S,M,T	Mortality	50	–	–	LC50	1.88	.74	4- and 14-day acute values similar.	Stubblefield, 1990
Oncorhynchus mykiss	Rainbow trout	Fingerling, 18 g	14	F,M,T	Mortality	50	–	–	LC50	1.56	.74	4- and 14-day acute values similar.	Stubblefield, 1990
Oncorhynchus mykiss	Rainbow trout	5.6 g	30	F,M,T	Various	20	–	–	NOEC	.11	.22	No effects on survival, growth, swimming, or acclimation.	Hollis and others, 2000a
Oncorhynchus mykiss	Rainbow trout	–	30	F,M,T	Mortality	25	–	–	LOEC	2	.25	80 percent mortality in first 5 days.	Hollis and others, 2000b

Table 17. Other data on effects of cadmium to freshwater organisms.—Continued

Species	Common name	Size or age at test initiation	Exposure duration (days)	Method	Effect	Hardness (mg/L as $CaCO_3$)	NOEC (µg/L)	LOEC (µg/L)	Summary statistic	Summary effect value (µg/L)	Corresponding acute or chronic criterion value (µg/L)	Notes	Reference
Oncorhynchus mykiss	Rainbow trout	–	30	F,M,T	Physiology	47	–	–	LOEC	2	0.37	Impaired swimming performance.	Hollis and others, 2000b
Oncorhynchus mykiss	Rainbow trout	~1.4 g	30	F,M,T	Mortality	140	3	10	MATC	5.48	.72	No effects on survival, growth, swimming, or ion composition at NOEC; 30 percent mortality but no other effects at LOEC.	Hollis and others, 1999
Oncorhynchus mykiss	Rainbow trout	10-12 g	30	F,M,T	Bioaccumu-lation	140	3	–	NOEC	3	.72	No effects on survival, growth; dietary calcium supplementation reduced bioaccumulation, but also reduced growth.	Franklin and others, 2005
Oncorhynchus mykiss	Rainbow trout	Juvenile	4	F,M,T	Mortality	25	–	–	LC50	2.0	.42	Michael A. Cairns data, as cited by Chapman (1982). Chapman (1982) suggested that the data would be suitable for use in criteria derivation and that the complete data may be in Cairns' thesis. However, his Oregon State University M.S. thesis was located and reviewed but no relevant data were found. Cairns also was located but could provide no additional information.	Chapman, 1982
Oncorhynchus mykiss	Rainbow trout	Embryo (ELS)	~72	F,M,T	Mortality	25	2.6	3.7	MATC	3.10	.25	Michael A. Cairns data, as cited by Chapman (1982). See above.	Chapman, 1982
Oncorhynchus mykiss	Rainbow trout	Juvenile	120	F,M,T	Biochemical	320	1	10	LOEC	10	1.20	Important biochemical functions impaired. Exposure concentrations considered too wide to calculate MATC.	Arillo and others, 1984

Table 17. Other data on effects of cadmium to freshwater organisms.—Continued

Species	Common name	Size or age at test initiation	Exposure duration (days)	Method	Effect	Hardness (mg/L as $CaCO_3$)	NOEC (µg/L)	LOEC (µg/L)	Summary statistic	Summary effect value (µg/L)	Corresponding acute or chronic criterion value (µg/L)	Notes	Reference
Oncorhynchus mykiss	Rainbow trout	Adult	540	?	Hatching	102	–	0.2	LOEC	<u>0.2</u>	<u>0.59</u>	Reduced survival (52 vs. 90 percent for control) of 4-day old larvae of rainbow trout after parents were exposed to a concentration of 0.2 µg/L at 102 mg/L hardness for 18 months, which is well below the published chronic criterion. The exposed parents had tissue concentrations that were roughly 7 times that of the control fish. This magnitude of effect in an important biological endpoint could justify adjusting the criterion per Stephan and others (1985) recommendations. However, the description of this test was too sparse to answer important questions about the methods and results, such as how the fish were exposed, whether exposure concentrations were measured at sufficient frequency to define the exposure concentrations, and whether treatments were replicated. Because of these uncertainties and because the LOEC is about 10 times lower than any other adverse effect concentrations for this genus, these results probably do not warrant adjusting the chronic criterion.	Birge and others, 1981, citing unpublished data by W.J. Birge and J.A. Black
Orconectes juvenilis	Crayfish	3rd to 5th instars, ~0.02 g	4	R,M,T	Mortality	44	–	–	LC50	60	.67	–	Wigginton, 2005
Orconectes juvenilis	Crayfish	Adult, 4.6 g	4	R,M,T	Mortality	44	–	–	LC50	2,440	.67	Adults were 40 times more resistant than newly hatched crayfish.	Wigginton, 2005
Orconectes placidus	Placid crayfish	3rd to 5th instars, ~0.02 g	4	R,M,T	Mortality	55	–	–	LC50	37	.80	–	Wigginton, 2005
Orconectes placidus	Placid crayfish	Adult, 7 g	4	R,M,T	Mortality	44	–	–	LC50	487	.67	Adults were 13 times more resistant than newly hatched crayfish.	Wigginton, 2005

Table 17. Other data on effects of cadmium to freshwater organisms.—Continued

Species	Common name	Size or age at test initiation	Exposure duration (days)	Method	Effect	Hardness (mg/L as $CaCO_3$)	NOEC (µg/L)	LOEC (µg/L)	Summary statistic	Summary effect value (µg/L)	Corresponding acute or chronic criterion value (µg/L)	Notes	Reference
Physa integra	Snail	6-15 mm	21	F,M,T	Mortality	45	3	8	EC10	5	0.36	High control survival and concentration-response through day 21, afterwards controls and all treatments suffered high mortalities that did not correspond to concentrations.	Spehar and others, 1978a
Pimephales promelas	Fathead minnow	24-48 hours	7	S,M,D	Mortality	17	–	–	LC50	4.4	.20	10- and 14-day exposures more sensitive than 7-day exposures. Because longer exposures were available for a given test condition (hardness 17), only the 10- and 14-day results were used in SMAV and hardness regressions.	Suedel and others, 1997
Pimephales promelas	Fathead minnow	24-48 hours	10	S,M,D	Mortality	17	–	–	LC50	1.6	.20	10- and 14-day exposures more sensitive than 7-day exposures. Because longer exposures were available for a given test condition (hardness 17), only the 10- and 14-day results were used in SMAV and hardness regressions.	Suedel and others, 1997
Pimephales promelas	Fathead minnow	24-48 hours	14	S,M,D	Mortality	17	–	–	LC50	2.3	.20	10- and 14-day exposures more sensitive than 7-day exposures. Because longer exposures were available for a given test condition (hardness 17), only the 10- and 14-day results were used in SMAV and hardness regressions.	Suedel and others, 1997
Pimephales promelas	Fathead minnow	Embryo (ELS)	7	S,M,D	Mortality	17	4	6	MATC	4.90	.20	10- and 14-day exposures more sensitive than 7-day exposures. Because longer exposures were available for a given test condition (hardness 17), only the 10- and 14-day results were used in SMAV and hardness regressions.	Suedel and others, 1997

Table 17. Other data on effects of cadmium to freshwater organisms.—Continued

Species	Common name	Size or age at test initiation	Exposure duration (days)	Method	Effect	Hardness (mg/L as $CaCO_3$)	NOEC (µg/L)	LOEC (µg/L)	Summary statistic	Summary effect value (µg/L)	Corresponding acute or chronic criterion value (µg/L)	Notes	Reference
Pimephales promelas	Fathead minnow	Embryo (ELS)	10	S,M,D	Mortality	17	1	2	MATC	1.41	0.20	10- and 14-day exposures more sensitive than 7-day exposures. Because longer exposures were available for a given test condition (hardness 17), only the 10- and 14-day results were used in SMAV and hardness regressions.	Suedel and others, 1997
Pimephales promelas	Fathead minnow	Embryo (ELS)	14	S,M,D	Mortality	17	2	3	MATC	2.45	.20	10- and 14-day exposures more sensitive than 7-day exposures. Because longer exposures were available for a given test condition (hardness 17), only the 10- and 14-day results were used in SMAV and hardness regressions.	Suedel and others, 1997
Pimephales promelas	Fathead minnow	Adult	21	F,M,T	Behavior	349	13	18	MATC	15.3	1.26	Cd exposed minnows were more vulnerable to predation and displayed altered behavior patterns, including abnormal schooling behavior.	Sullivan and others, 1978
Potamopyrgus antipodarum	New Zealand mud snail	Juvenile, 3-4 mm	2	R,M,T	Mortality	148	–	250	LOEC	250	.74	100 percent mortality. Mortality was delayed—following exposure for 48 hours and transfer to clean water for observation, after 100 hours in clean water all had died. 48-hour LC50 was 560 µg/L.	Jensen and Forbes, 2001
Procambarus clarkii	Red swamp crayfish	3rd to 5th instars, ~0.02 g	4	R,M,T	Mortality	–	–	–	LC50	42	624	–	Wigginton, 2005
Procambarus clarkii	Red swamp crayfish	Adult, 18.5 g	4	R,M,T	Mortality	–	–	–	LC50	53	2,660	Adults were 4 times more resistant than newly hatched crayfish.	Wigginton, 2005
Prosopium williamsoni	Mountain whitefish	Field collected, 209 g	14	F,M,T	Mortality	50	2.09	4.08	MATC	2.92	.38	Best estimate of a chronic value for this genus.	Stubblefield, 1990
Pteronarcys dorsata	Stonefly	20-40 mm	28	F,M,T	Mortality	45	238	–	NOEC	238	.36	–	Spehar and others, 1978a
Ptychocheilus lucius	Colorado pikeminnow	Larva	4	S,U	Mortality	199	–	–	LC50	78	2.30	No analytical confirmation of exposure concentrations.	Buhl, 1997

Table 17. Other data on effects of cadmium to freshwater organisms.—Continued

Species	Common name	Size or age at test initiation	Exposure duration (days)	Method	Effect	Hardness (mg/L as $CaCO_3$)	NOEC (µg/L)	LOEC (µg/L)	Summary statistic	Summary effect value (µg/L)	Corresponding acute or chronic criterion value (µg/L)	Notes	Reference
Ptychocheilus lucius	Colorado pikeminnow	Juvenile	4	S,U	Mortality	199	–	–	LC50	108	2.30	No analytical confirmation of exposure concentrations.	Buhl, 1997
Rana sphenocephala	Southern leopord frog	Tadpoles	103	S,M,D	Growth and mortality	60	–	0.5	LOEC	.5	.43	Reduced survival (75 vs. 90 percent for controls) and a non-significant trend of older and larger metamorphs exposed to Cd through periphyton (about 10 mg/kg dry weight and water (5 to <0.1 µg/L). Adverse effects were noted in the lowest cadmium exposure which was dosed with a single slug resulting in an initial nominal aqueous concentration of 5 µg/L. Aqueous concentrations dropped to <0.1 µg/L during the course of the test. Tissue-residue LOEC was 14.0 mg/kg dry weight (whole body).	James and others, 2005
Salmo trutta	Brown trout	Swimup fry	4	F,M,T	Mortality	29	–	–	LC50	1.26	.47	4- and 30-day acute values similar. LC50s recalculated from author's data using EPA Probit program.	Brinkman and Hansen, 2004a
Salmo trutta	Brown trout	Swimup fry	30	F,M,T	Mortality	29	–	–	LC50	1.08	.47	4- and 30-day acute values similar. LC50s recalculated from author's data using EPA Probit program.	Brinkman and Hansen, 2004a
Salmo trutta	Brown trout	Swimup fry	4	F,M,T	Mortality	68	–	–	LC50	3.79	.95	4- and 30-day acute values similar. LC50s recalculated from author's data using EPA Probit program.	Brinkman and Hansen, 2004a
Salmo trutta	Brown trout	Swimup fry	30	F,M,T	Mortality	68	–	–	LC50	3.69	0.95	4- and 30-day acute values similar. LC50s recalculated from author's data using EPA Probit program.	Brinkman and Hansen, 2004a
Salmo trutta	Brown trout	Swimup fry	4	F,M,T	Mortality	151	–	–	LC50	10.26	1.83	4- and 30-day acute values similar. LC50s recalculated from author's data using EPA Probit program.	Brinkman and Hansen, 2004a
Salmo trutta	Brown trout	Swimup fry	30	F,M,T	Mortality	151	–	–	LC50	9.65	1.83	4- and 30-day acute values similar. LC50s recalculated from author's data using EPA Probit program.	Brinkman and Hansen, 2004a
Salvelinus fontinalis	Brook trout	Yearling	4	F,M,T	Mortality	47	–	–	LC50	5,080	.71	Yearlings are apparently an insensitive life stage.	Holcombe and others, 1983

Table 17. Other data on effects of cadmium to freshwater organisms.—Continued

Species	Common name	Size or age at test initiation	Exposure duration (days)	Method	Effect	Hardness (mg/L as $CaCO_3$)	NOEC (µg/L)	LOEC (µg/L)	Summary statistic	Summary effect value (µg/L)	Corresponding acute or chronic criterion value (µg/L)	Notes	Reference
Salvelinus fontinalis	Brook trout	Fry, 0.21 g	4	S,M,T	Mortality	41	–	–	LC50	0.69	0.63	LC50 estimated as the geometric mean of the lowest exposure (0 percent survival at 1.6 µg/L initial measured concentration) and the control treatment (100 percent survival at 0.3 µg/L initial measured concentration). LC50 was not included in table 15 because of data quality uncertainties. Only 2 concentrations were tested (1.5 and 15 µg/L nominal). Cd concentration in control water at 96-hours recorded as at 2.9 µg/L. Authors suggested this may have been an analytical error since 100 percent mortality occurred in the 1.5 µg/L treatment and zero mortaility occurred in controls, however some other treatments also had unexplained increases in Cd during the exposure, raising the possibility of lab contamination. Concentrations in filtered samples were considerably higher than in unfiltered samples, further suggesting sample contamination.	Carroll and others, 1979
Salvelinus fontinalis	Brook trout	Sub-adult, 8 month of	10	R,M,T	Mortality	20	8	18	MATC	12.0	.22	LC50 was 14 µg/L. Data point not used because older fish may be more resistant than younger fish. Nonstandard duration for either acute or chronic exposures.	Jop and others, 1995
Salvelinus namaycush	Lake trout	2-year old	106-112	F,M,T	Behavior	81	–	.5	LOEC	.5	.37	Reduced effectiveness at capturing unexposed fathead minnow prey.	Kislalioglu and others, 1996
Salvelinus namaycush	Lake trout	2-year old	240-277	F,M,T	Behavior	90	–	0.5	LOEC	.5	.55	Reduced effectiveness at capturing similarly exposed rainbow trout fingerlings.	Scherer and others, 1997
Thymallus arcticus	Arctic grayling	Juvenile	4	S,U	Mortality	41	–	–	LC50	4	.63	No analytical confirmation of exposure concentrations.	Buhl and Hamilton, 1991
Xyrauchen texanus	Razorback sucker	Larva	4	S,U	Mortality	199	–	–	LC50	139	2.30	No analytical confirmation of exposure concentrations.	Buhl, 1997
Xyrauchen texanus	Razorback sucker	Juvenile	4	S,U	Mortality	199	–	–	LC50	160	2.30	No analytical confirmation of exposure concentrations.	Buhl, 1997

This page left intentionally blank

Appendix – Final Review

Criteria derivation is a complex exercise that should be carefully reviewed by rechecking each step of the process and the derivation guidelines. Stephan and others (1985, p. 56) identified 17 questions that should be especially addressed. Since their review questions assumed that the chronic criterion would be derived using acute-to-chronic ratios, which was not the case here, review questions paralleling those for the final acute value were added for the development of the final chronic value. The questions and my responses follow:

1. *If unpublished data are used, are they well documented?*

Several unpublished data sources were used. In each case they were available in writing so that the summaries of test values and conditions presented here could be reproduced, or the data summaries presented here were reviewed and confirmed by the original investigators. One unpublished dataset was particularly pertinent (matched acute and chronic tests with rainbow trout by MA. Cairns). M.A. Cairns was located, but no detailed documentation of methods and results could be found. These data are presented in table 17 of the main report (Other data) but not directly used in criteria calculations.

2. *Are all required data available?*

Yes, the 8-family minimum species diversity rules were met for both the acute and chronic datasets.

3. *Is the range of acute values for any species greater than a factor of 10?*

Yes. *Daphnia magna* acute values varied by more than a factor of 100, even after hardness-adjustment. However, except for clearly resistant genotypes, this variability was considered representative of the variability found in field populations. Whether this greater range of values for *Daphnia* results is an artifact of its larger dataset or is unique to Daphnia and cadmium is not known. Available *Daphnia magna* values with copper suggest less variability than with cadmium. Biotic ligand model adjusted acute values varied by a maximum factor of 30, with most values varying by less than a factor of 10 (U.S. Environmental Protection Agency, 2003a)

4. *Is the range of Species Mean Acute Values for any genus greater than a factor of 10?*

No, the greatest range (*Daphnia*) was a factor of 5.

5. *Is there more than a factor of 10 difference between the four lowest Genus Mean Acute Values?*

No, differences were less than a factor of 2.

6. *Are any of the four lowest Genus Mean Acute Values questionable?*

Not in the final iteration. Earlier, one of the lowest GMAVs (Striped bass, *Morone saxatilis*, now listed in table 17) was based on unmeasured exposures. Since this seemed questionable, and no obvious basis for selectively removing values reported from unmeasured exposures, all test values based upon unmeasured exposures were deleted from the acute and chronic datasets.

7. *Is the Final Acute Value reasonable in comparison with the Species Mean Acute Values and Genus Mean Acute Values?*

Yes, when plotted as a species-sensitivity distribution and fit to a logistic curve (not shown), the data were evenly distributed. The 5th percentile FAV (2.41 µg/L) fell between the GMAVs with cumulative probabilities of the 4th and 6th percentiles (2.1 and 2.6 µg/L respectively).

8. *For any "important" or "critical" species[3], is the geometric mean of the acute values from tests in which the concentrations of test material were measured lower than the Final Acute Value[4]?*

Yes, species mean acute values for cutthroat trout, rainbow trout, and bull trout (all based on measured values) were all lower than the calculated FAV. The FAV was lowered to the lowest of these SMAVs.

9. *Are any of the chronic values questionable?*

Of the values retained for use, none were obviously suspect. Although few if any of the chronic test values could be considered ideal and the results often raised questions, none of the table 16 values seemed so "questionable' as to be discarded. Assigning values to tests with unbounded low effects were particularly troublesome (i.e., adverse effects were observed at the lowest exposure). Taking a geometric mean of the low-effect concentration and the control was

[3] As described in methods under "Extrapolating small toxicity test datasets to aquatic communities," Stephan and others (1985) limited their consideration of "important" species in this context to commercially or recreationally important species. However, in the context of site-specific criteria, Stephan and others (1994) defined the term "critical" species more broadly and that broader definition seems more appropriate here.

[4] Stephan and others (1985, p. 56) limited this question to "acute values from flow-through tests in which concentrations were measured" however since renewal and static tests in which concentrations were measured were also used to develop the Final Acute Value this question was modified (see the section "Flow through, renewal, and static test exposures").

usually unsatisfactory because the control was often a "less than" value too. In these situations, chronic values were selected on a case-by-case basis considering the magnitude of effect and other values for the species if any.

10. Is the range of chronic values for any species greater than a factor of 10?

As with acute data, *Daphnia magna* values were highly variable. Chronic values varied by over a factor of 20. As with the acute data, this variability is assumed to at least partly reflect that found in field populations. Whether this high variability in the sensitivity of *Daphnia magna* to cadmium reflects inherent differences from that of other species or chemicals or if the difference is an artifact of the intensive studies with *Daphnia magna* and cadmium is unknown. In EPA's (2003a) compilation of copper results, chronic values for Daphnia magna only varied by about a factor of three. No values were excluded solely due to differences from other values.

11. Is the range of Species Mean Chronic Values for any genus greater than a factor of 10?

Yes, within the genus Salvelinus the SMCV for lake trout was 10.3X higher than the SMCV for bull trout. Because the difference was greater than a factor of 10, the lake trout SMCV was based upon a single test, and because the source of that test for lake trout data included no information on what percent reductions in survival corresponded to the NOEC or LOEC values, the lake trout value was excluded from the GMCV calculation. Otherwise the greatest range was a factor of 6.6 for *Daphnia*.

12. Is there more than a factor of 10 difference between the four lowest Genus Mean Chronic Values?

No, there was only about a factor of 3.6 difference between the lowest and 4th lowest GMCVs. .

13. Are any of the four lowest Genus Mean Chronic Values questionable?

As with question number 6, not in the final iteration. Earlier, one of the lowest GMCVs (cladoceran, *Moina macrocopa*, was based on unmeasured exposures (now listed in table 17, for "Other data."

14. Are chronic values available for acutely sensitive species?

Yes. Chronic values are available for species within each of each of the four most acutely sensitive genera. No chronic values are available for the most acutely sensitive species, cutthroat trout *Oncorhynchus clarki*. However, chronic data are available for two other species of *Oncorhynchus* and are assumed to be reasonably representative of chronic sensitivity for other members of the genus *Oncorhynchus* (at least more so than values for species from other genera). Cutthroat trout and rainbow trout SMAVs were similar, with overlapping acute values.

15. Is the range of acute-chronic ratios greater than a factor of 10?

Yes. The ACRs ranged from 1.1 to 424. Since the species mean acute-chronic ratios tended to increase with species mean acute values, acute-chronic ratios were calculated for species with species mean acute values that were close to the final acute value. To judge how close "close" should be, species mean acute values that were within ~10X of the final acute value were used. This resulted in a final ACR that was similar to the FAV–FCV ratio. Although presented in the report, acute-chronic ratios were not used in derivation of the chronic cadmium criterion but are presented in the report.

16. Is the Final Chronic Value reasonable in comparison with the available acute and chronic data?

Yes. The final chronic value fell slightly above the most sensitive species with a cumulative probability value of 0.05. The chronic species sensitivity distribution was fairly even and fit well to a curve (fig. 5). A few acute values from the most sensitive genus *Oncorhynchus* are close to the final chronic value and one was lower. However, these were the extremes of the distribution of available values for this genus. Overall, the most sensitive species mean acute value (cutthroat trout) was 3.4X greater than the final chronic value, which is probably enough higher that minimal acute toxicity would result at FCV concentrations.

17. Is the measured or predicted chronic value for any "critical" species below the Final Chronic Value?

No, unless *Hyalella azteca* is considered "critical."

18. Are any of the "other data" important?

Possibly. Two independent tests found that the ability of *Salvelinus* trout to capture prey was impaired at exposures at or below chronic criterion concentrations. However, a long-term whole lake ecosystem experiment studied cadmium exposed *Salvelinus* populations without finding any evidence of reduced body conditions, making the extrapolation of these tests to the wild uncertain. Regardless, these "other data" caution that careful bioassessment of fish health, populations and fish and macroinvertebrate assemblages would be prudent in situations where cadmium concentrations in receiving waters approach criterion concentrations.

19. Do any data look like they might be outliers?

Not in the final compilation of acute and chronic values that seemed acceptable for use (tables 15 and 16). Several values appeared to be outliers and thus were given extra scrutiny. Based on this scrutiny, values from several tests were not used in the FAV or FCV calculations because they appeared to have tested life stages that were not the most sensitive, organisms were pre-exposed to cadmium, or otherwise questionable data (e.g., *Chironomus riparius*, brook trout, or coho salmon, table 17). One value for *Ceriodaphnia dubia* was not used solely because it was considered an outlier (table 16).

20. Are there any deviations from the guidelines? Are they acceptable?

There were three deviations from the Guidelines. (1) The guidelines gave flow-through toxicity tests priority over static tests. Here, flow-through, renewal, and static test results were all used with equal priority assuming that they are otherwise acceptable (for example, used sensitive life stages, provided sensitive results in comparison to other tests within the genus, and concentrations were measured). (2) The guidelines would accept results of tests in which the exposure concentrations were not measured if no better data for a species were available. Here no test results based upon unmeasured concentrations were used since sufficient values with measured concentrations to meet minimum diversity requirements were available. (3) The guidelines gave preference to results of chronic tests conducted over an organism's full life-cycle over shorter-term tests. Here shorter-term tests also were used if results were more sensitive, or similar in sensitivity to results from life-cycle tests. See "Data acceptability review."

These deviations from the guidelines were each supported by a technical rationale. Considering the data available in 2005, but not available in 1984 when the guidelines were written, the deviations seemed more appropriate than did rigid adherence to the guidelines. Still, these deviations from the guidance could be debated; Three peer reviewers commented that they concurred with the deviations; one commented that they did not fully agree with the deviations.

21. On the basis of all available pertinent laboratory and field information, are the criteria consistent with sound scientific evidence? If they are not, other criteria, either higher or lower should be derived using appropriate modifications to criteria derivation Guidelines.

Most data indicate that measurable, adverse biological effects are unlikely in lake or stream ecosystems that only rarely exceed the criteria values calculated here. Behavioral data with *Salvelinus* (lake and brook trout) are ambiguous, with adverse effects demonstrated in laboratory settings but with no obvious adverse effects observed in a long-term whole lake experiment. Because of this uncertainty, careful biomonitoring and assessment of fish health, and benthic macroinvertebrate or zooplankton assemblages in streams or lakes, respectively, could be important if discharges were to cause receiving waters to approach criteria conditions.

Glossary

Acute toxicity Acute toxicity tests are generally used to determine the concentration of test material that produces a specific adverse effect on a specified percentage of test organisms during a short exposure. Because death is an obviously important adverse effect and is easily detected for many species, the most common acute toxicity test is the acute lethality test. Experimentally, effects on 50 percent of a group of test organisms are the most reproducible and easily determined measure of toxicity, and 96 hours is often a convenient, useful exposure duration. Therefore, the measure of acute toxicity most often used with fishes, macroinvertebrates, and amphibians is the 96-hour LC50. However, because immobilization is a severe effect and is not easy to distinguish from death for some species, the measure of acute toxicity most often used with daphnids and midge larvae is the 48-hour EC50 based on death plus immobilization (American Society for Testing and Materials, 1997).

ACR Acute to chronic ratio, used for estimating chronic effect values from acute effects data.

Alevin Newly hatched fish that still contain a yolk sac. Summary of life stage terms: alevins are newly hatched fish that still contain a yolk sac. Swim-up fry are fish that are 4–6 weeks post hatching, have resorbed their yolk sacs and have begun feeding, and have just swum up from their incubation cups or natural substrate to swim swimming freely in the water column. For the salmonids, larger juveniles that may be up to two years old are often referred to as parr. Juvenile salmon and anadromous trout that are physiologically adapted to seawater are called smolts.

BAF Bioaccumulation factor. A BAF is intended to account for net uptake from both food and water in a real-world situation. A BAF almost has to be measured in a field situation under reasonably steady-state conditions in which predators accumulate the material directly from the water and by consuming prey that itself could have accumulated the material from both food and water.

BCF Bioconcentration factor. A BCF is the quotient of the concentration of a material in one or more tissues of an aquatic organism divided by the average concentration in the solution in which the organism had been living. A BCF is intended to account only for net uptake directly from water and thus almost has to determined through laboratory testing.

CCC Criterion continuous concentration, synonymous with “chronic criterion.”

Chronic toxicity As used here, chronic toxicity generally refers to adverse effects to organisms during a long-term exposure to concentrations of test material. Effects that are considered “adverse” include lethal or sublethal effects that can be related to survival and reproduction rates. The duration of “long-term” exposures varies by species and test protocol, but must be longer than that used for acute exposures and long enough to measure sublethal effects such as growth. “Long-term” exposures may be as brief as seven days in short-term tests for estimating chronic toxicity. Unlike “acute toxicity,” the usage of the term “chronic toxicity usage varies in the ecotoxicology literature. See also “Life-cycle versus shorter-term “chronic” data” and “Statistical interpretation of chronic tests.”

CMC Criterion maximum concentration, synonymous with “acute criterion.”

Critical species In the context of deriving site specific water-quality criteria and related assessments that use the SSD concept, “critical species” have been defined to include 1) keystone species that are of great ecological value because their loss would indirectly affect many other species or ecosystem function; 2) species of special conservation status through treaties, laws, or policies as threatened, endangered, or vulnerable; 3) recreationally valued species; or 4) commercially important species (U.S. Environmental Protection Agency, 1994; Posthuma and others, 2002; Stephan and others, 1994b).

Chronic limits “A chronic value may be obtained by calculating the geometric mean of the lower and upper chronic limits from a chronic test or by analyzing chronic data using regression analysis. A lower chronic limit is the highest tested concentration (a) in an acceptable chronic test, (b) which did not cause an unacceptable amount of adverse effect on any specified biological measurements, and (c) below which no tested concentration caused an unacceptable effect. An upper chronic limit is the lowest tested concentration (a) in an acceptable chronic test, (b) which did cause an unacceptable amount of adverse effect on any specified biological measurements, and (c) above which all tested concentration also caused such an effect.” (Stephan and others 1985, p. 39)

Dissolved metals Operationally defined as those metals that pass through a 0.45 µm filter, which includes metals chemically dissolved in water and very small colloidal particles.

DOM Dissolved organic matter.

dw Dry weight.

ECp Concentration causing effects in *p* percentage of the test population, such as the EC50, EC20, EC10, and so on.

ELS Early-life stage toxicity test. Embryos and subsequent larvae of fish species are maintained for 28 to 120 days, depending on species in treatments of water with differing concentrations of test materials added.

EPA U.S. Environmental Protection Agency

FAV Final acute value

FCV Final chronic value

Flow-through toxicity test A test in which the test solution flows through the test chamber on a once-through basis throughout the test. Compare with renewal or static toxicity tests. Usage note: The term "flow-through" could imply a test with flowing water that is constructed to mimic a stream environment. This is not the case; rather the term refers to the once-through, continuous delivery of test solutions (or frequent delivery in designs using a metering system that cycles every few minutes). Flows on the order of about 5-volume replacements per 24 hours are insufficient to cause discernable flow velocities. In this report, tests that are designed to mimic stream environments are called "artificial stream" tests.

Fry Juvenile fish that have resorbed their yolk sac. See listing of fish life stage terms under "alevin."

GMAV Genus mean acute value, calculated as the geometric mean of all SMAVs for that genus.

GMCV Genus mean chronic value, calculated as the geometric mean of all SMCVs for that genus.

ILL Incipient lethal level. The concentration of a chemical that is lethal to 50 percent of the test organisms as a result of exposure for periods sufficiently long that acute lethal action has essentially ceased. Also may be referred to as a time-independent LC50, incipient LC50, or asymptotic LC50 concentration.

IBI Index of biotic integrity. IBIs are empirical additive indexes consisting of various metrics describing an assemblage such species composition, diversity, functional organization, presence or absence of pollution sensitive or tolerant taxa (Karr, 1991).

IDEQ Idaho Department of Environmental Quality.

Instar A stage of an insect or other arthropod between molts.

JGS Juvenile growth and survival test; a chronic test initiated with fish as swim-up fry or with invertebrates such as amphipods shortly after hatching.

LC Life cycle test

LC50 Concentration lethal to 50 percent of the organisms in a toxicity test.

LOEC Lowest-observed-effect concentration, see NOEC.

MATC A maximum acceptable toxicant concentration (MATC) is a hypothetical test concentration which is assumed to be a threshold for toxic effects. Its point estimate is the geometric mean of the NOEC and the LOEC.

NAWQA USGS National Water Quality Assessment Program

NOEC As used in this report, the no-observed-effect concentration (NOEC) from a toxicity test is the highest test concentration that results in responses that are not appreciably different from the control responses. The lowest-observed-effect concentration (LOEC) is the first treatment that is greater than the NOEC. Note: this usage is slightly different from that often used elsewhere where the NOEC is often defined as the concentration that is not statistically different from controls. See text section on text "Interpretation of chronic effects"

Parr Juvenile salmonids that are freshwater adapted and usually between 2-months and 2-years in age. See listing of different fish life stages under "alevin."

Renewal toxicity test The "renewal" technique is like the static technique except that test organisms are periodically exposed to fresh test solution of the same composition, usually once every 24 h or 48 h, by replacing nearly all the test solution.

SMAV Species mean acute value, calculated as the geometric mean of all acceptable acute values for that species, following the adjustment of each acute value to a common hardness value using observed hardness-toxicity relations.

SMCV Species mean chronic value, calculated in the same way as SMAVs.

SSD Species sensitivity distribution, statistical distribution estimated from toxicity data for a dataset of species responses and are considered to be a sample of an entire population (in the statistical sense) of species responses. Note: SSDs should more precisely be called "taxa sensitivity distributions" since they may be developed at the species, genus, family or other taxonomic level. However, the common usage seems to be "SSD" regardless of taxonomic level, and that usage is kept in this report.

Static toxicity test In "static" exposures, test solutions and organisms are placed in chambers and kept there for the duration of the test.

Swim-up fry Fry that have left their nests in or on natural substrates or their artificial incubation cups and are freely swimming in the water column and feeding. See listing of different fish life stages under "alevin."

Total concentration Metals concentration determined from an unfiltered sample.

TRV Tissue reference value. A TRV is considered as a tissue chemical concentration above which adverse effects might occur to aquatic species following chronic exposure and below which it is unlikely that adverse effects will occur.

USGS U.S. Geological Survey

This page left intentionally blank

Manuscript approved for publication, September 21, 2006

Prepared by the USGS Publishing Network,

Publishing Service Center, Tacoma, Washington
Bill Gibbs
Bob Crist
Debra Grillo
Sharon Wahlstrom
Bobbie Jo Richey

For more information concerning the research in this report, contact the
Director, Idaho Water Science Center,
U.S. Geological Survey, 230 Collins Road
Boise, Idaho 83702
http://id.water.usgs.gov

USGS

Mebane

Cadmium Risks to Freshwater Life: Derivation and Validation of Low-Effect Criteria Values using Laboratory and Field Studies

SIR 2006–5245