

different waters of the state which may be deemed by the commission to be relevant insofar as possible pursuant to any federal water pollution control act. These shall be reevaluated and modified as required by any federal water pollution control act.”

§ 644.026(7), RSMo.

4. The Missouri Clean Water Commission is empowered with the authority to “[a]dopt, amend, promulgate, or repeal after due notice and hearing, rules and regulations to enforce, implement, and effectuate the powers and duties of sections 644.006 to 644.141 and any required of this state by any federal water pollution control act, and as the commission may deem necessary to prevent, control and abate existing or potential pollution.” § 644.026(8), RSMo.

5. Section 536.041, RSMo provides that:

[a]ny person may petition an agency requesting the adoption, amendment or repeal of any rule. Any agency receiving such a petition or other request in writing to adopt, amend or repeal any rule shall forthwith furnish a copy thereof to the joint committee on administrative rules and to the commissioner of administration, together with the action, if any, taken or contemplated by the agency as a result of such petition or request, and the agency’s reasons therefor.

Background – Water Quality Standards

6. Section 303(c) of the federal Clean Water Act (CWA), 33 U.S.C.

§ 1313(c), requires States to adopt water quality standards for waters of the United States within their applicable jurisdictions.

7. States are also required to review their water quality standards at least once every three years (“triennial review”) and, if appropriate, to revise or adopt new standards. 33 U.S.C. § 1313(c)(1).

8. Section 303(c) of the CWA, 33 U.S.C. § 1313(c), authorizes Missouri to amend any water quality standards, as appropriate, at any time. Therefore, Missouri may amend its water quality standards sooner than every three years.

9. During the next year, the Missouri Department of Natural Resources is planning to initiate a rulemaking as part of the triennial review process.

10. Section 304(a) of the CWA, 33 U.S.C. § 1314(a), requires EPA to develop and publish water quality criteria that reflect the latest scientific knowledge with respect to specific pollutants.

11. In establishing statewide numeric criteria for toxic pollutants, a state may adopt the same numeric limits as in the criteria developed by EPA pursuant to section 304(a). However, EPA’s recommended water quality criteria do not impose legally-binding requirements. States retain the discretion to adopt, where appropriate, other scientifically defensible water quality criteria that differ from EPA’s 304(a) criteria recommendations. Therefore, States may adjust the national criteria to reflect more recently available and site-specific information.

12. Upon adoption, States incorporate their state-specific criteria into their statewide water quality standards as enforceable ambient water quality criteria.

13. States are required to submit new or amended water quality standards to EPA for approval. 33 U.S.C. § 1313(c)(2)(A).

Proposed Amendment to Lead Water Quality Standard

Protection of Aquatic Life

14. Pursuant to its rulemaking authority set forth in § 644.026.1(8), RSMo, the Missouri Clean Water Commission has promulgated WQS for various pollutants including lead. The lead WQS is set forth in Table A of the WQS. 10 CSR 20-7.031.

15. In 1984, EPA published an updated 304(a) criteria document entitled *Update of Ambient Water Quality Criteria for Lead – 1984*, EPA 4405-84-027 (hereafter, “EPA lead criteria document”).

16. Missouri has lead WQS protection for the protection of aquatic life, drinking water supplies, and groundwater. The limit for protection of aquatic life is defined in a hardness dependent equation. *See* 10 CSR 20-7.031, Table A – Criteria for Designated Uses.

17. Since 1984, additional studies have been conducted on the toxicity of lead to aquatic organisms. These studies offer valuable information to provide the basis to revise ambient water quality criteria to ensure adequate protection consistent with the best available scientific information.

Criteria Development Procedures

18. EPA guidelines for deriving numeric aquatic life criteria (EPA, 1985b) require toxicity data for at least one species of freshwater animal in at least eight different families. In general, acute criteria are determined by calculating species mean acute values (SMAVs) and genus mean acute values (GMAVs), selecting the four most

sensitive genera, and applying a statistical equation to calculate a Final Acute Value that is intended to protect 95% of a group of diverse genera (EPA, 1985b).

19. EPA guidelines for deriving numeric aquatic life criteria (EPA, 1985b) require toxicity data for at least one species of freshwater animal in at least eight different families. In general, chronic criteria are determined using the same approach used for acute criteria, or, if insufficient chronic toxicity data are available, by applying appropriate acute-to-chronic ratios (ACRs) to the final acute values calculated from the GMAVs described above.

20. For many metals, including lead, toxicity varies with hardness, and criteria are expressed as an equation of the form:

$$\text{criterion} = \exp(a * \ln(\text{Hardness}) + b)$$

where *a* and *b* are determined from a hardness-toxicity regression. To determine the equation, the toxicity data are normalized for hardness, and a final acute value is determined as described above. A least-squares regression of the acute toxicity values on the corresponding hardness values is performed (EPA, 1985b) to determine the values of *a* and *b* in the equation.

21. Chronic criteria may be calculated using a similar approach, if sufficient chronic toxicity data are available. If sufficient data are not available, then an acute-to-chronic ratio (ACR) is applied to the acute criteria equation to determine the chronic criteria.

Aquatic Toxicity Data

22. The 1984 EPA criteria were based on acute toxicity data for only 10 species, and a hardness-based regression based on data for only three species (EPA, 1985a). Today, considerably more data are available. The International Lead Zinc Research Organization (ILZRO) funded research to support the development of biotic ligand model-based ambient water quality criteria for lead. The first phase of this work has resulted in the compilation of acute and chronic lead toxicity data that can be used to calculate revised lead criteria, as described below.

Acute Toxicity

23. The ILZRO research compiled data from EPA's previous criteria materials, more recent studies published in the scientific literature, and several reports developed for ILZRO, resulting in acceptable acute toxicity data for 36 species and 31 genera (WindWard, 2012).

24. The 1984 criteria were based on hardness-toxicity regressions for three species (EPA, 1985a). ILZRO compiled data for seven species that included tests at a range of hardness values, reported quantifiable toxicity results, and are suitable for use in hardness-toxicity regressions. These seven species were used to develop a revised hardness-toxicity relationship.

25. Following EPA procedures (1985b), hardness-toxicity regressions were developed for each of the seven species and the four lowest GMAVs were used to

calculate the Final Acute Value using the statistical procedures described in the EPA (1985b) guidelines.

26. These calculations yielded a revised acute criterion, normalized to a hardness of 50 mg/l. The slope for the hardness-toxicity regression was then used to determine the hardness-based equation to yield that revised criterion at a hardness of 50 mg/l. The resulting acute equation is:

$$\text{Acute criterion} = \exp(0.5085 * \ln(\text{Hardness}) + 2.9581)$$

This equation results in higher acute criteria at lower hardness values, and lower criteria at high hardness, as compared to current Missouri acute criteria.

Chronic Toxicity

27. The ILZRO research has provided substantially more data than were used in the 1984 criteria, including additional acute-to-chronic ratios (ACRs) that can be used to calculate revised chronic criteria based on the acute toxicity criteria presented above.

28. ACRs were available for seven species. Consistent with EPA procedures (1985b), only ACRs for species whose SMAVs are close to the final acute value were used to determine the final ACR of 9.01.

29. The final acute value at the normalized hardness (50 mg/l) was divided by the final ACR of 9.01 to determine the final chronic value at that hardness. The chronic equation was then derived using the hardness-toxicity slope, as described in the EPA guidance (1985b).

30. The resulting chronic equation is:

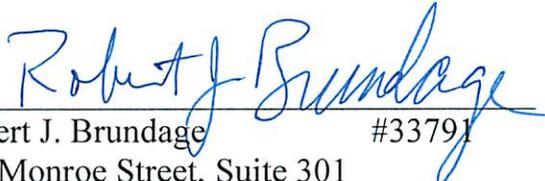
$$\text{Chronic criterion} = \exp(0.5085 * \ln(\text{Hardness}) + 0.7597)$$

31. AIM attaches hereto and incorporates by reference a December 6, 2012 memorandum authored by LimnoTech titled *Recommended Revised Water Quality Criteria for Lead* describing in more detail information supporting the proposed changes to the lead criteria requested by AIM.

WHEREFORE, Associated Industries of Missouri respectfully requests the Missouri Clean Water Commission direct the Missouri Department of Natural Resources' staff to commence a rulemaking to amend the lead WQS consistent with the recommendation set forth in this petition and to furnish a copy thereof to the Joint Committee on Administrative Rules and to the Commissioner of Administration, together with the Commission's order to the staff directing action consistent with this Petition including reasons for such action.

Respectfully submitted,

NEWMAN, COMLEY & RUTH P.C.

By: 
Robert J. Brundage #33791
601 Monroe Street, Suite 301
P.O. Box 537
Jefferson City, MO 65102-0537
(573) 634-2266
(573) 636-3306 FAX
rbrundage@ncrpc.com

Attorneys for Petitioner
Associated Industries of Missouri

FROM: Kathy Hall
Hans Holmberg

DATE: December 6, 2012

TO: Robert Brundage, NCR

SUBJECT: Recommended Revised Water Quality Criteria for Lead

The U.S. EPA's National Recommended Water Quality Criteria are published pursuant to Section 304(a) of the Clean Water Act (CWA) and provide guidance for states and tribes to use in adopting water quality standards. EPA's recommended aquatic life criteria for lead were developed in 1984 (EPA, 1985a) and have not been updated since. Additional studies on the toxicity of lead to aquatic organisms have been conducted since the development of the current criteria; these can be used to revise ambient water quality criteria to ensure adequate protection consistent with the best available scientific information. This memorandum describes procedures for criteria development, summarizes available toxicity data, and presents recommended revised lead criteria based on the most recent available toxicity literature. We recommend these revised criteria be adopted statewide.

Criteria Development Procedures

EPA guidelines for deriving numeric aquatic life criteria (EPA, 1985b) require toxicity data for at least one species of freshwater animal in at least eight different families. In general, acute criteria are determined by calculating species mean acute values (SMAVs) and genus mean acute values (GMAVs), selecting the four most sensitive genera, and applying a statistical equation to calculate a criterion that is intended to protect 95% of a group of diverse genera (EPA, 1985b).

For many metals, including lead, toxicity varies with hardness, and criteria are expressed as an equation of the form:

$$\text{criterion} = \exp(a * \ln(\text{Hardness}) + b)$$

where "a" and "b" are determined from a hardness-toxicity regression. To determine the equation, the toxicity data are normalized for hardness, and a final acute value is determined as described above. A least-squares regression of the acute toxicity values on the corresponding hardness values is performed (EPA, 1985b) to determine the values of "a" and "b" in the equation. Chronic criteria may be calculated using a similar approach, if sufficient chronic toxicity data are available. If sufficient data are not available, then an acute-to-chronic ratio (ACR) is applied to the acute criteria equation to determine the chronic criteria.

Aquatic Toxicity Data

The 1984 EPA criteria were based on acute toxicity data for only 10 species, and a hardness-based regression based on data for only three species (EPA, 1985a); considerably more data are currently available. The International Lead Zinc Research Organization (ILZRO) is funding

research to support the development of biotic ligand model-based ambient water quality criteria for lead. The first phase of this work (WindWard, 2012) has resulted in the compilation of acute and chronic lead toxicity data that can be used to calculate revised lead criteria, as described below.

Acute Toxicity

The ILZRO research compiled data from EPA's previous criteria materials, more recent studies published in the scientific literature, and several reports developed for ILZRO, resulting in acceptable acute toxicity data for 36 species and 31 genera (WindWard, 2012). Attachment 1 summarizes these data. The available data indicate that toxicity is related to hardness. The 1984 criteria were based on hardness-toxicity regressions for three species (EPA, 1985a). The data shown in Attachment 1 include data for seven species that included tests at a range of hardness values, reported quantifiable toxicity results, and are suitable for use in hardness-toxicity regressions. These seven species (*Baetis tricaudatus*, *Ceriodaphnia dubia*, *Daphnia magna*, *Gyraulus sp.*, *Oncorhynchus clarki lewisi*, *Oncorhynchus mykiss*, *Pimephales promelas*) were used to develop a revised hardness-toxicity relationship.

Following EPA procedures (1985b), hardness-toxicity regressions were developed for each of the seven species. These regressions were used to adjust all toxicity values to 50 mg/l hardness. Data for each species were examined for potential outliers using normal probability plots; none were identified. The data for each species were normalized by geometric mean acute value and geometric mean, and individual species regressions were performed on the normalized values, consistent with the EPA (1985b) procedures. The resulting slopes for each species were compared and determined to be sufficiently similar to allow the use of a pooled data set. The pooled data for the seven species was then used in the final regression. Figure 1 shows the resulting hardness-toxicity regression, which has a slope of 0.5085.

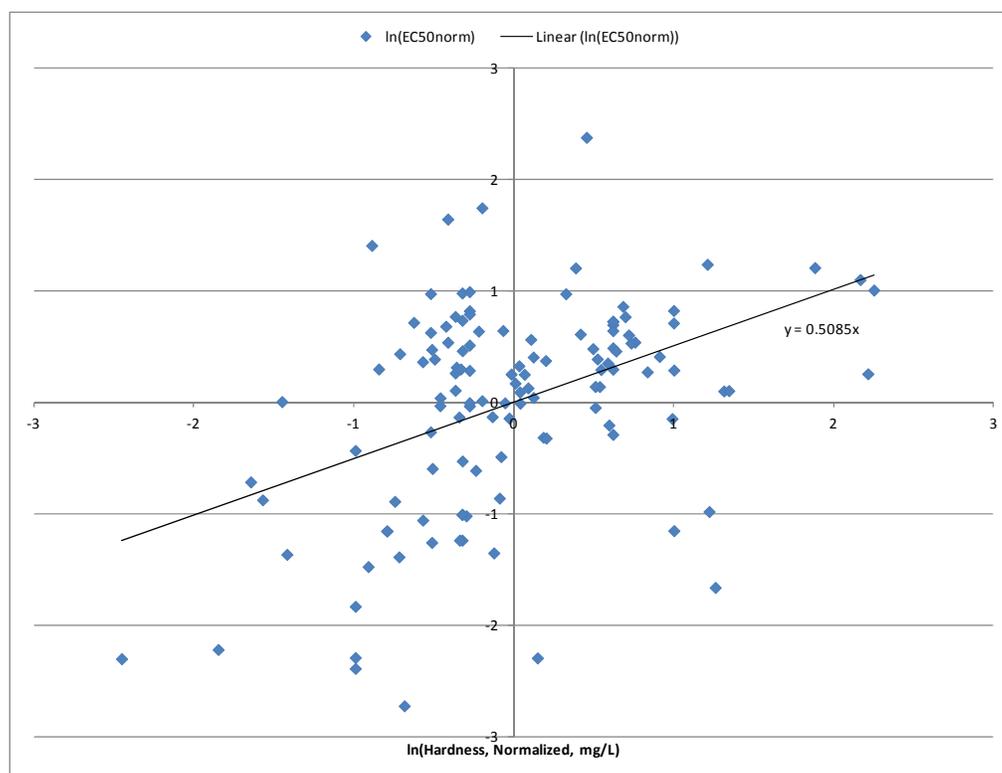


Figure 1. Hardness-Toxicity Regression

This revised regression was used to normalize the available acute toxicity data based on hardness (acute values were normalized to a hardness of 50 mg/l). The species mean acute value (SMAV) and genus mean acute value (GMAV) were calculated for each species and genus. The GMAVs were ranked, the cumulative probabilities were calculated, and the four lowest GMAVs were used to calculate the Final Acute Value, using the statistical procedures described in the EPA (1985b) guidelines. The four most sensitive genera were *Hyaella*, *Gammarus*, *Lampsilis*, and *Ceriodaphnia*. Note that the reported acute value for *Hyaella* was reported as “less than” (“<151 ug/l” before the hardness normalization); because *Hyaella* was the most sensitive organism in the database, it was included in the calculation, with the acute value assumed as the hardness-normalized “less than” value of 126 ug/l. The values used in the derivation of the proposed acute criterion are shown in Table 1. These calculations yielded a revised acute criterion, normalized to a hardness of 50 mg/l. The slope for the revised hardness-toxicity regression was then used to determine the hardness-based criteria equation to yield that acute criterion at 50 mg/l.

Table 1. Derivation of the proposed acute lead criterion

Genus	GMAV	Rank	Cumulative probability (P)	ln(GMAV)	ln(GMAV) ²	sqrt(P)
<i>Hyaella</i>	126.0	1	0.0313	4.6634	21.7477	0.1768
<i>Gammarus</i>	136.5	2	0.0625	4.9165	24.1724	0.2500
<i>Lampsilis</i>	212.5	3	0.0938	5.3589	28.7175	0.3062
<i>Ceriodaphnia</i>	247.5	4	0.1250	5.5113	30.3748	0.3536
n=	31	Sum	0.3125	20.4502	105.0124	1.0865

The resulting acute equation is:

$$\text{Acute criterion} = \exp(0.5085 * \ln(\text{Hardness}) + 2.9581)$$

This equation results in higher acute criteria at lower hardness values, and lower criteria at high hardness, as shown in Figure 2.

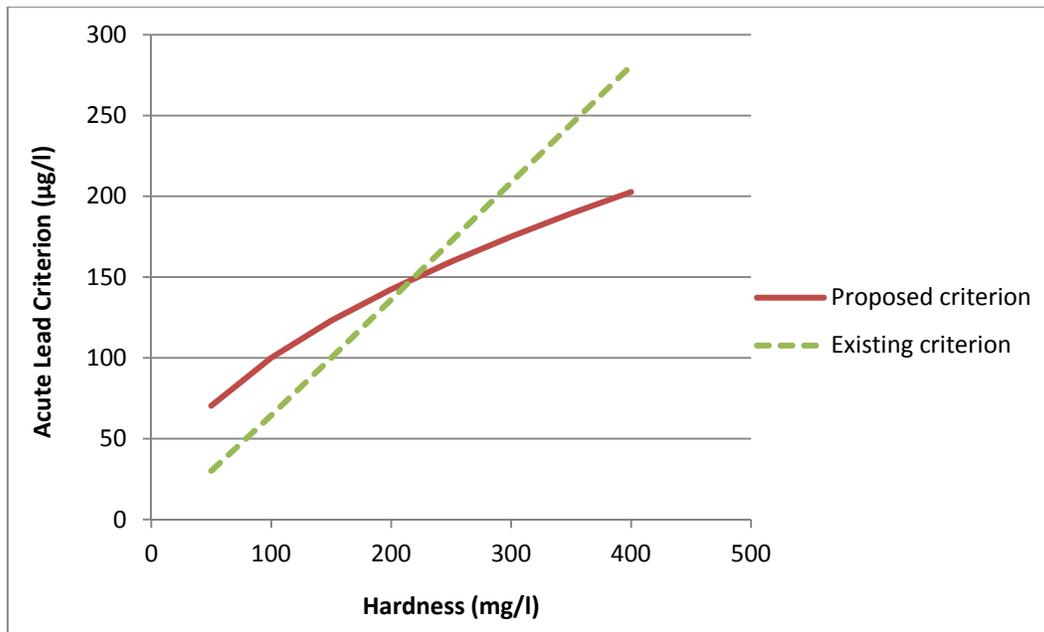


Figure 2. Comparison of Proposed and Existing Acute Criteria for Lead

Chronic Toxicity

The ILZRO research used the same data sources to compile toxicity data for chronic toxicity. These data (WindWard, 2012) do not meet EPA's minimum phylogenetic diversity requirements for direct development of chronic toxicity criteria. However, substantially more data are available than were used in the 1984 criteria, including additional acute-to-chronic ratios (ACRs) that can be used to calculate revised chronic criteria based on the acute toxicity criteria presented above.

ACRs were available for seven species, as summarized in Attachment 2. Figure 3 shows the relationship between the species mean acute value (SMAV) and the species mean acute-to-chronic ratio (SMACR). Consistent with EPA procedures (1985b), only ACRs for species whose SMAVs are close to the final acute value were used to determine the final ACR. The final acute value at 50 mg/l hardness is 140 ug/l, so the calculation of the final ACR was based on the SMACRs for *Ceriodaphnia dubia*, *Daphnia magna*, *Lampsilis siliquoidea*, and *Oncorhynchus mykiss*, which had SMAVs ranging from 217 ug/l to 416 ug/l. The resulting final ACR was 9.01.

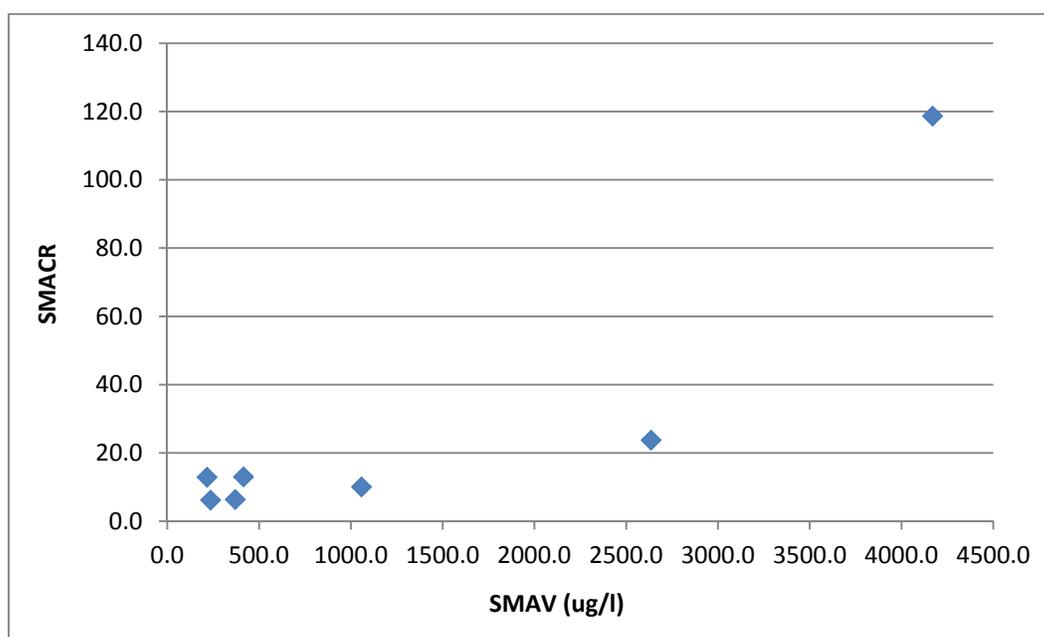


Figure 3. Relationship Between Species Mean Acute Values and Species Mean Acute-to-Chronic Ratios

The final acute value at the normalized hardness (50 mg/l) was divided by the final ACR of 9.01 to determine the final chronic value at that hardness. The chronic equation was then derived using the hardness-toxicity slope, as described in the EPA guidance (1985b).

The resulting chronic equation is:

$$\text{Chronic criterion} = \exp(0.5085 * \ln(\text{Hardness})) + 0.7597$$

This equation results in noticeably different chronic criteria, as shown in Figure 4.

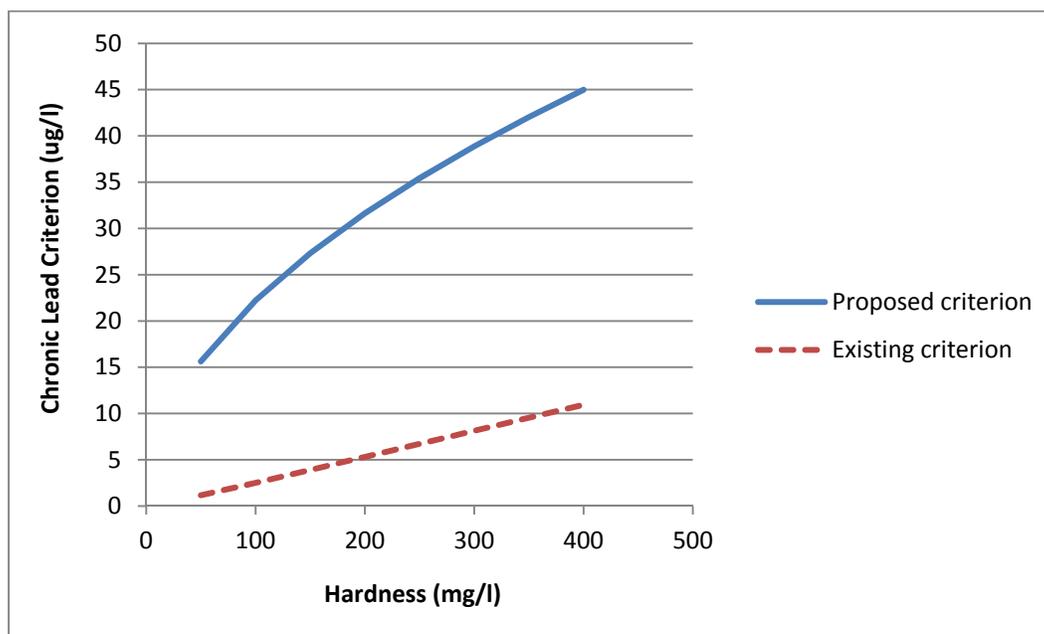


Figure 4. Comparison of Proposed and Existing Chronic Criteria for Lead

References

U.S. Environmental Protection Agency (EPA), 1985a. *Ambient Water Quality Criteria for Lead – 1984*. EPA 440/5-84-027. Office of Water Regulations and Standards, Criteria and Standards Division, Washington, D.C. January 1985.

U.S. Environmental Protection Agency (EPA), 1985b. *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses*. Office of Research and Development, Environmental Research Laboratories; Duluth, Minnesota; Narragansett, Rhode Island; Corvallis, Oregon.

<http://water.epa.gov/scitech/swguidance/standards/criteria/aqlife/upload/85guidelines.pdf>

WindWard Environmental, LLC, 2012. *Phase 1 of BLM-based AWQC Project for Pb: Update of Acute and Chronic Toxicity Tables*. Submitted to International Lead Zinc Research Organization (ILZRO). September 6, 2012.

Attachment 1
Acute Toxicity Data (WindWard, 2012)

Species	Hardness (mg/l)	Result Reported as Total (T) or Dissolved (D)	Reported EC50 (µg/L)	EC50, Diss. ¹ (µg/L)	Dissolved EC50, Adjusted to 50 mg/l Hardness ² (µg/L)	Reference
<i>Aplexa hypnorum</i> (snail)	60.9	T	1340	1340	1212	Call et al. 1981
<i>Arctopsyche</i> sp. (caddisfly)	22	D	>1255	>1255	>1905	Mebane et al. 2012
<i>Baetis tricaudatus</i> (mayfly)	15	D	592	592	1092	Mebane et al. 2012
<i>Baetis tricaudatus</i> (mayfly)	18	D	752	752	1264	Mebane et al. 2012
<i>Baetis tricaudatus</i> (mayfly)	20	D	664	664	1058	Mebane et al. 2008, 2012
<i>Baetis tricaudatus</i> (mayfly)	22	D	426	426	647	Mebane et al. 2012
<i>Baetis tricaudatus</i> (mayfly)	39	D	1002	1002	1137	Mebane et al. 2012
<i>Baetis tricaudatus</i> (mayfly)	67	D	>952	>952	>820	Mebane et al. 2012
<i>Baetis tricaudatus</i> (mayfly)	84	D	>683	>683	>525	Mebane et al. 2012
<i>Baetis tricaudatus</i> (mayfly)	17	D	>494	>494	>855	Mebane et al. 2012
<i>Baetis tricaudatus</i> (mayfly)	11	D	322	322	695	Mebane et al. 2012
<i>Baetis tricaudatus</i> (mayfly)	13	D	511	511	1014	Mebane et al. 2012
<i>Baetis tricaudatus</i> (mayfly)	19	D	640	640	1047	Mebane et al. 2012
<i>Baetis tricaudatus</i> (mayfly)	33	D	>952	>952	>1176	Mebane et al. 2012
<i>Baetis tricaudatus</i> (mayfly)	41	D	<1250	<1250	<1383	Mebane et al. 2012
<i>Ceriodaphnia dubia</i> (cladoceran)	25	D	29.1	29.1	41	Diamond et al. 1997
<i>Ceriodaphnia dubia</i> (cladoceran)	25	D	187	187	266	Diamond et al. 1997
<i>Ceriodaphnia dubia</i> (cladoceran)	25	D	46.1	46.1	66	Diamond et al. 1997
<i>Ceriodaphnia dubia</i> (cladoceran)	25	D	26.4	26.4	38	Diamond et al. 1997
<i>Ceriodaphnia dubia</i> (cladoceran)	69.5	T	400	400	338	Tsui et al. 2005
<i>Ceriodaphnia dubia</i> (cladoceran)	82.4	T	255	209	162	Cooper et al. 2009
<i>Ceriodaphnia dubia</i> (cladoceran)	40	D	540	540	605	Parametrix 2010a
<i>Ceriodaphnia dubia</i> (cladoceran)	135	D	622	622	375	Parametrix 2010a
<i>Ceriodaphnia dubia</i> (cladoceran)	155	D	379	379	213	Parametrix 2010a
<i>Ceriodaphnia dubia</i> (cladoceran)	272	D	>2033	>2033	>859	Parametrix 2010a
<i>Ceriodaphnia dubia</i> (cladoceran)	51	D	656	656	649	Parametrix 2010a
<i>Ceriodaphnia dubia</i> (cladoceran)	51	D	279	279	276	Parametrix 2010a
<i>Ceriodaphnia dubia</i> (cladoceran)	51	T	287	287	284	Parametrix 2010a
<i>Ceriodaphnia dubia</i> (cladoceran)	51	T	1614	482	477	Parametrix 2010a
<i>Ceriodaphnia dubia</i> (cladoceran)	50	D	104	104	104	Parametrix 2010a
<i>Ceriodaphnia dubia</i> (cladoceran)	38.1	D	100	100	115	Esbaugh et al. 2011
<i>Ceriodaphnia dubia</i> (cladoceran)	167.2	D	435	435	235	Esbaugh et al. 2011
<i>Ceriodaphnia dubia</i> (cladoceran)	225.5	D	996	996	463	Esbaugh et al. 2011
<i>Ceriodaphnia dubia</i> (cladoceran)	27.7	D	1180	1180	1593	Esbaugh et al. 2011
<i>Ceriodaphnia dubia</i> (cladoceran)	15.8	D	290	290	521	Esbaugh et al. 2011
<i>Ceriodaphnia dubia</i> (cladoceran)	228.1	D	108	108	50	Esbaugh et al. 2011
<i>Ceriodaphnia dubia</i> (cladoceran)	16.3	D	73.5	73.5	130	Esbaugh et al. 2011
<i>Ceriodaphnia dubia</i> (cladoceran)	5.8	D	28.8	28.8	86	Esbaugh et al. 2011
<i>Ceriodaphnia dubia</i> (cladoceran)	47.0	D	395	395	408	Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	116	D	387	387	252	Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	76.0	D	433	433	350	Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	125	D	597	597	375	Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	183	D	385	385	199	Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	250	D	319	319	141	Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	41.0	D	425	425	470	Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	54.0	D	546	546	525	Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	36.0	D	591	591	698	Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	102	D	532	532	370	Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	99.0	D	964	964	681	Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	106	D	3116	3116	2126	Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	51.0	D	384	384	380	Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	51.0	D	779	779	771	Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	44.0	D	571	571	609	Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	40.0	D	765	765	857	Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	33.0	D	446	446	551	Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	13	D	141	141	280	AquaTox 2012
<i>Ceriodaphnia dubia</i> (cladoceran)	14	D	120	120	229	AquaTox 2012
<i>Ceriodaphnia dubia</i> (cladoceran)	78	D	29	29	23	AquaTox 2012
<i>Ceriodaphnia dubia</i> (cladoceran)	48.3	D	389	389	396	De Schampelaere 2012
<i>Ceriodaphnia dubia</i> (cladoceran)	115	D	332	332	217	De Schampelaere 2012
<i>Ceriodaphnia dubia</i> (cladoceran)	183	D	91.0	91.0	47	De Schampelaere 2012
<i>Ceriodaphnia dubia</i> (cladoceran)	237	D	54.6	54.6	25	De Schampelaere 2012
<i>Ceriodaphnia dubia</i> (cladoceran)	125	D	579	579	363	De Schampelaere 2012
<i>Ceriodaphnia dubia</i> (cladoceran)	125	D	470	470	295	De Schampelaere 2012
<i>Ceriodaphnia dubia</i> (cladoceran)	125	D	549	549	345	De Schampelaere 2012
<i>Ceriodaphnia dubia</i> (cladoceran)	125	D	596	596	374	De Schampelaere 2012
<i>Ceriodaphnia dubia</i> (cladoceran)	125	D	388	388	243	De Schampelaere 2012
<i>Ceriodaphnia dubia</i> (cladoceran)	122	D	235	235	149	De Schampelaere 2012
<i>Ceriodaphnia dubia</i> (cladoceran)	125	D	216	216	136	De Schampelaere 2012
<i>Ceriodaphnia dubia</i> (cladoceran)	121	D	410	410	262	De Schampelaere 2012
<i>Ceriodaphnia dubia</i> (cladoceran)	40	D	221	221	248	De Schampelaere 2012

Species	Hardness (mg/l)	Result Reported as Total (T) or Dissolved (D)	Reported EC50 (µg/L)	EC50, Diss. ¹ (µg/L)	Dissolved EC50, Adjusted to 50 mg/l Hardness ² (µg/L)	Reference
<i>Ceriodaphnia dubia</i> (cladoceran)	112	D	275	275	182	De Schampelaere 2012
<i>Ceriodaphnia dubia</i> (cladoceran)	181	D	249	249	129	De Schampelaere 2012
<i>Ceriodaphnia dubia</i> (cladoceran)	258	D	320	320	139	De Schampelaere 2012
<i>Chironomus dilutus</i> (midge)	32	D	1955	1955	2453	Mebane et al. 2012
<i>Chironomus dilutus</i> (midge)	32	D	3617	3617	4538	Mebane et al. 2008, 2012
<i>Cottus confusus</i> (shorthead sculpin)	21	D	>855	>855	1329	Mebane et al. 2012
<i>Daphnia magna</i> (cladoceran)	54	T	612	612	589	Chapman et al. Manuscript
<i>Daphnia magna</i> (cladoceran)	110	T	952	554	371	Chapman et al. Manuscript
<i>Daphnia magna</i> (cladoceran)	152	T	1910	562	319	Chapman et al. Manuscript
<i>Daphnia magna</i> (cladoceran)	44	T	95	95	101	Yim et al. 2006
<i>Daphnia magna</i> (cladoceran)	150	T	894	439	251	Yim et al. 2006
<i>Drunella</i> sp. (mayfly)	19.5	D	>267	>267	>431	Mebane et al. 2012
Dytiscidae (Coleoptera)	39	D	>1035	>1035	>1174	Mebane et al. 2012
<i>Epeorus</i> sp. (mayfly)	17	D	>494	>494	>855	Mebane et al. 2012
<i>Epeorus</i> sp. (mayfly)	11	D	>346	>346	>747	Mebane et al. 2012
<i>Gammarus pseudolimnaeus</i> (amphipod)	48.3	T	140	140	142	Call et al. 1983
<i>Gammarus pseudolimnaeus</i> (amphipod)	45	T	124	124	131	Spehar et al. 1978
<i>Gila elegans</i> (bonytail chub)	199	T	>170000	>785	>389	Buhl 1997
<i>Gyraulus</i> sp. (snail)	18	D	544	544	915	Mebane et al. 2012
<i>Gyraulus</i> sp. (snail)	20	D	537	537	856	Mebane et al. 2012
<i>Gyraulus</i> sp. (snail)	19	D	380	380	622	Mebane et al. 2012
<i>Gyraulus</i> sp. (snail)	22	D	796	796	1208	Mebane et al. 2012
<i>Gyraulus</i> sp. (snail)	39	D	981	981	1113	Mebane et al. 2012
<i>Gyraulus</i> sp. (snail)	67	D	>952	>952	>820	Mebane et al. 2012
<i>Gyraulus</i> sp. (snail)	84	D	>683	>683	>525	Mebane et al. 2012
<i>Gyraulus</i> sp. (snail)	13	D	644	644	1278	Mebane et al. 2012
<i>Gyraulus</i> sp. (snail)	19	D	>1035	>1035	>1693	Mebane et al. 2012
<i>Gyraulus</i> sp. (snail)	33	D	>952	>952	>1176	Mebane et al. 2012
<i>Gyraulus</i> sp. (snail)	41	D	>683	>683	>756	Mebane et al. 2012
<i>Hyalella azteca</i> (amphipod)	71	D	<151	<151	<126	Besser et al. 2005
<i>Lampsilis rafinesqueana</i> (neosho mucket)	42	D	188	188	205	Wang et al. 2010
<i>Lampsilis siliquoidea</i> (fatmucket)	47	D	>299	>299	>309	Wang et al. 2010
<i>Lampsilis siliquoidea</i> (fatmucket)	41	D	142	142	157	Wang et al. 2010
<i>Lampsilis siliquoidea</i> (fatmucket)	47	D	298	298	308	Wang et al. 2010
<i>Lampsilis siliquoidea</i> (fatmucket)	40	D	>426	>426	>477	Wang et al. 2010
<i>Lecane hamata</i> (rotifer)	85	T	680	503	384	Pérez-Legaspi and Rico-Martínez 2001
<i>Lecane luna</i> (rotifer)	85	T	140	140	107	Pérez-Legaspi and Rico-Martínez 2001
<i>Lecane quadridentata</i> (rotifer)	85	T	3700	616	470	Pérez-Legaspi and Rico-Martínez 2001
<i>Lumbriculus variegatus</i> (oligochaete)	290	T	>8000	>8000	>3273	Schubauer-Berigan et al. 1993
<i>Lumbriculus variegatus</i> (oligochaete)	290	T	>8000	>2307	>944	Schubauer-Berigan et al. 1993
<i>Lumbriculus variegatus</i> (oligochaete)	290	T	>8000	>494	>202	Schubauer-Berigan et al. 1993
<i>Oncorhynchus clarki lewisi</i> (westslope)	32	D	>123	>123	>154	Mebane et al. 2012
<i>Oncorhynchus clarki lewisi</i> (westslope)	31.5	D	>54	>54	>68	Mebane et al. 2012
<i>Oncorhynchus clarki lewisi</i> (westslope)	32	D	215	215	270	Mebane et al. 2012
<i>Oncorhynchus clarki lewisi</i> (westslope)	30.5	D	>72	>72	>93	Mebane et al. 2012
<i>Oncorhynchus clarki lewisi</i> (westslope)	31.5	D	362	362	458	Mebane et al. 2012
<i>Oncorhynchus clarki lewisi</i> (westslope)	56	D	487	487	460	Mebane et al. 2012
<i>Oncorhynchus clarki lewisi</i> (westslope)	68	D	>414	>414	>354	Mebane et al. 2012
<i>Oncorhynchus clarki lewisi</i> (westslope)	72.5	D	>409	>409	>339	Mebane et al. 2012
<i>Oncorhynchus clarki lewisi</i> (westslope)	62.5	D	>153	>153	>137	Mebane et al. 2012
<i>Oncorhynchus clarki lewisi</i> (westslope)	62.5	D	>197	>197	>176	Mebane et al. 2012
<i>Oncorhynchus clarki lewisi</i> (westslope)	11.4	D	47	47	100	Mebane et al. 2012
<i>Oncorhynchus clarki lewisi</i> (westslope)	40.5	D	>387	>387	>431	Mebane et al. 2012
<i>Oncorhynchus clarki lewisi</i> (westslope)	21	D	>855	>855	>1329	Mebane et al. 2012
<i>Oncorhynchus kisutch</i> (coho salmon)	41.3	T	7000	725	799	Buhl and Hamilton 1990
<i>Oncorhynchus kisutch</i> (coho salmon)	41.3	T	21700	810	893	Buhl and Hamilton 1990
<i>Oncorhynchus kisutch</i> (coho salmon)	41.3	T	4180	686	756	Buhl and Hamilton 1990
<i>Oncorhynchus kisutch</i> (coho salmon)	41.3	T	>18000	>796	>877	Buhl and Hamilton 1990
<i>Oncorhynchus mykiss</i> (rainbow trout)	41.3	T	30000	835	920	Buhl and Hamilton 1990
<i>Oncorhynchus mykiss</i> (rainbow trout)	41.3	T	<1700	<619	<682	Buhl and Hamilton 1990
<i>Oncorhynchus mykiss</i> (rainbow trout)	290	D	1470	1470	601	Davies et al. 1976
<i>Oncorhynchus mykiss</i> (rainbow trout)	385	D	1320	1320	468	Davies et al. 1976
<i>Oncorhynchus mykiss</i> (rainbow trout)	32	D	1170	1170	1468	Davies et al. 1976
<i>Oncorhynchus mykiss</i> (rainbow trout)	20	D	138	138	220	Mebane et al. 2008, 2012
<i>Oncorhynchus mykiss</i> (rainbow trout)	32	D	127	127	159	Mebane et al. 2008, 2012
<i>Oncorhynchus mykiss</i> (rainbow trout)	32	D	160	160	201	Mebane et al. 2008, 2012
<i>Oncorhynchus mykiss</i> (rainbow trout)	120	D	1000	1000	641	Rogers et al. 2003
<i>Oncorhynchus mykiss</i> (rainbow trout)	120	T	1000	894	573	Sloman et al. 2003
<i>Oncorhynchus mykiss</i> (rainbow trout)	20	D	138	138	220	Mebane et al. 2012
<i>Oncorhynchus mykiss</i> (rainbow trout)	31.5	D	127	127	161	Mebane et al. 2012
<i>Oncorhynchus mykiss</i> (rainbow trout)	32	D	160	160	201	Mebane et al. 2012
<i>Oncorhynchus mykiss</i> (rainbow trout)	19	D	591	591	967	Mebane et al. 2012
<i>Oncorhynchus mykiss</i> (rainbow trout)	25	D	631	631	898	Mebane et al. 2012
<i>Oncorhynchus mykiss</i> (rainbow trout)	32	D	916	916	1149	Mebane et al. 2012

Species	Hardness (mg/l)	Result Reported as Total (T) or Dissolved (D)	Reported EC50 (µg/L)	EC50, Diss. ¹ (µg/L)	Dissolved EC50, Adjusted to 50 mg/l Hardness ² (µg/L)	Reference
<i>Oncorhynchus mykiss</i> (rainbow trout)	33.5	D	969	969	1188	Mebane et al. 2012
<i>Oncorhynchus mykiss</i> (rainbow trout)	28.5	D	>98	>98	>130	Mebane et al. 2012
<i>Oncorhynchus mykiss</i> (rainbow trout, Kootenai)	21	D	180	180	280	Mebane et al. 2012
<i>Paraleptophlebia</i> sp. (mayfly)	11	D	>346	>346	>747	Mebane et al. 2012
<i>Physa</i> sp. (snail)	22	D	1159	1159	1759	Mebane et al. 2012
<i>Pimephales promelas</i> (fathead minnow)	33	D	622	622	768	Esbaugh et al. 2011
<i>Pimephales promelas</i> (fathead minnow)	26	D	3598	3598	5017	Esbaugh et al. 2011
<i>Pimephales promelas</i> (fathead minnow)	16	D	41	41	73	Esbaugh et al. 2011
<i>Pimephales promelas</i> (fathead minnow)	5	D	68	68	219	Esbaugh et al. 2011
<i>Pimephales promelas</i> (fathead minnow)	19	D	178	178	291	Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	32	D	744	744	934	Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	52	D	1015	1015	995	Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	66	D	1068	1068	927	Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	65	D	1148	1148	1005	Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	301	D	1719	1719	690	Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	20	D	608	608	969	Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	21	D	1075	1075	1671	Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	22	D	1356	1356	2059	Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	21	D	3249	3249	5050	Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	22	D	816	816	1239	Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	23	D	996	996	1478	Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	22	D	698	698	1060	Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	23	D	370	370	549	Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	28	D	162	162	218	Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	29	D	265	265	350	Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	30	D	624	624	809	Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	25	D	340	340	484	Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	43.9	T	2100	1662	1776	Spehar and Fiandt 1986
<i>Pimephales promelas</i> (fathead minnow)	290	T	810	810	331	Schubauer-Berigan et al. 1993
<i>Pimephales promelas</i> (fathead minnow)	290	T	>5400	>2277	>931	Schubauer-Berigan et al. 1993
<i>Pimephales promelas</i> (fathead minnow)	290	T	>5400	>473	>193	Schubauer-Berigan et al. 1993
<i>Ptychocheilus lucius</i> (Colorado pikeminnow)	199	T	>170000	>785	>389	Buhl 1997
<i>Ptychocheilus lucius</i> (Colorado pikeminnow)	199	T	>170000	>785	>389	Buhl 1997
<i>Ptychocheilus lucius</i> (Colorado pikeminnow)	199	T	>170000	>785	>389	Buhl 1997
<i>Rhithrogena</i> sp. (mayfly)	15	D	>737	>737	>1359	Mebane et al. 2012
<i>Rhithrogena</i> sp. (mayfly)	18	D	>985	>985	>1656	Mebane et al. 2012
<i>Rhithrogena</i> sp. (mayfly)	19	D	>166	>166	>272	Mebane et al. 2012
<i>Salvelinus fontinalis</i> (brook trout)	44.3	T	4100	2470	2627	Holcombe et al. 1976
<i>Simulium</i> sp. (black fly)	22	D	415	415	630	Mebane et al. 2012
<i>Simulium</i> sp. (black fly)	39	D	961	961	1090	Mebane et al. 2012
<i>Sweltsa</i> sp. (stonefly)	15	D	>737	>737	>1359	Mebane et al. 2012
<i>Sweltsa</i> sp. (stonefly)	19.5	D	253	253	408	Mebane et al. 2012
<i>Sweltsa</i> sp. (stonefly)	17	D	>494	>494	>855	Mebane et al. 2012
<i>Tanytarsus dissimilis</i> (midge)	46	T	224000	202530	211302	Call et al. 1983
<i>Tanytarsini</i> sp. (chironomid)	39	D	>1035	>1035	>1174	Mebane et al. 2012
<i>Tanytarsini</i> sp. (chironomid)	22	D	>1255	>1255	>1905	Mebane et al. 2012
<i>Thymallus arcticus</i> (Arctic grayling)	41.3	T	>36000	>850	>937	Buhl and Hamilton 1990
<i>Thymallus arcticus</i> (Arctic grayling)	41.3	T	>36000	>850	>937	Buhl and Hamilton 1990
<i>Thymallus arcticus</i> (Arctic grayling)	41.3	T	12000	765	843	Buhl and Hamilton 1990
<i>Thymallus arcticus</i> (Arctic grayling)	41.3	T	<320	<320	<353	Buhl and Hamilton 1990
<i>Thymallus arcticus</i> (Arctic grayling)	41.3	T	<1700	<619	<682	Buhl and Hamilton 1990
<i>Thymallus arcticus</i> (Arctic grayling)	41.3	T	<1000	<581	<581	Buhl and Hamilton 1990
<i>Tipula</i> sp. (crane fly)	39	D	>1035	>1035	>1174	Mebane et al. 2012
<i>Xyrauchen texanus</i> (razorback sucker)	199	T	>170000	>785	>389	Buhl 1997
<i>Xyrauchen texanus</i> (razorback sucker)	199	T	>170000	>785	>389	Buhl 1997

1 For tests in which only total recoverable or nominal Pb concentrations were reported, dissolved Pb concentrations were estimated as noted by WindWard (2012)

2 Dissolved EC50s normalized to 50 mg/l hardness using revised hardness regression described in memo text

**Attachment 2
Acute to Chronic Ratios (WindWard, 2012)**

Species	Hardness (mg/l)	Acute Value, Dissolved (µg/L)	Hardness-adjusted Acute Value (µg/L)	Chronic EC20, Dissolved (µg/L)	Hardness-adjusted Chronic Value (µg/L)	ACR1	Reference
<i>Baetis tricaudatus</i>	20	664	1058	66	105	10.1	Mebane et al. 2008
<i>Ceriodaphnia dubia</i>	100	186	131	-			Spehar and Fiantdt 19862
<i>Ceriodaphnia dubia</i>	82.4	208.8	162	2.2	2	94.9	Cooper et al. 2009
<i>Ceriodaphnia dubia</i>	42	540	590	21.4	23	25.2	Parametrix 2010a,b
<i>Ceriodaphnia dubia</i>	134	622	377	34.4	21	18.1	Parametrix 2010a,b
<i>Ceriodaphnia dubia</i>	272	2033	>859	64.9	27		Parametrix 2010a,b
<i>Ceriodaphnia dubia</i>	50.5	656	653	41.6	41	15.8	Parametrix 2010a,b
<i>Ceriodaphnia dubia</i>	50.5	279	278	26.7	27	10.4	Parametrix 2010a,b
<i>Ceriodaphnia dubia</i>	53	104	101	67.5	66	1.5	Parametrix 2010a,b
<i>Ceriodaphnia dubia</i>	35.5	100	119	35.4	42	2.8	Esbaugh et al. 2011, 2012
<i>Ceriodaphnia dubia</i>	162	435	239	22.6	12	19.2	Esbaugh et al. 2011, 2012
<i>Ceriodaphnia dubia</i>	225	996	464	96.7	45	10.3	Esbaugh et al. 2011, 2012
<i>Ceriodaphnia dubia</i>	28	1180	1585	223	299	5.3	Esbaugh et al. 2011, 2012
<i>Ceriodaphnia dubia</i>	16	290	518	12.1	22	24.0	Esbaugh et al. 2011, 2012
<i>Ceriodaphnia dubia</i>	40	221	248	63.6	71	3.5	De Schampelaere 2012; Nys et al. 2012
<i>Ceriodaphnia dubia</i>	111.5	275	183	91	61	3.0	De Schampelaere 2012; Nys et al. 2012
<i>Ceriodaphnia dubia</i>	164	249	136	112	61	2.2	De Schampelaere 2012; Nys et al. 2012
<i>Ceriodaphnia dubia</i>	228.5	320	148	101	47	3.2	De Schampelaere 2012; Nys et al. 2012
<i>Ceriodaphnia dubia</i>	108	388	262	79.4	54	4.9	De Schampelaere 2012; Nys et al. 2012
<i>Ceriodaphnia dubia</i>	107.5	235	159	80.8	55	2.9	De Schampelaere 2012; Nys et al. 2012
<i>Ceriodaphnia dubia</i>	107.5	216	146	80.1	54	2.7	De Schampelaere 2012; Nys et al. 2012
<i>Ceriodaphnia dubia</i>	105.5	410	280	153	105	2.7	De Schampelaere 2012; Nys et al. 2012
<i>Ceriodaphnia dubia</i>	13	141	280	53.1	105	2.7	AquaTox 2012
<i>Ceriodaphnia dubia</i>	14	120	229	23	44	5.2	AquaTox 2012
<i>Ceriodaphnia dubia</i>	78	29	23	8.9	7	3.3	AquaTox 2012
<i>Chironomus dilutus</i>	32	3323	4170	28.00	35	118.7	Mebane et al. 2008
<i>Daphnia magna</i>	53	612	594	14.5	14	42.2	Chapman et al. Manuscript
<i>Daphnia magna</i>	106	554	378	109.0	74	5.1	Chapman et al. Manuscript
<i>Daphnia magna</i>	151.5	562	320	54.9	31	10.2	Chapman et al. Manuscript
<i>Lampisilis siliquoidea</i>	45	206	217	16	17	12.9	Wang et al. 2010
<i>Oncorhynchus mykiss</i>	30	1170	1517	21.0	27	55.7	Davies et al. 1976
<i>Oncorhynchus mykiss</i>	19.7	120	193	34	55	3.5	Mebane et al. 2008
<i>Oncorhynchus mykiss</i>	29.4	133	174	102	134	1.3	Mebane et al. 2008
<i>Pimephales promelas</i>	44	1662	1774	-			Spehar and Fiantdt 1986
<i>Salvelinus fontinalis</i>	44	2470	2636	104	111	23.8	Holcombe et al. 1976



200 West Mercer St. • Suite 401 • Seattle, WA 98119
Phone: 206.378.1364 • Fax: 206.973.3048 • www.windwardenv.com

TECHNICAL MEMORANDUM

To: Jasim Chowdhury, Ph.D.
From: David DeForest
Subject: Phase 1 of BLM-based AWQC Project for Pb: Update of Acute and Chronic Toxicity Tables
Date: September 6, 2012

This technical memorandum provides the acute and chronic Pb toxicity tables that have been updated to support the development of biotic ligand model (BLM)-based ambient water quality criteria (AWQC) for Pb following U.S. Environmental Protection Agency (USEPA) guidelines. The following first provides a summary of how the toxicity data were compiled, followed by brief summaries of the acute toxicity data, chronic toxicity data, and the development of acute-chronic ratios (ACRs).

1 SOURCES OF TOXICITY TEST DATA

Acute and chronic Pb toxicity data were compiled from studies that met the USEPA guidelines for AWQC development (USEPA 1985a). These guidelines provide minimum requirements for acceptable test durations, endpoints, and other methodological components of toxicity tests. The initial sources of Pb toxicity data were the USEPA's 1984 AWQC document for Pb (USEPA 1985b) and the USEPA's draft toxicity tables for updated hardness-based AWQC from 2008 (USEPA 2008). The toxicity data compiled from these sources were augmented with several recent studies that have been published in the scientific literature, as well as from study reports that were developed by the independent research laboratories for the International Lead Zinc Research Organization (ILZRO).

For each acute and chronic toxicity test, data for BLM parameters (temperature, pH, dissolved organic carbon [DOC], Ca, Mg, Na, K, SO₄, Cl, and alkalinity) were compiled when

available. When concentration data were not available for DOC or ions, they were estimated based on the recommendations provided in Appendix C of the USEPA's AWQC document for Cu (USEPA 2007). If data for BLM parameters were not reported, and could not be reliably estimated, the test was excluded¹. For those tests in which only total recoverable Pb concentrations were reported, estimations made by Ronny Blust (University of Antwerp) for the dissolved Pb concentrations were used. The dissolved Pb concentrations were calculated using a combination of the inorganic Pb solubility translator and the effect of fulvic acid and humic acid binding on Pb solubility as predicted by WHAM VI (Blust 2012).

2 RESULTS

Acceptable acute toxicity data for Pb, including water chemistry data, were compiled for 36 species and 31 genera (Table 1), and acceptable chronic toxicity data were compiled for 14 species and 12 genera (Table 2). The USEPA's current chronic AWQC for Pb (USEPA 1985b) were derived using an acute-chronic ratio (ACR) of 51.29 because the minimum phylogenetic diversity requirements were not met for chronic toxicity data. Based on the updated acute and chronic datasets, several additional ACRs were derived. Because Pb solubility is highly variable depending on water chemistry, and varies between the higher concentrations associated with acute toxicity and the lower concentrations associated with chronic toxicity, ACRs were only derived from acute and chronic studies in which dissolved Pb was measured. The preliminary² final ACR based on the updated toxicity data is 6.2 (this is the geometric mean of the species mean ACRs) (Table 3). This final preliminary ACR was derived following USEPA guidelines, which includes meeting the minimum requirement of ACRs being available from at least three families provided that (1) one is a fish, (2) one is an invertebrate, and (3) one is an acutely sensitive freshwater species, and exclusion of the very high ACR of 118.7 for *C. dilutus* is justified because it is an acutely insensitive species.

We are currently planning on developing chronic BLM-based Pb criteria using both the ACR approach and using empirical chronic Pb toxicity data. Chronic Pb toxicity data are

¹ If it can be reliably determined that any excluded species would not be among the four most sensitive taxa, these data may be reconsidered for use in the development of AWQC for Pb (or at least considered as part of a sensitivity evaluation). This approach was used by DeForest and Van Genderen (2012) in deriving acute and chronic BLM-based 5th percentiles for zinc following USEPA guidelines.

² This ACR is considered preliminary because there could be reason to reassess this value once we get to the stage of deriving criteria (for example, if the resulting chronic criteria were determined not to be adequately protective based on empirical chronic toxicity data).

currently available for all of the eight minimum phylogenetic diversity requirements (Table 4), although smallmouth bass, representing a third family in the phylum Chordata, is not included in Table 2 because not all BLM parameters were measured or could be reasonably estimated. Nevertheless, the smallmouth bass toxicity data suggest this species is not chronically sensitive and would not be among the four most sensitive taxa. Further, there is precedence for deriving chronic AWQC from empirical chronic data even when all of the phylogenetic diversity requirements are not met. The chronic freshwater criteria developed in 1985 for Cd is one example (USEPA 1985c)³. In that case, use of empirical chronic data was preferred because the ACRs were highly variable and there was no pattern in the ACRs. For Pb, there is an overall relationship between the magnitude of ACRs and the associated acute values (i.e., the ACRs tend to increase with decreasing acute sensitivity), but ACRs within a species are still rather variable between tests and different chemistry types. For example, *C. dubia* is among the most acutely sensitive species to Pb, but the ACRs range between 1.5 and >31. Accordingly, it may be reasonably argued that chronic Pb criteria should be derived based on empirical chronic data, rather than through incorporation of an uncertain ACR.

3 NEXT STEP

The 2nd phase of the project will be to derive acute and chronic BLM-based AWQC for Pb using the toxicity data provided in Tables 1 and 2. In order to discuss the details of how this 2nd phase will be completed, a conference call with Bob Santore and Karel De Schamphelaere, as well as with the rest of the project team, is recommended.

4 REFERENCES

- AquaTox. 2012. Report on the toxicity of lead to the freshwater invertebrate, *Ceriodaphnia dubia*. Draft report. Prepared for the International Lead Zinc Research Organization. 21 pp. + appendices.
- Besser JM, Brumbaugh WG, Brunson EL, Ingersoll CG. 2005. Acute and chronic toxicity of lead in water and diet to the amphipod *Hyalella azteca*. *Environ Toxicol Chem* 24:1807-1815.
- Blust R. 2012. Chemical analysis and speciation modeling of lead solubility under ecotoxicity testing relevant exposure scenarios including the effect of dissolved organic matter. Draft report. Department of Biology, University of Antwerp, Belgium. 27 pp.
- Borgmann U, Kramar O, Loveridge C. 1978. Rates of mortality, growth, and biomass production of *Lymnaea palustris* during chronic exposure to lead. *J Fish Res Board Canada* 35:1109-1115.

³ The AWQC for Cd were updated in 2001, with the sufficient freshwater chronic data being available to meet the eight family rule (USEPA 2001).

- Brix KV, Esbaugh AJ, Munley KM, Grosell M. 2012. Investigations into the mechanism of lead toxicity to the freshwater pulmonate snail, *Lymnaea stagnalis*. *Aquat Toxicol* 106-107:147-156.
- Buhl KJ, Hamilton SJ. 1990. Comparative toxicity of inorganic contaminants released by placer mining to early life stages of salmonids. *Ecotoxicol Environ Saf* 20:325-342.
- Buhl KJ. 1997. Relative sensitivity of three endangered fishes, Colorado squawfish, bonytail, and razorback sucker, to selected metal pollutants. *Ecotoxicol Environ Saf* 37:186-192.
- Call DJ, Brooke LT, Ahmad N, Richter JE. 1983. Toxicity and metabolism studies with EPA priority pollutants and related organisms in freshwater organisms. Environmental Research Laboratory - Duluth, USEPA, Duluth, MN. EPA-600/3-83-095.
- Call DJ, Brooke LT, Ahmad N, Vaishnav DD. 1981. Aquatic pollutant hazard assessments and development of a hazard prediction technology by quantitative structure-activity relationships. Center for Lake Superior Environmental Studies, University of Wisconsin-Superior, Superior, WI. USEPA Cooperative Agreement No. CR 809234010.
- Chapman GA, Ota S, Recht F. 1980. Effects of water hardness on the toxicity of metals to *Daphnia magna*. USEPA, Corvallis, OR.
- Cooper NL, Bidwell JR, Kumar, A. 2009. Toxicity of copper, lead, and zinc mixtures to *Ceriodaphnia dubia* and *Daphnia carinata*. *Ecotoxicol Environ Saf* 72:1523-1528.
- Coughlan DJ, Gloss SP, Kubota J. 1986. Acute and sub-chronic toxicity of lead to the early life stages of smallmouth bass (*Micropterus dolomieu*). *Water Air Soil Pollut* 28:265-275.
- Davies PH, Goettl Jr JP, Sinley JR, Smith NF. 1976. Acute and chronic toxicity of lead to rainbow trout *Salmo gairdneri*, in hard and soft water. *Water Res* 10:199-206.
- DeForest DK, Van Genderen EJ. 2012. Application of USEPA guidelines in a bioavailability-based assessment of ambient water quality criteria for zinc in freshwater. *Environ Toxicol Chem* 31:1264-1272.
- De Schamphelaere KAC, Janssen CR. 2012. Effects of pH, Ca, and dissolved organic carbon on chronic toxicity of lead to the rotifer, *Brachionus calyciflorus*. Ghent University. Prepared for the International Lead Zinc Research Organization.
- De Schamphelaere KAC. 2012. Excel tables provided to J. Chowdhury, International Lead Zinc Research Organization. Acute toxicity tables for *Ceriodaphnia dubia* exposed to lead under varying Ca or pH in natural water or water amended with humic acid.
- Diamond JM, Koplisch DE, McMahon III J, Rost R. 1997. Evaluation of the water-effect ratio procedure for metals in a riverine system. *Environ Toxicol Chem* 16:509-520.
- Esbaugh AJ, Brix KV, Mager EM, De Schamphelaere K, Grosell M. 2012. Multi-linear regression analysis, preliminary biotic ligand modeling, and cross species comparison of the effects of water chemistry on chronic lead toxicity in invertebrates. *Comp Biochem Physiol C* 155:423-431.
- Esbaugh AJ, Brix KV, Mager EM, Grosell M. 2011. Multi-linear regression models predict the effects of water chemistry on acute lead toxicity to *Ceriodaphnia dubia* and *Pimephales promelas*. *Comp Biochem Physiol Part C* 154:137-145.

Grosell M, Gerdes RM, Brix KV. 2006. Chronic toxicity of lead to three freshwater invertebrates – *Brachionus calyciflorus*, *Chironomus tentans*, and *Lymnaea stagnalis*. Environ Toxicol Chem 25:97-104.

Holcombe GW, Benoit DA, Leonard EN, McKim JM. 1976. Long-term effects of lead exposure on three generations of brook trout (*Salvelinus fontinalis*). J Fish Research Board Can 33:1731-1741.

Mager EM, Brix KV, Gerdes RM, Ryan AC, Grosell M. 2011b. Effects of water chemistry on the chronic toxicity of lead to the cladoceran, *Ceriodaphnia dubia*. Ecotoxicol Environ Saf 74:238-243.

Mager EM, Esbough AJ, Brix KV, Ryan AC, Grosell M. 2011a. Influences of water chemistry on the acute toxicity of lead to *Pimephales promelas* and *Ceriodaphnia dubia*. Comp Biochem Physiol Part C 153:82-90.

Mebane CA, Dillon FS, Hennessy DP. 2012. Acute toxicity of cadmium, lead, zinc, and their mixtures to stream-resident fish and invertebrates. Environ Toxicol Chem 31:1334-1348.

Mebane CA, Hennessy DP, Dillon FS. 2008. Developing acute-to-chronic toxicity ratios for lead, cadmium, and zinc using rainbow trout, a mayfly, and a midge. Water Air Soil Pollut 188:41-66.

Nguyen LTH, Janssen CR, De Schamphelaere KAC. 2012. Chronic toxicity of Pb to *Chironomus riparius* in five natural waters. Final report submitted to International Lead Zinc Research Organisation. Ghent University, Ghent, Belgium. 16 pp. + annex.

Nys C, Janssen CR, De Schamphelaere KAC. 2012. Estimation of the competitive effects of Ca²⁺ and H⁺ (pH) on chronic toxicity of Pb²⁺ *Ceriodaphnia dubia*: Development and validation of a Biotic Ligand Model (BLM). Ghent University, Ghent, Belgium. 27 pp. + annex.

Parametrix. 2007. Evaluation of chronic lead toxicity to the great pond snail, *Lymnaea stagnalis*. Prepared for the Lead Development Association International. Albany, OR.

Parametrix. 2010a. Acute toxicity of lead to the cladoceran, *Ceriodaphnia dubia*, under varying calcium and pH water quality conditions. Prepared for the International Lead Zinc Research Organization. Albany, OR.

Parametrix. 2010b. Chronic toxicity of lead to the cladoceran, *Ceriodaphnia dubia*, under varying calcium and pH water quality conditions. Prepared for the International Lead Zinc Research Organization. Albany, OR.

Parametrix. 2010c. Chronic toxicity of lead to the fathead minnow, *Pimephales promelas*: a comparison of three different testing methodologies. Prepared for the International Lead Zinc Research Organization. Albany, OR.

Parametrix. 2010d. Chronic toxicity of lead to the fathead minnow, *Pimephales promelas*, in natural waters. Prepared for the International Lead Zinc Research Organization. Albany, OR.

Pérez-Legaspi, I.A., Rico-Martínez R. 2001. Acute toxicity tests on three species of the genus *Lecane* (Rotifera: Monogononta). Hydrobiol 446/447:375-381.

Rogers JT, Richards JG, Wood CM. 2003. Ionoregulatory disruption as the acute toxic mechanism for lead in the rainbow trout (*Oncorhynchus mykiss*). Aquat Toxicol 64:215-234.

Schubauer-Berigan MK, Dierkes JR, Monson PD, Ankley GT. 1993. pH-dependent toxicity of Cd, Cu, Ni, Pb and Zn to *Ceriodaphnia dubia*, *Pimephales promelas*, *Hyaella azteca*, and *Lumbriculus variegatus*. *Environ Toxicol Chem* 12:1261-1266.

Sloman KA, Baker DW, Ho CG, McDonald DG, Wood CM. 2003. The effects of trace metal exposure on agonistic encounters in juvenile rainbow trout, *Oncorhynchus mykiss*. *Aquat Toxicol* 63:187-196.

Spehar RL, Anderson RL, Fiandt JT. 1978. Toxicity and bioaccumulation of cadmium and lead in aquatic invertebrates. *Environ Pollut* 15:195-208.

Spehar RL, Fiandt JT. 1986. Acute and chronic effects of water quality criteria-based metal mixtures on three aquatic species. *Environ Toxicol Chem* 5:917-931.

Tsui MTK, Wang W-X, Chu LM. 2005. Influence of glyphosate and its formulation (Roundup®) on the toxicity and bioavailability of metals to *Ceriodaphnia dubia*. *Environ Pollut* 138:59-68.

USEPA (United States Environmental Protection Agency). 1985a. Guidelines for deriving numerical national water quality criteria for the protection of aquatic organisms and their uses. Washington, D.C. NTIS No. PB85-227049. 98 pages.

USEPA (United States Environmental Protection Agency). 1985b. Ambient aquatic life water quality criteria for lead. Office of Water, Regulations and Standards, Criteria and Standards Division. Washington, D.C. EPA 440/5-84-027.

USEPA (United States Environmental Protection Agency). 1985c. Ambient water quality criteria for cadmium. Office of Water, Regulations and Standards, Criteria and Standards Division. Washington, D.C. EPA 440/5-84-032.

USEPA (United States Environmental Protection Agency). 2001. 2001 update of ambient water quality criteria for cadmium. Office of Water, Washington, D.C. EPA-822-R-01-001.

USEPA (United States Environmental Protection Agency). 2007. Aquatic life ambient freshwater quality criteria - copper. Office of Water, Washington, D.C. EPA-822-R-07-001.

USEPA (United States Environmental Protection Agency). 2008. Draft updated acute and chronic lead toxicity tables.

Wang N, Ingersoll CG, Ivey CD, Hardesty DK, May TW, Augspurger T, Roberts AD, Van Genderen E, Barnhart MC. 2010. Sensitivity of early life stages of freshwater mussels (Unionidae) to acute and chronic toxicity of lead, cadmium, and zinc in water. *Environ Toxicol Chem* 29:2053-2063.

Yim JH, Kim KW, Kim SD. 2006. Effect of hardness on acute toxicity of metal mixtures using *Daphnia magna*: Prediction of acid mine drainage toxicity. *J Hazard Mater* 138:16-21.

Table 1. Acute freshwater toxicity data for lead.

Species	Pb Form	Age, Size, or LifeStage	Method	Temp (°C)	pH	DOC (mg/L)	HA (%)	Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	SO ₄ (mg/L)	Cl (mg/L)	Alkalinity (mg/L)	Hardness (mg/L)	Total (T) or Diss. (D)	Reported EC50 (µg/L)	EC50, Diss. ¹ (µg/L)	Diss. EC50 Adjusted to 85 mg/L Hardness (µg/L)	SMAV Adjusted to 85 mg/L Hardness (µg/L)	GMAV Adjusted to 85 mg/L Hardness (µg/L)	Reference
<i>Aplexa hypnorum</i> (snail)	Nitrate	Adult	FT,M	23.5	7.3	1.1	10	13.6	3.0	1.3	0.6	3.4	1.2	47.5	60.9	T	1340	1340	2049	2049	2049	Call et al. 1981
<i>Arctopsyche</i> sp. (caddisfly)	Chloride	Larvae	R,M	10.4	6.7	0.6	10	5.9	1.8	1.5	0.4	3.6	0.7	24	22	D	>1255	>1255	>7013	>7013	>7013	Mebane et al. 2012
<i>Baetis tricaudatus</i> (mayfly)	Chloride	Larvae	R,M	10.6	6.6	0.6	10	4.0	1.2	1.1	0.3	2.3	0.5	14	15	D	592	592	5367	4116	4116	Mebane et al. 2012
<i>Baetis tricaudatus</i> (mayfly)	Chloride	Larvae	R,M	11.1	7.1	0.6	10	4.8	1.5	1.3	0.4	3.3	0.6	16	16	D	752	752	5425			Mebane et al. 2012
<i>Baetis tricaudatus</i> (mayfly)	Chloride	Larvae	R,M	10.3	6.7	0.6	10	5.4	1.6	1.4	0.4	3.2	0.6	16	20	D	664	664	4189			Mebane et al. 2008, 2012
<i>Baetis tricaudatus</i> (mayfly)	Chloride	Larvae	R,M	10.4	6.7	0.6	10	5.9	1.8	1.5	0.4	3.6	0.7	24	22	D	426	426	2380			Mebane et al. 2012
<i>Baetis tricaudatus</i> (mayfly)	Chloride	Larvae	R,M	10.4	6.9	0.6	10	10.5	3.2	2.2	0.7	8.4	1.2	29	39	D	1002	1002	2701			Mebane et al. 2012
<i>Baetis tricaudatus</i> (mayfly)	Chloride	Larvae	R,M	10.4	7.3	0.6	10	17.8	5.5	4.6	1.0	27.7	2.0	42	67	D	>952	>952	>1299*			Mebane et al. 2012
<i>Baetis tricaudatus</i> (mayfly)	Chloride	Larvae	R,M	10.4	7.6	0.6	10	22.2	6.9	5.7	1.2	35.7	2.4	57	84	D	>683	>683	>693*			Mebane et al. 2012
<i>Baetis tricaudatus</i> (mayfly)	Chloride	Larvae	R,M	10.4	7.6	0.6	10	4.7	1.4	1.3	0.3	3.3	0.6	16	17	D	>494	>494	>3853*			Mebane et al. 2012
<i>Baetis tricaudatus</i> (mayfly)	Chloride	Larvae	FT,M	5.2	7.3	0.6	10	2.9	0.9	1.0	0.2	1.9	0.4	11	11	D	322	322	4348			Mebane et al. 2012
<i>Baetis tricaudatus</i> (mayfly)	Chloride	Larvae	R,M	8.0	7.3	0.6	10	3.5	1.1	1.1	0.3	1.8	0.4	11	13	D	511	511	5579			Mebane et al. 2012
<i>Baetis tricaudatus</i> (mayfly)	Chloride	Larvae	R,U	5.4	6.5	0.6	10	5.1	1.6	1.3	0.4	3.6	0.7	16	19	D	640	640	4310			Mebane et al. 2012
<i>Baetis tricaudatus</i> (mayfly)	Chloride	Larvae	R,U	5.4	6.6	0.6	10	8.9	2.7	1.9	0.6	6.5	1.0	28	33	D	>852	>852	>3175*			Mebane et al. 2012
<i>Baetis tricaudatus</i> (mayfly)	Chloride	Larvae	R,U	5.4	6.7	0.6	10	11.1	3.4	2.3	0.7	13.4	1.1	38	41	D	>1250	>1250	<3162*			Mebane et al. 2012
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	R,M	20	7.6	0.5	10	4.2	3.6	7.8	0.6	24.2	0.6	23.0	25	D	26.1	26.1	138	380	380	Diamond et al. 1997
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	R,M	20	7.6	0.5	10	4.2	3.6	7.8	0.6	24.2	0.6	23.0	25	D	187	187	888			Diamond et al. 1997
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	R,M	20	7.6	0.5	10	4.2	3.6	7.8	0.6	24.2	0.6	23.0	25	D	46.1	46.1	219			Diamond et al. 1997
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	R,M	20	7.6	0.5	10	4.2	3.6	7.8	0.6	24.2	0.6	23.0	25	D	26.4	26.4	125			Diamond et al. 1997
<i>Ceriodaphnia dubia</i> (cladoceran)	Chloride	Neonates (<24-hr)	S,M	25	8.1	4.0	10	11.5	9.9	21.5	2.1	66.6	1.6	50.9	69.5	T	400	400	517			Tau et al. 2005
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	S,M	25.3	7.5	0.5	10	13.6	11.7	25.5	2.0	46.4	1.8	65.0	62.4	T	255	209	217			Cooper et al. 2009
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	S,M	20	7.5	0.5	10	10.0	3.6	9.9	3.2	31.7	3.1	24	40	D	540	540	1418			Parametrix 2010a
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	S,M	20	8.0	<0.5	10	40.3	8.4	35.8	6.7	119	8.1	88	135	D	622	622	345			Parametrix 2010a
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	S,M	20	7.9	<0.5	10	38.1	14.6	75.0	7.0	206	6.1	96	155	D	379	379	178			Parametrix 2010a
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	S,M	20	8.2	<0.5	10	77.9	18.9	56.1	6.8	257	6.0	120	272	D	>2033	>2033	462			Parametrix 2010a
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	S,M	20	6.1	<0.5	10	15.3	3.2	13.3	3.2	40.1	17.5	16	51	D	656	656	1248			Parametrix 2010a
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	S,M	20	7.0	<0.5	10	15.2	3.1	13.0	3.0	37.4	6.5	20	51	D	279	279	537			Parametrix 2010a
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	S,U	20	8.1	<0.5	10	15.2	3.1	14.5	3.2	39.5	3.4	32	51	T	287	287	552*			Parametrix 2010a
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	S,U	20	8.0	<0.5	10	15.2	3.1	14.5	3.2	39.5	3.4	36	51	T	1614	482	927*			Parametrix 2010a
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	S,M	20	8.4	<0.5	10	15.0	3.1	14.8	3.4	38.3	3.1	38	50	D	104	104	204			Parametrix 2010a
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	R,M	26	7.3	2.6	10	11.8	2.1	14.7	2.5	6.8	27.2	21.1	39.1	D	100	100	278			Esbaugh et al. 2011
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	R,M	26	8.2	1.3	10	35.6	19.0	6.1	3.3	58.2	6.5	67.1	167.2	D	435	435	184			Esbaugh et al. 2011
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	R,M	26	8.6	7.8	10	66.4	2.4	9.2	2.3	2.1	13.9	193	225.5	D	996	996	287			Esbaugh et al. 2011
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	R,M	26	7.3	17.6	10	8.7	1.5	5.3	3.4	2.4	8.9	13.6	27.7	D	1180	1180	4909			Esbaugh et al. 2011
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	R,M	26	7.2	7.5	10	4.0	1.4	2.5	1.8	1.9	4.1	6.1	15.8	D	290	290	2474			Esbaugh et al. 2011
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	R,M	26	8.4	1.8	10	46.2	27.4	33.9	7.1	47.3	33.2	178	228.1	D	108	108	31			Esbaugh et al. 2011
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	R,M	26	5.7	0.6	10	4.2	1.4	1.8	2.2	2.0	14.5	0.3	16.3	D	73.5	73.5	602			Esbaugh et al. 2011
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	R,M	26	6.8	1.2	10	1.3	0.6	1.9	2.3	1.8	2.6	3.5	5.8	D	28.8	28.8	669			Esbaugh et al. 2011
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	R,M	25	7.6	2.6	10	12.6	3.5	29.0	2.5	6.2	37.7	25.3	47.0	D	395	395	640			Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	R,M	25	8.0	4.0	10	33.2	7.5	54.5	4.3	19.7	75.2	54.3	116	D	387	387	260			Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	R,M	25	7.4	2.9	10	24.8	3.2	16.9	2.0	8.6	39.4	29.4	76.0	D	433	433	498			Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	R,M	25	7.4	2.9	10	43.6	3.2	18.2	2.0	8.6	39.4	29.4	125	D	597	597	365			Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	R,M	25	7.4	3.4	10	66.2	3.2	29.2	2.3	99.6	34.2	24.5	183	D	385	385	145			Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	R,M	25	7.4	3.9	10	92.3	3.5	32.4	2.6	147	38.2	27.8	250	D	319	319	81			Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	R,M	25	7.8	2.7	18	10.3	3.6	27.1	2.3	6.1	31.4	35.7	41.0	D	425	425	1075			Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	R,M	25	7.8	3.2	28	14.5	4.1	28.0	2.6	7.2	35.9	39.7	54.0	D	546	546	973			Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	R,M	25	7.5	2.1	10	8.9	3.3	28.9	2.2	6.1	30.5	31.1	36.0	D	591	591	1764			Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	R,M	25	7.9	5.0	26	28.1	7.2	56.9	4.3	19.8	77.9	56.3	102	D	532	532	422			Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	R,M	25	8.2	10.8	67	27.0	7.3	56.3	4.3	20.7	76.3	59.2	99.0	D	964	964	794			Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	R,M	25	8.2	15.6	77	31.9	5.9	92.7	11.0	35.4	120	62.5	108	D	3116	3116	2353			Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	R,M	25	8.0	2.6	10	16.0	2.5	49.8	2.9	12.8	49.1	42.9	51.0	D	384	384	736			Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	R,M	25	8.2	2.8	10	15.9	2.5	64.9	2.9	9.2	52.6	66.3	51.0	D	779	779	1493			Mager et al. 2011a
<i>Ceriodaphnia dubia</i> (cladoceran)	Nitrate	Neonates (<24-hr)	R,M	25	8.1	2.7	10	11.8	3.5	67.5	2.7	10.4	51.2									

Table 1. Acute freshwater toxicity data for lead.

Species	Pb Form	Age, Size, or Lifestage	Method	Temp (°C)	pH	DOC (mg/L)	HA (%)	Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	SO ₄ (mg/L)	Cl (mg/L)	Alkalinity (mg/L)	Hardness (mg/L)	Total (T) or Diss. (D)	Reported EC50 (µg/L)	EC50, Diss. ¹ (µg/L)	Diss. EC50 Adjusted to 85 mg/L Hardness	SMAV Adjusted to 85 mg/L Hardness	GMAV Adjusted to 85 mg/L Hardness	Reference
<i>Ceriodaphnia dubia</i> (cladoceran)	Chloride	Neonates (<24-hr)	S.M	24	6.3	3.3	10	43.9	3.8	66.3	4.0	134.7	67.8	0.8 (DIC)	125	D	358	358				De Schampelaere 2012
<i>Ceriodaphnia dubia</i> (cladoceran)	Chloride	Neonates (<24-hr)	S.M	24	6.9	3.9	10	42.7	3.8	66.3	4.0	134.7	67.8	2.1 (DIC)	122	D	235	235				De Schampelaere 2012
<i>Ceriodaphnia dubia</i> (cladoceran)	Chloride	Neonates (<24-hr)	S.M	24	7.6	3.4	10	43.6	3.8	66.3	4.0	115.5	67.8	7.0 (DIC)	125	D	216	216				De Schampelaere 2012
<i>Ceriodaphnia dubia</i> (cladoceran)	Chloride	Neonates (<24-hr)	S.M	24	8.2	2.7	10	42.2	3.8	66.3	4.0	9.8	67.8	30.8 (DIC)	121	D	410	410				De Schampelaere 2012
<i>Ceriodaphnia dubia</i> (cladoceran)	Chloride	Neonates (<24-hr)	S.M	24	7.0	3.1	10	9.7	3.8	16.1	4.0	9.8	14.7	2.8 (DIC)	40	D	221	221				De Schampelaere 2012
<i>Ceriodaphnia dubia</i> (cladoceran)	Chloride	Neonates (<24-hr)	S.M	24	7.0	3.3	10	38.6	3.8	14.6	4.0	81.9	14.7	2.6 (DIC)	112	D	275	275				De Schampelaere 2012
<i>Ceriodaphnia dubia</i> (cladoceran)	Chloride	Neonates (<24-hr)	S.M	24	7.0	2.7	10	66.0	3.8	11.3	4.0	153.9	14.7	2.1 (DIC)	181	D	249	249				De Schampelaere 2012
<i>Ceriodaphnia dubia</i> (cladoceran)	Chloride	Neonates (<24-hr)	S.M	24	7.0	2.4	10	98.9	3.8	9.1	4.0	225.9	14.7	2.2 (DIC)	258	D	320	320				De Schampelaere 2012
<i>Chironomus dilutus</i> ² (midge)	Chloride	Larvae	R.M	22.2	7.6	0.6	10	8.6	2.6	1.9	0.6	9.6	1.3	27	32	D	1955	1955	6780			Mebane et al. 2012
<i>Chironomus dilutus</i> ² (midge)	Chloride	Larvae	R.M	22.5	7.1	0.6	10	8.6	2.6	1.9	0.6	9.7	1.3	36	32	D	3617	3617	12544			Mebane et al. 2008, 2012
<i>Cottus bairdii</i> (shorthead sculpin)	Chloride	30-60 mm	R.M	9.0	7.3	0.6	10	5.6	1.7	1.4	0.4	3.8	0.7	22	21	D	>855	>855	>5069	>5069	>5069	Mebane et al. 2012
<i>Daphnia magna</i> (cladoceran)	Nitrate	Neonates (<24-hr)	S.M	19.4	7.6	1.1	10	15.4	3.9	13.2	3.2	25.3	12.1	48.0	54	T	612	612	1090	353	353	Chapman et al. Manuscript
<i>Daphnia magna</i> (cladoceran)	Nitrate	Neonates (<24-hr)	S.M	19.4	8.1	1.1	10	31.9	7.5	31.9	5.8	62.7	23.1	83.0	110	T	952	952	399			Chapman et al. Manuscript
<i>Daphnia magna</i> (cladoceran)	Nitrate	Neonates (<24-hr)	S.M	19.4	8.3	1.1	10	44.1	10.3	44.1	8.1	86.6	31.9	120	152	T	1910	562	268			Chapman et al. Manuscript
<i>Daphnia magna</i> (cladoceran)	Nitrate	Neonates (<24-hr)	S.U	25	7.8	0.5	10	7.2	6.3	13.6	1.1	42.2	1.0	30.0	44	T	95	95	220			Yim et al. 2006
<i>Daphnia magna</i> (cladoceran)	Nitrate	Neonates (<24-hr)	S.U	25	8.0	0.5	10	24.6	21.4	46.3	4.2	144	3.4	121	150	T	894	439	213			Yim et al. 2006
<i>Daphnia magna</i> (cladoceran)	Nitrate	Neonates (<24-hr)	S.U	25	8.0	0.5	10	5.2	1.6	1.3	0.4	4.1	0.7	16	18.5	D	>267	>267	>1740	>1740	>1740	Mebane et al. 2012
<i>Daphnia magna</i> (cladoceran)	Nitrate	Neonates (<24-hr)	S.U	25	8.0	0.5	10	10.5	3.2	2.2	0.7	8.4	1.2	29	39	D	>1035	>1035	>2790	>2790	>2790	Mebane et al. 2012
<i>Daphnia magna</i> (cladoceran)	Nitrate	Neonates (<24-hr)	S.U	25	8.0	0.5	10	4.7	1.4	1.3	0.3	3.3	0.6	16	17	D	>494	>494	>3833	>4232	>4232	Mebane et al. 2012
<i>Daphnia magna</i> (cladoceran)	Nitrate	Neonates (<24-hr)	S.U	25	8.0	0.5	10	2.9	0.9	1.0	0.2	1.9	0.4	11	11	D	>346	>346	>4672			Mebane et al. 2012
<i>Daphnia magna</i> (cladoceran)	Nitrate	Neonates (<24-hr)	S.U	25	8.0	0.5	10	14.3	3.2	1.4	0.8	3.6	1.3	40.8	48.3	T	140	140	287	283	283	Call et al. 1983
<i>Daphnia magna</i> (cladoceran)	Nitrate	Neonates (<24-hr)	S.U	25	8.0	0.5	10	13.6	3.0	1.3	0.6	3.4	1.2	42.0	45	T	124	124	279			Spehar et al. 1978
<i>Daphnia magna</i> (cladoceran)	Nitrate	Neonates (<24-hr)	S.U	25	8.0	0.5	10	46.0	20.0	49.0	0.1	174	21.0	106	199	T	>170000	>785	>266	>266	>266	BuH 1997
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	0.053 g	FT.M	17.6	6.5	1.1	10	4.8	1.5	1.3	0.4	3.3	0.6	16	18	D	544	544	3925	3315	3315	Mebane et al. 2012
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5-7 mm	FT.M	15	7.4	1.1	10	5.4	1.6	1.4	0.4	3.2	0.6	16	20	D	537	537	3388			Mebane et al. 2012
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5 d	FT.M	15	7.4	1.1	10	5.1	1.6	1.3	0.4	3.3	0.6	17	19	D	380	380	2559			Mebane et al. 2012
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5 d	FT.M	15	7.4	1.1	10	5.4	1.6	1.4	0.4	3.2	0.6	16	22	D	796	796	4446			Mebane et al. 2012
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5 d	FT.M	15	7.4	1.1	10	10.5	3.2	2.2	0.7	8.4	1.2	29	39	D	981	981	2645			Mebane et al. 2012
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5 d	FT.M	15	7.4	1.1	10	17.8	5.5	4.8	1.0	27.7	2.0	42	67	D	>952	>952	>1289*			Mebane et al. 2012
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5 d	FT.M	15	7.4	1.1	10	22.2	6.9	5.7	1.2	35.7	2.4	57	84	D	>683	>683	>693*			Mebane et al. 2012
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5 d	FT.M	15	7.4	1.1	10	3.5	1.1	1.1	0.3	1.8	0.4	11	13	D	644	644	7031*			Mebane et al. 2012
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5 d	FT.M	15	7.4	1.1	10	5.1	1.8	1.3	0.4	3.6	0.7	16	19	D	>1035	>1035	>6970*			Mebane et al. 2012
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5 d	FT.M	15	7.4	1.1	10	8.9	2.7	1.9	0.6	6.5	1.0	26	33	D	>952	>952	>3175*			Mebane et al. 2012
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5 d	FT.M	15	7.4	1.1	10	11.1	3.4	2.3	0.7	13.4	1.1	38	41	D	>683	>683	>1728*			Mebane et al. 2012
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5 d	FT.M	15	7.4	1.1	10	18.6	6.1	5.5	0.6	12.7	5.8	64.0	71	D	>151	>151	>190	<190	<190	Besser et al. 2005
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5 d	FT.M	15	7.4	1.1	10	6.7	5.8	11.5	1.0	38.8	0.9	34.0	42	D	188	188	461	461	469	Wang et al. 2010
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5 d	FT.M	15	7.4	1.1	10	7.5	6.5	11.5	1.1	43.5	1.0	30.0	47	D	>299	>299	>636*			Wang et al. 2010
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5 d	FT.M	15	7.4	1.1	10	6.5	5.8	11.5	1.0	37.9	0.9	36.0	41	D	142	142	359			Wang et al. 2010
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5 d	FT.M	15	7.4	1.1	10	7.5	6.5	11.5	1.1	43.5	1.0	40.0	47	D	298	298	634			Wang et al. 2010
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5 d	FT.M	15	7.4	1.1	10	6.4	5.5	14.5	1.0	37.0	0.9	39.0	40	D	>426	>426	>1112*			Wang et al. 2010
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5 d	FT.M	15	7.4	1.1	10	14.0	12.1	26.3	2.1	81.4	1.9	65.0	85	T	680	503	503	503	351	Pérez-Legaspi and Rico-Martínez 2001
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5 d	FT.M	15	7.4	1.1	10	14.0	12.1	26.3	2.1	81.4	1.9	65.0	85	T	140	140	140			Pérez-Legaspi and Rico-Martínez 2001
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5 d	FT.M	15	7.4	1.1	10	14.0	12.1	26.3	2.1	81.4	1.9	65.0	85	T	3700	616	616	616		Pérez-Legaspi and Rico-Martínez 2001
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5 d	FT.M	15	7.4	1.1	10	47.9	41.5	89.8	7.2	278	6.5	235	290	T	>8000	>8000	>1677	>438	>438	Schubauer-Berigan et al. 1993
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5 d	FT.M	15	7.4	1.1	10	47.9	41.5	89.8	7.2	278	6.5	235	290	T	>8000	>2307	>484			Schubauer-Berigan et al. 1993
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5 d	FT.M	15	7.4	1.1	10	47.9	41.5	89.8	7.2	278	6.5	235	290	T	>8000	>494	>104			Schubauer-Berigan et al. 1993
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5 d	FT.M	15	7.4	1.1	10	8.6	2.6	1.9	0.8	4.8	0.8	27.5	32	D	>123	>123	>427*	832	1168	Mebane et al. 2012
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5 d	FT.M	15	7.4	1.1	10	8.5	2.6	1.9	0.8	6.0	0.9	32	31.5	D	>54	>54	>191*			Mebane et al. 2012
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5 d	FT.M	15	7.4	1.1	10	8.8	2.6	1.9	0.8	6.0	1.0	26.5	32	D	215	215	746			Mebane et al. 2012
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5 d	FT.M	15	7.4	1.1	10	8.8	2.6	1.9	0.8	4.8	0.8	27.5	30.5	D	>72	>72	>265*			Mebane et al. 2012
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5 d	FT.M	15	7.4	1.1	10	8.5	2.6	1.9	0.8	5.7	0.9	28.8	31.5	D	362	362	1281			Mebane et al. 2012
<i>Gammarus pseudolimnensis</i> (amphipod)	Nitrate	5 d	FT.M	15	7.4	1.1	10	16.7	5.1	4.3	1.0	29.1	2.1	48.5	56	D	487	487				

Table 1. Acute freshwater toxicity data for lead.

Species	Pb Form	Age, Size, or Lifestage	Method	Temp (°C)	pH	DOC (mg/L)	HA (%)*	Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	SO ₄ (mg/L)	Cl (mg/L)	Alkalinity (mg/L)	Hardness (mg/L)	Total (T) or D.Ds.	Reported EC50 (µg/L)	EC50, D.Ds. ¹ (µg/L)	Dias. EC50 Adjusted to 85 Hardness (µg/L)	SMAV Adjusted to 85 Hardness (µg/L)	GMAV Adjusted to 85 Hardness (µg/L)	Reference
<i>Oncorhynchus mykiss</i> (rainbow trout)	Nitrate	66 mm	S.M	14	8.2	1.6	10	82.7	32.4	16.0	1.2	112	25.8	267	365	D	1320	1320	193*			Davies et al. 1976
<i>Oncorhynchus mykiss</i> (rainbow trout)	Nitrate	145 mm	FT,M	10	6.9	1.6	10	12.5	3.1	10.9	1.4	2.8	4.0	30.0	32	D	1170	1170	4058*			Davies et al. 1976
<i>Oncorhynchus mykiss</i> (rainbow trout)	Chloride	Swim-up fry	FT,M	9.4	6.6	0.6	10	5.4	1.6	1.4	0.4	3.2	0.6	16.0	20	D	138	138	982			Mebane et al. 2008, 2012
<i>Oncorhynchus mykiss</i> (rainbow trout)	Chloride	Swim-up fry	FT,M	9.0	7.6	0.6	10	8.6	2.8	1.9	0.6	6.4	1.0	26.5	32	D	127	127	433			Mebane et al. 2008, 2012
<i>Oncorhynchus mykiss</i> (rainbow trout)	Chloride	Swim-up fry	FT,M	9.5	7.5	0.6	10	8.6	2.6	1.9	0.6	6.4	1.0	27.0	32	D	160	160	546			Mebane et al. 2008, 2012
<i>Oncorhynchus mykiss</i> (rainbow trout)	Nitrate	Juvenile (1.5-3.5g)	FT,M	9.5	8.0	3	10	40.1	4.9	13.8	2.0	24.0	28.4	116	120	D	1000	1000	644			Rogers et al. 2003
<i>Oncorhynchus mykiss</i> (rainbow trout)	Nitrate	Juvenile	FT,U	13	8.0	3	10	40.0	4.9	13.8	0.8	0.8	24.8	82.0	120	T	1000	684	492*			Stoman et al. 2003
<i>Oncorhynchus mykiss</i> (rainbow trout)	Chloride	Fry	R,M	9.4	6.6	0.6	10	5.4	1.6	1.4	0.4	3.2	0.6	16	20	D	138	138	671			Mebane et al. 2012
<i>Oncorhynchus mykiss</i> (rainbow trout)	Chloride	Fry	R,M	9.0	7.6	0.6	10	6.6	2.6	1.9	0.6	6.4	1.0	27	31.5	D	127	127	449			Mebane et al. 2012
<i>Oncorhynchus mykiss</i> (rainbow trout)	Chloride	Fry	R,M	9.5	7.5	0.6	10	6.6	2.6	1.9	0.6	6.4	1.0	27	32	D	160	160	555			Mebane et al. 2012
<i>Oncorhynchus mykiss</i> (rainbow trout)	Chloride	Fry	R,M	6.9	6.9	0.6	10	5.1	1.6	1.3	0.4	2.9	0.6	21	19	D	591	591	3980			Mebane et al. 2012
<i>Oncorhynchus mykiss</i> (rainbow trout)	Chloride	Fry	R,M	7.0	7.0	0.6	10	6.7	2.1	1.6	0.5	5.0	0.8	23	25	D	631	631	2996			Mebane et al. 2012
<i>Oncorhynchus mykiss</i> (rainbow trout)	Chloride	Fry	R,M	7.1	7.1	0.6	10	6.6	2.6	1.9	0.6	6.5	1.0	28	32	D	916	916	3177			Mebane et al. 2012
<i>Oncorhynchus mykiss</i> (rainbow trout)	Chloride	Fry	R,M	7.0	6.9	0.6	10	9.0	2.6	2.0	0.6	11.6	1.0	27	33.5	D	969	969	3170			Mebane et al. 2012
<i>Oncorhynchus mykiss</i> (rainbow trout)	Chloride	Fry	R,M	10.9	7.4	0.6	10	7.7	2.3	1.7	0.5	5.3	0.9	25	28.5	D	>98	>98	>394*			Mebane et al. 2012
<i>Oncorhynchus mykiss</i> (rainbow trout)	Chloride	Fry	R,M	9.0	7.3	0.6	10	5.9	1.6	1.4	0.4	3.8	0.7	22	21	D	180	180	1067			Mebane et al. 2012
<i>Paraleptophlebia</i> sp. (mayfly)	Chloride	Larvae	R,M	8.0	7.3	0.6	10	2.9	0.9	1.0	0.2	1.9	0.4	11	11	D	>346	>346	>4672	>4672	>4672	Mebane et al. 2012
<i>Physa</i> sp. (snail)	Chloride	Larvae	R,M	10.3	6.7	0.6	10	5.4	1.6	1.4	0.4	3.2	0.6	18.0	22	D	1159	1159	6476	6476	6476	Mebane et al. 2012
<i>Pimephales promelas</i> (fathead minnow)	Nitrate	<24 hrs	R,M	26	7.6	2.2	10	10.1	1.8	15.3	2.0	5.0	25.0	17.8	33	D	622	622	2058	2476	2476	Ebsbaugh et al. 2011
<i>Pimephales promelas</i> (fathead minnow)	Nitrate	<24 hrs	R,M	26	7.1	15.9	10	7.8	1.6	6.2	1.9	2.4	6.9	11.4	26	D	3588	3588	16267			Ebsbaugh et al. 2011
<i>Pimephales promelas</i> (fathead minnow)	Nitrate	<24 hrs	R,M	26	5.7	0.6	10	4.0	1.4	3.8	0.3	2.4	15.2	0.3	16	D	41	41	348			Ebsbaugh et al. 2011
<i>Pimephales promelas</i> (fathead minnow)	Nitrate	<24 hrs	R,M	26	6.7	1.3	10	0.9	0.6	2.3	1.6	1.9	2.3	3.8	5	D	68	68	2681			Ebsbaugh et al. 2011
<i>Pimephales promelas</i> (fathead minnow)	Nitrate	8-d	FT,M	22.7	7.6	0.9	10	5.3	1.3	11.9	1.0	3.8	15.3	9.1	19	D	178	178	1239			Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	Nitrate	8-d	FT,M	24.4	7.5	1.2	10	10.3	1.5	14.3	1.0	14.7	20.1	11.8	32	D	744	744	2578			Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	Nitrate	8-d	FT,M	23.6	7.6	1.1	10	17.8	1.7	14.8	1.0	32.1	20.9	10.1	52	D	1015	1015	1919			Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	Nitrate	8-d	FT,M	24.5	7.4	1.1	10	23.6	1.7	15.1	1.0	48.7	21.6	9.6	66	D	1068	1068	1478			Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	Nitrate	8-d	FT,M	19	7.7	1.1	10	22.0	2.3	22.1	1.1	36.2	26.0	21.1	65	D	1148	1148	1630			Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	Nitrate	8-d	FT,M	17.5	7.5	1.1	10	118.2	1.5	14.6	0.8	194	16.4	13.3	301	D	1719	1719	343			Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	Nitrate	8-d	FT,M	23.8	7.6	1.4	35	5.5	1.4	13.8	1.0	5.2	21.2	8.4	20	D	608	608	3932			Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	Nitrate	8-d	FT,M	24	7.6	1.7	48	6.2	1.5	13.9	1.0	5.0	21.5	9.3	21	D	1075	1075	6216			Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	Nitrate	8-d	FT,M	24.9	7.5	2.6	65	6.1	1.5	14.2	1.0	5.0	21.6	8.8	22	D	1356	1356	7794			Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	Nitrate	8-d	FT,M	24.6	7.6	5.0	82	5.9	1.5	14.5	1.0	5.3	21.9	12.4	21	D	3249	3249	19361			Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	Nitrate	8-d	FT,M	22.2	7.5	1.1	10	6.3	1.6	19.6	1.1	6.1	26.1	21.4	22	D	816	816	4477			Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	Nitrate	8-d	FT,M	22.1	7.9	1.1	10	6.4	1.6	25.0	1.1	6.1	25.6	37.2	23	D	996	996	5373			Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	Nitrate	8-d	FT,M	22.3	8.0	1.2	10	6.0	1.7	31.1	1.3	4.0	21.0	50.1	22	D	698	698	3919			Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	Nitrate	8-d	FT,M	22.5	7.5	1.1	10	6.6	1.7	31.3	1.1	6.0	69.3	12.9	23	D	370	370	1909			Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	Nitrate	8-d	FT,M	19.8	5.4	1.6	10	6.7	1.5	16.5	0.5	5.0	52.2	21.0	28	D	162	162	671			Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	Nitrate	8-d	FT,M	20.5	6.3	1.4	10	9.1	1.5	12.9	0.4	3.9	32.0	24.0	29	D	265	265	1055			Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	Nitrate	8-d	FT,M	20.5	7.5	1.2	10	9.4	1.5	13.3	0.4	4.1	14.7	25.7	30	D	624	624	2377			Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	Nitrate	8-d	FT,M	20	8.3	1.5	10	7.5	1.5	24.1	0.4	4.6	20.5	33.4	25	D	340	340	1646			Mager et al. 2011a
<i>Pimephales promelas</i> (fathead minnow)	Nitrate	30-d	FT,M	25	7.4	2.0	10	12.9	2.8	1.2	0.6	3.2	1.1	42.4	43.9	T	2100	1662	3654			Spehar and Flindt 1986
<i>Pimephales promelas</i> (fathead minnow)	Chloride	<24 hrs	S,M	25	6.3	0.5	10	47.9	41.5	89.8	7.2	278	6.5	235	290	T	810	810	108*			Schubauer-Berigan et al. 1993
<i>Pimephales promelas</i> (fathead minnow)	Chloride	<24 hrs	S,M	25	7.1	0.5	10	47.9	41.5	89.8	7.2	278	6.5	235	290	T	>5400	>2277	>720*			Schubauer-Berigan et al. 1993
<i>Pimephales promelas</i> (fathead minnow)	Chloride	<24 hrs	S,M	25	8.3	0.5	10	47.9	41.5	89.8	7.2	278	6.5	235	290	T	>5400	>473	>720*			Schubauer-Berigan et al. 1993
<i>Ptychocheilus lucius</i> (Colorado pikeminnow)	Nitrate	100-d	S,U	25	8.0	0.5	10	46.0	20.0	49.0	0.1	174	21.0	106	199	T	>170000	>785	>266	>266	>266	Buhl 1997
<i>Ptychocheilus lucius</i> (Colorado pikeminnow)	Nitrate	155-d	S,U	25	8.0	0.5	10	46.0	20.0	49.0	0.1	174	21.0	106	199	T	>170000	>785	>266			Buhl 1997
<i>Ptychocheilus lucius</i> (Colorado pikeminnow)	Nitrate	9-d	S,U	25	8.0	0.5	10	46.0	20.0	49.0	0.1	174	21.0	106	199	T	>170000	>785	>266			Buhl 1997
<i>Rhithrogena</i> sp. (mayfly)	Chloride	Larvae	R,M	10.6	6.6	0.6	10	4.0	1.2	1.1	0.3	2.3	0.5	14	15	D	>737	>737	>6706	>3763	>3763	Mebane et al. 2012
<i>Rhithrogena</i> sp. (mayfly)	Chloride	Larvae	R,M	11.1	7.1	0.6	10	4.8	1.5	1.3	0.4	3.3	0.6	16	18	D	>985	>985	>7106			Mebane et al. 2012
<i>Rhithrogena</i> sp. (mayfly)	Chloride	Larvae	FT,M	13.6	6.7	0.6	10	5.1	1.6	1.3	0.4	1.9	0.4	19	19	D	>166	>166	>1118			Mebane et al. 2012
<i>Salvelinus fontinalis</i> (brook trout)	Nitrate	72-wk	FT,M	12	7.2	1.1	10	13.1	2.9	1.3	0.5	3.3	1.2	42.6	44.3	T	4100	2470	5661	5661	5661	Holcombe et al. 1976
<i>Simulium</i> sp. (black fly)	Chloride	Larvae	R,M	10.4	6.7	0.6	10	5.9	1.8	1.5	0.4	3.6	0.7	24	22	D	415	415	2319	2451	2451	Mebane et al. 2012
<i>Simulium</i> sp. (black fly)	Chloride	Larvae	R,M	10.4	6.9	0.6	10	10.5	3.2	2.2	0.7	8.4	1.2	29	39	D	961	961	2591			Mebane et al. 2012
<i>Swelta</i> sp. (stonefly)	Chloride	Larvae	R,M	10.6	6.6	0.6	10	4.0	1.2	1.1	0.3	2.3	0.5	14	15	D	>737	>737	>6706*	1648	1648	Mebane et al. 2012
<i>Swelta</i> sp. (stonefly)	Chloride	Larvae	FT,M	10.7	6.8	0.6	10	5.2	1.6	1.3	0.4	3.1	0.7	18	19.5	D	253	253	1648			Mebane et al. 2012
<i>Swelta</i> sp. (stonefly)	Chloride	Larvae	FT,M	5.2	7.3	0.6	10	4.7	1.4	1.3	0.3	3.3	0.6	16.0	17	D	>494	>494	>3833*			Mebane et al. 2012
<i>Tanytarsus dissimilis</i> (midge)	Nitrate	Larvae	FT,M	16.6	5.9	1.1	10	13.6	3.0	1.3	0.6	1.2	3.4	12.1	46	T	224000	202530*	442536	442536	442536	Call et al. 1983
<i>Tanytarsini</i> sp. (chironomid)	Chloride	Larvae	R,M	10.4	6.9	0.6	10	10.5	3.2	2.2	0.7	8.4	1.2	29	39	D	>1035	>1035	>2790	>4424	>4424	Mebane et al. 2012
<i>Tanytarsini</i> sp. (chironomid)	Chloride	Larvae	R,M	10.3	6.7	0.6	10	5.4	1.6	1.4	0.4	3.2	0.6	18	22	D	>1255	>1255	>7013			Mebane et al. 2012
<i>Thymallus arcticus</i> (Arctic grayling)	Nitrate	Alevin	S,U	12	7.6	0.5	10	6.9	6.0	12.9	1.1	40.0	0.9	30.9	41.3	T	>36000	>850	>62999*	<802		

Table 1. Active freshwater toxicity data for lead.

Species	Pb Form	Age, Sex, or Lifestage	Method	Temp (°C)	pH	DOC (mg/L)	HA (%)	Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	SO ₄ (mg/L)	Cl (mg/L)	Alkalinity (mg/L)	Hardness (mg/L)	Total (T) or Reported (R) EC50 (µg/L)	Dis. EC50 Adjusted to 85 (µg/L)	SMAV Adjusted to 85 (µg/L)	GMAV Adjusted to 85 (mg/L)	Reference
<i>Xytrichon retentus</i> (macrobrachy sucker)	Nitrate	100-d	S,U	25	8.0	0.5	10	46.0	20.0	49.0	0.1	74	21.0	106	189	>170000	>185	>266	>266	Burt 1997
<i>Xytrichon retentus</i> (macrobrachy sucker)	Nitrate	6-d	S,U	25	6.0	0.5	10	46.0	20.0	49.0	0.1	174	21.0	106	189	>170000	>185	>266	>266	Burt 1997

NR = not reported

S = static

R = renewal

FT = flow-through

M = measured

U = unmeasured

T = total recoverable Pb

D = dissolved Pb

EC50 = median effect concentration

SMAV = species mean acute value

GMAV = genus mean acute value

DOC = dissolved organic carbon

HA = humic acid (a default value of 10% is assumed unless otherwise noted)

DIC = dissolved inorganic carbon

* For tests in which only total recoverable or nominal Pb concentrations were reported, dissolved Pb concentrations were estimated by R. Blett (Univ. of Antwerp) using a combination of the inorganic Pb solubility translator and the effect of FA + HA binding on Pb solubility as predicted by WHAM VI.

* Formerly *Chironomus tentans*

* The reported EC50 was outside of the calibration range so the dissolved Pb concentration was estimated using the EPA's hardness-based translator.

* Excluded from calculation of SMAV.

Table 3. Freshwater acute-chronic ratios (ACRs) for lead.

Species	Chronic Endpoint	Hardness (mg/L)	Acute Value, TR (µg/L)	Geometric Mean of NOEC and LOEC, TR (µg/L)	Chronic EC20, TR (µg/L)	Acute Value, Diss. (µg/L)	Geometric Mean of NOEC and LOEC, Diss. (µg/L)	Chronic EC20, Diss. (µg/L)	ACR ¹	Species Mean ACR	Reference
<i>Baetis tricaudatus</i>	10-d molts	20	-	-	-	664	130	66	10.1	10.1	Mebane et al. 2008
<i>Ceriodaphnia dubia</i>	7-d reproduction	100	248	52	-	186	39	-	4.8*	5.4	Spehar and Fiandt 1986 ⁴
<i>Ceriodaphnia dubia</i>	7-d reproduction	82.4	254.9	3.8	2.7	208.8	3.1	2.2	94.9*		Cooper et al. 2009
<i>Ceriodaphnia dubia</i>	7-d reproduction	40-44	-	-	-	540	58.3	21.4	25.2		Parametrix 2010a,b
<i>Ceriodaphnia dubia</i>	7-d reproduction	133-135	-	-	-	622	60.2	34.4	18.1		Parametrix 2010a,b
<i>Ceriodaphnia dubia</i>	7-d reproduction	272	-	-	-	>2033	60.8	64.9	>31.3**		Parametrix 2010a,b
<i>Ceriodaphnia dubia</i>	7-d reproduction	50-51	-	-	-	656	60.7	41.6	15.8		Parametrix 2010a,b
<i>Ceriodaphnia dubia</i>	7-d reproduction	50-51	-	-	-	279	26.8	26.7	10.4		Parametrix 2010a,b
<i>Ceriodaphnia dubia</i>	7-d reproduction	51-55	-	-	-	104	43.4	67.5	1.5		Parametrix 2010a,b
<i>Ceriodaphnia dubia</i>	7-d reproduction	33-38	-	-	-	100	-	35.4	2.8		Esbaugh et al. 2011, 2012
<i>Ceriodaphnia dubia</i>	7-d reproduction	157-167	-	-	-	435	-	22.6	19.3		Esbaugh et al. 2011, 2012
<i>Ceriodaphnia dubia</i>	7-d reproduction	224-226	-	-	-	996	-	96.7	10.3		Esbaugh et al. 2011, 2012
<i>Ceriodaphnia dubia</i>	7-d reproduction	28	-	-	-	1180	-	223	5.3		Esbaugh et al. 2011, 2012
<i>Ceriodaphnia dubia</i>	7-d reproduction	16	-	-	-	290	-	12.1	24.0		Esbaugh et al. 2011, 2012
<i>Ceriodaphnia dubia</i>	7-d reproduction	40	-	-	-	221	103	63.6	3.5		De Schampelaere 2012; Nys et al. 2012
<i>Ceriodaphnia dubia</i>	7-d reproduction	111-112	-	-	-	275	107	91	3.0		De Schampelaere 2012; Nys et al. 2012
<i>Ceriodaphnia dubia</i>	7-d reproduction	147-181	-	-	-	249	114	112	2.2		De Schampelaere 2012; Nys et al. 2012
<i>Ceriodaphnia dubia</i>	7-d reproduction	199-258	-	-	-	320	113	101	3.2		De Schampelaere 2012; Nys et al. 2012
<i>Ceriodaphnia dubia</i>	7-d reproduction	91-125	-	-	-	388	61.2	79.4	4.9		De Schampelaere 2012; Nys et al. 2012
<i>Ceriodaphnia dubia</i>	7-d reproduction	93-122	-	-	-	235	57.5	80.8	2.9		De Schampelaere 2012; Nys et al. 2012
<i>Ceriodaphnia dubia</i>	7-d reproduction	90-125	-	-	-	216	52.8	80.1	2.7		De Schampelaere 2012; Nys et al. 2012
<i>Ceriodaphnia dubia</i>	7-d reproduction	90-121	-	-	-	410	182	153	2.7		De Schampelaere 2012; Nys et al. 2012
<i>Ceriodaphnia dubia</i>	6-d reproduction	13	-	-	-	141	49	53.1	2.6		AquaTox 2012
<i>Ceriodaphnia dubia</i>	6-d reproduction	14	-	-	-	120	57	23	5.2		AquaTox 2012
<i>Ceriodaphnia dubia</i>	6-d reproduction	78	-	-	-	29	11	8.9	3.3		AquaTox 2012
<i>Chironomus dilutus</i>	20-d weight	32	-	-	-	3323	65	28.00	118.7	118.7***	Mebane et al. 2008
<i>Daphnia magna</i>	21-d reproduction	53	612	12.26	14.5	612	12.26	14.5	42.2*	-	Chapman et al. Manuscript
<i>Daphnia magna</i>	21-d reproduction	106	952	118.8	109	554	118.8	109.0	5.1*		Chapman et al. Manuscript
<i>Daphnia magna</i>	21-d reproduction	151.5	1910	128.1	54.9	562	128.1	54.9	10.2*		Chapman et al. Manuscript
<i>Lamprolaima siliquoidea</i>	28-d survival	45****	-	-	-	206	10	16	12.9	12.9	Wang et al. 2010
<i>Oncorhynchus mykiss</i>	19-mo abnormalities	28-32	-	18.9	21.0	1170	18.9	21.0	55.7*	2.1	Davies et al. 1976
<i>Oncorhynchus mykiss</i>	69-d survival	19.7	-	-	-	120	36	34	3.5		Mebane et al. 2008
<i>Oncorhynchus mykiss</i>	62-d length	29.4	-	-	-	133	12	102	1.3		Mebane et al. 2008
<i>Pimephales promelas</i>	32-d growth	44	2100	329	-	1662	329	-	5.1*	-	Spehar and Fiandt 1986
<i>Salvelinus fontinalis</i>	12-wk growth (3rd generation)	44	4100	83.1	104	2470	83.1	104	23.8*	23.8*	Holcombe et al. 1976
Preliminary geometric mean ACR:										6.2	

TR = total recoverable

¹ ACR is based on the chronic EC20, where available, or the chronic value if no EC20 is available.

⁴ The acute and chronic toxicity data for *C. dubia* from Spehar and Fiandt (1986) were not included in Tables 1 and 2 because insufficient information on test water chemistry was provided and concentrations of water chemistry parameters could not be reasonably estimated. The mean dissolved-to-total recoverable Pb ratio in the study was 0.75.

* ACR excluded from calculation of the species mean ACR because dissolved Pb concentrations were not measured in both the acute and chronic toxicity tests.

** ACR excluded from the species mean ACR for *C. dubia* because several other determinant ACRs (i.e., non "greater than" values) are available for this species.

*** Excluded from final geometric mean ACR because the species is acutely insensitive to Pb.

**** The acute value is geometric mean of two acute tests conducted at a hardness of 41 and 47 mg/L and the chronic value was conducted at 46 mg/L.

Table 4. Chronic lead toxicity studies meeting the USEPA's minimum phylogenic diversity requirements.

Requirement	Representative Species
Family Salmonidae in the class Osteichthyes	Rainbow trout, brook trout
Second family in the class Osteichthyes, preferably a commercially or recreational important species (e.g., bluegill, channel catfish, etc.)	Fathead minnow
Third family in the phylum Chordata (may be in the class Osteichthyes or may be an amphibian, etc.)	Smallmouth bass ¹
Planktonic crustacean (e.g., cladocerans, copepod, etc.)	<i>Ceriodaphnia dubia</i> , <i>Daphnia magna</i>
Benthic crustacean (e.g., ostracod, isopod, amphipod, crayfish, etc.)	<i>Hyalella azteca</i>
Insect (e.g., mayfly, dragonfly, damselfly, stonefly, caddisfly, mosquito, midge, etc.)	<i>Baetic tricaudatus</i> , <i>Chironomus dilutus</i> , <i>C. riparius</i>
Family in a phylum other than Arthropoda or Chordata (e.g., Rotifera, Annelida, Mollusca, etc.)	<i>Brachionus calyciflorus</i> , <i>Phylodina rapida</i>
Family in any order of insect or any phylum not already represented	<i>Lampsilis siliquoidea</i> , <i>Lymnaea palustris</i> , <i>L. stagnalis</i>

¹ Chronic toxicity data meeting USEPA guidelines are available for smallmouth bass, in the family Centrarchidae (Coughlan et al. 1986); however, it was not included in Table 2 because levels of all BLM parameters in the test water could not be reasonably assumed.

**AMBIENT AQUATIC LIFE WATER QUALITY CRITERIA FOR
LEAD**

**U.S. ENVIRONMENTAL PROTECTION AGENCY
OFFICE OF RESEARCH AND DEVELOPMENT
ENVIRONMENTAL RESEARCH LABORATORIES
DULUTH, MINNESOTA
NARRAGANSETT, RHODE ISLAND**

DISCLAIMER

This report has been reviewed by the Criteria and Standards Division, Office of Water Regulations and Standards, U.S. Environmental Protection Agency, and approved for publication. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

AVAILABILITY NOTICE

This document is available to the public through the National Technical Information Service (NTIS), 5285 Port Royal Road, Springfield, VA 22161.
NTIS ACCESSION NUMBER - PB 85-227432

FOREWORD

Section 304(a)(1) of the Clean Water Act of 1977 (P.L. 95-217) requires the Administrator of the Environmental Protection Agency to publish criteria for water quality accurately reflecting the latest scientific knowledge on the kind and extent of all identifiable effects on health and welfare which may be expected from the presence of pollutants in any body of water, including ground water. This document is a revision of proposed criteria based upon a consideration of comments received from other Federal agencies, State agencies, special interest groups, and individual scientists. The criteria contained in this document replace any previously published EPA aquatic life criteria.

The term "water quality criteria" is used in two sections of the Clean Water Act, section 304(a)(1) and section 303(c)(2). The term has a different program impact in each section. In section 304, the term represents a non-regulatory, scientific assessment of ecological effects. The criteria presented in this publication are such scientific assessments. Such water quality criteria associated with specific stream uses when adopted as State water quality standards under section 303 become enforceable maximum acceptable levels of a pollutant in ambient waters. The water quality criteria adopted in the State water quality standards could have the same numerical limits as the criteria developed under section 304. However, in many situations States may want to adjust water quality criteria developed under section 304 to reflect local environmental conditions and human exposure patterns before incorporation into water quality standards. It is not until their adoption as part of the State water quality standards that the criteria become regulatory.

Guidelines to assist the States in the modification of criteria presented in this document, in the development of water quality standards, and in other water-related programs of this Agency, have been developed by EPA.

Edwin L. Johnson
Director
Office of Water Regulations and Standards

ACKNOWLEDGMENTS

Duane A. Benoit
(freshwater author)
Environmental Research Laboratory
Duluth, Minnesota

John H. Gentile
(saltwater author)
Environmental Research Laboratory
Narragansett, Rhode Island

Charles E. Stephan
(document coordinator)
Environmental Research Laboratory
Duluth, Minnesota

David J. Hansen
(saltwater coordinator)
Environmental Research Laboratory
Narragansett, Rhode Island

Statistical Support: John W. Rogers

Clerical Support: Terry L. Highland

CONTENTS

	<u>Page</u>
Foreword	iii
Acknowledgments	iv
Tables	vi
Introduction	1
Acute Toxicity to Aquatic Animals	4
Chronic Toxicity to Aquatic Animals	7
Toxicity to Aquatic Plants	9
Bioaccumulation	10
Other Data	10
Unused Data	11
Summary	15
National Criteria	16
References	41

TABLES

	<u>Page</u>
1. Acute Toxicity of Lead to Aquatic Animals	19
2. Chronic Toxicity of Lead to Aquatic Animals	23
3. Ranked Genus Mean Acute Values with Species Mean Acute-Chronic Ratios	25
4. Toxicity of Lead to Aquatic Plants	28
5. Bioaccumulation of Lead by Aquatic Organisms	30
6. Other Data on Effects of Lead on Aquatic Organisms	32

Introduction*

Because of the variety of forms of lead (Boggess, 1977; Callahan, et al. 1979) and lack of definitive information about their relative toxicities, no available analytical measurement is known to be ideal for expressing aquatic life criteria for lead. Previous aquatic life criteria for lead (U.S. EPA, 1980) were expressed in terms of total recoverable lead (U.S. EPA, 1983a), but this measurement is probably too rigorous in some situations. Acid-soluble lead (operationally defined as the lead that passes through a 0.45 μm membrane filter after the sample is acidified to pH = 1.5 to 2.0 with nitric acid) is probably the best measurement at the present for the following reasons:

1. This measurement is compatible with nearly all available data concerning toxicity of lead to, and bioaccumulation of lead by, aquatic organisms. Very few test results were rejected just because it was likely that they would have been substantially different if they had been reported in terms of acid-soluble lead. For example, results reported in terms of dissolved lead were not used if the concentration of precipitated lead was substantial.
2. On samples of ambient water, measurement of acid-soluble lead should measure all forms of lead that are toxic to aquatic life or can be readily converted to toxic forms under natural conditions. In addition, this measurement should not measure several forms, such as lead that is occluded in minerals, clays, and sand or is strongly sorbed to

*An understanding of the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" (Stephan, et al. 1985), hereafter referred to as the Guidelines, is necessary in order to understand the following text, tables, and calculations.

particulate matter, that are not toxic and are not likely to become toxic under natural conditions. Although this measurement (and many others) will measure soluble, complexed forms of lead, such as the EDTA complex of lead, that probably have low toxicities to aquatic life, concentrations of these forms probably are negligible in most ambient water.

3. Although water quality criteria apply to ambient water, the measurement used to express criteria is likely to be used to measure lead in aqueous effluents. Measurement of acid-soluble lead should be applicable to effluents because it will measure precipitates, such as carbonate and hydroxide precipitates of lead, that might exist in an effluent and dissolve when the effluent is diluted with receiving water. If desired, dilution of effluent with receiving water before measurement of acid-soluble lead might be used to determine whether the receiving water can decrease the concentration of acid-soluble lead because of sorption.
4. The acid-soluble measurement should be useful for most metals, thus minimizing the number of samples and procedures that are necessary.
5. The acid-soluble measurement does not require filtration at the time of collection, as does the dissolved measurement.
6. The only treatment required at the time of collection is preservation by acidification to $\text{pH} = 1.5$ to 2.0 , similar to that required for the total recoverable measurement.
7. Durations of 10 minutes to 24 hours between acidification and filtration probably will not affect the result substantially.
8. The carbonate system has a much higher buffer capacity from $\text{pH} = 1.5$ to 2.0 than it does from $\text{pH} = 4$ to 9 (Weber and Scumm, 1963).
9. Differences in pH within the range of 1.5 to 2.0 probably will not affect the result substantially.

10. The acid-soluble measurement does not require a digestion step, as does the total recoverable measurement.

11. After acidification and filtration of the sample to isolate the acid-soluble lead, the analysis can be performed using either atomic absorption spectroscopy or ICP-atomic emission spectroscopy (U.S. EPA, 1983a), as with the total recoverable measurement.

Thus, expressing aquatic life criteria for lead in terms of the acid-soluble measurement has both toxicological and practical advantages. On the other hand, because no measurement is known to be ideal for expressing aquatic life criteria for lead or for measuring lead in ambient water or aqueous effluents, measurement of both acid-soluble lead and total recoverable lead in ambient water or effluent or both might be useful. For example, there might be cause for concern if total recoverable lead is much above an applicable limit, even though acid-soluble lead is below the limit.

Unless otherwise noted, all concentrations reported herein are expected to be essentially equivalent to acid-soluble lead concentrations. All concentrations are expressed as lead, not as the chemical tested. The criteria presented herein supersede previous aquatic life water quality criteria for lead (U.S. EPA, 1976, 1980) because these new criteria were derived using improved procedures and additional information. Whenever adequately justified, a national criterion may be replaced by a site-specific criterion (U.S. EPA, 1983b), which may include not only site-specific criterion concentrations (U.S. EPA, 1983c), but also site-specific durations of averaging periods and site-specific frequencies of allowed exceedences (U.S. EPA, 1985). The latest literature search for information for this document was conducted in May, 1984; some newer information was also used.

Acute Toxicity to Aquatic Animals

Acute tests were conducted at three different levels of water hardness with Daphnia magna (Chapman, et al. Manuscript), demonstrating that daphnids were three times more sensitive to lead in soft than in hard water (Table 1). The value in soft water agrees closely with the value in Table 6 for the same species in soft water (Biesinger and Christensen, 1972). Data in Table 1 also indicate that lead was more toxic to the rainbow trout, fathead minnow, and bluegill in soft than in hard water. The results of the acute tests conducted by Davies, et al. (1976) with rainbow trout in hard water are reported as unmeasured values in Table 1, because total lead concentrations were not measured, even though the dissolved concentrations were. Hale (1977) conducted an acute exposure of rainbow trout to lead and obtained an LC50 of 8,000 $\mu\text{g/L}$. This value is almost seven times greater than the LC50 obtained for rainbow trout in soft water by Davies, et al. (1976). Hale did not report water hardness; however, alkalinity and pH were reported to be 105 mg/L and 7.3, respectively, which suggests that this water was probably harder than the soft water used by Davies, et al. (1976).

Amphipods were reported by Spehar, et al. (1978) and Call, et al. (1983) to be more sensitive to lead than any other freshwater animal species thus far tested. Also, in exposures lasting up to 28 days the amphipod was far more sensitive to lead than a snail, cladoceran, chironomid, mayfly, stonefly, and caddisfly (Table 6) (Anderson, et al. 1980; Biesinger and Christensen, 1972; Nehring, 1976; Spehar, et al. 1978). Although results of tests on lead acetate were placed in Table 6 because of the possible effect of acetate on the toxicity of lead, Pickering and Henderson (1966) found that lead chloride (Table 1) and lead acetate (Table 6) were about equally toxic to the fathead minnow in static tests in soft water. Wallen, et al. (1957) reported that

lead oxide (Table 6) is much less acutely toxic than lead nitrate (Table 1) to the mosquitofish in water containing a high concentration of suspended clay particles.

Different species exhibit different sensitivities to lead, and many other factors might affect the results of tests of the toxicity of lead to aquatic organisms. Criteria can quantitatively take into account such a factor, however, only if enough data are available to show that the factor similarly affects the results of tests with a variety of species. Hardness is often thought of as having a major effect on the toxicity of lead, although the observed effect is probably due to one or more of a number of usually interrelated ions, such as hydroxide, carbonate, calcium, and magnesium. Hardness is used here as a surrogate for the ions which affect the results of toxicity tests on lead. An analysis of covariance (Dixon and Brown, 1979; Neter and Wasserman, 1974) was performed using the natural logarithm of the acute value as the dependent variable, species as the treatment or grouping variable, and the natural logarithm of hardness as the covariate or independent variable. This analysis of covariance model was fit to the data in Table 1 for the four species for which acute values are available over a range of hardness such that the highest hardness is at least three times the lowest and the highest is also at least 100 mg/L higher than the lowest. An F-test showed that, under the assumption of equality of slopes, the probability of obtaining four slopes as dissimilar as these is $P=0.03$. This was interpreted as indicating that it is unreasonable to assume that the slopes for these four species are the same. The slopes for Daphnia magna, fathead minnow, and bluegill (see end of Table 1) were close to the slope of 1.0 that is expected on the basis that lead, calcium, magnesium, and carbonate all have

a charge of two. The slope for rainbow trout was 2.475 and therefore was not used. A test of equality of slopes showed that $P=0.16$, indicating that it is not unreasonable to assume that the slopes for the three species are the same.

The pooled slope of 1.273 was used with the data in Table 1 to calculate Species Mean Acute Values at a hardness of 50 mg/L (Table 1). Genus Mean Acute Values (Table 3) were then calculated as geometric means of the available freshwater Species Mean Acute Values. Of the ten genera for which acute values are available, the most sensitive genus, Gammarus, was 1,650 times more sensitive than the most resistant, Tanytarsus. The freshwater Final Acute Value of 67.54 $\mu\text{g/L}$ was calculated at a hardness of 50 mg/L from the Genus Mean Acute Values in Table 3 using the procedure described in the Guidelines. Thus, the freshwater Criterion Maximum Concentration (in $\mu\text{g/L}$) = $e^{(1.273[\ln(\text{hardness})]-1.460)}$.

Tests of the acute toxicity of lead to saltwater organisms have been conducted with nine species of invertebrates and four species of fish (Table 1). In flow-through toxicity tests with two fish species, less than 50 percent of the test organisms were killed at 3,140 $\mu\text{g/L}$, which is the solubility of lead in sea water under the test conditions, but the acute value for the mummichog is 315 $\mu\text{g/L}$. The range of sensitivities of bivalve molluscs is also great, probably reflecting differences in life stage. The adult soft-shell clam had an LC50 of 27,000 $\mu\text{g/L}$, whereas the acute values with larvae of four species ranged from 476 to 2,450 $\mu\text{g/L}$. Of the eleven saltwater genera for which acute values are available, the most sensitive genus, Fundulus, was 85 times more sensitive than the most resistant, Mya (Table 3). The sensitivities of the six most sensitive genera differed by only a factor of 2.5, even though these six lowest Genus Mean Acute Values are from tests

conducted with a variety of species and life stages. The saltwater Final Acute Value was calculated to be 287.4 $\mu\text{g/L}$.

Chronic Toxicity to Aquatic Animals

Chapman, et al. (Manuscript) studied the chronic toxicity of lead to Daphnia magna at three different hardnesses (Table 2). The daphnids were nearly 11 times more sensitive to lead in soft water than in hard water. The value in soft water was about one-fourth that obtained by Biesinger and Christensen (1972) with the same species in a different soft water in a test in which the concentrations of lead were not measured (Table 6). The chronic values of Chapman, et al. were regressed against hardness; the slope was 2.328, but the 95% confidence limits were -8.274 and 12.931.

A life-cycle test on lead in hard water was conducted by Borgmann, et al. (1978) with a snail. These authors used biomass as their endpoint and reported that lead concentrations as low as 19 $\mu\text{g/L}$ significantly decreased survival, but not growth or reproduction. It is not clear, however, how these investigators arrived at such a low effect concentration. This publication did, however, contain suitable information for determining a chronic value. Chronic limits were taken directly from the cumulative percent survival figure which showed no observed effect on survival at 12 $\mu\text{g/L}$ and almost complete mortality at 54 $\mu\text{g/L}$. The chronic value (geometric mean of the lower and upper limits) for snails was therefore established at 25.46 $\mu\text{g/L}$ (Table 2).

Davies, et al. (1976) published results of an early life-stage test with rainbow trout in soft water (Table 2). Even though this test was started with embryos and continued for 19 months after hatch, it could not be considered a life-cycle test because no reproduction occurred. Davies, et al. (1976)

selected chronic limits based on a very low incidence of black-colored tails and spinal deformities (4.7 and 0.7 percent, respectively). For the purposes of deriving water quality criteria, such low percentages of such effects were not considered unacceptable. The concentration of 27 $\mu\text{g}/\text{L}$ was selected as the upper limit because it caused spinal curvature in 32.2 percent of the fish, whereas 13.2 $\mu\text{g}/\text{L}$ only caused curvature in 3.6 percent of the fish. The occurrence of black tails was not considered to be an unacceptable effect.

Spinal deformities were also caused by lead in a life-cycle test with brook trout (Holcombe, et al. 1976) and in an early life-stage test with rainbow trout (Sauter, et al. 1976). Results of tests by Sauter, et al. (1976) with the northern pike, walleye, lake trout, channel catfish, white sucker, and bluegill were not included in Tables 2 or 6 because of excessive mortality in the controls. Even though the hardnesses were similar, the chronic value obtained for rainbow trout by Sauter, et al. (1976) is higher than the chronic value derived from Davies, et al. (1976), possibly because Sauter, et al. exposed the fish for 2 months, whereas Davies, et al. exposed the fish for 19 months.

Davies, et al. (1976) described the long-term effects on rainbow trout fry and fingerlings exposed to various concentrations of lead for 19 months in hard and soft water (Table 6). Although these tests were neither life-cycle (no natural reproduction) nor early life-stage (no embryos exposed), they do provide information concerning the relationship between water hardness and the chronic toxicity of lead to fish. In the test in hard water, only 0 and 10 percent of the trout developed spinal deformities at measured lead concentrations of 190 and 380 $\mu\text{g}/\text{L}$, respectively. In soft water 44 and 97

percent of the trout developed spinal deformities at measured lead concentrations of 31 and 62 µg/L, respectively. These results strongly demonstrate that lead is more chronically toxic in soft water than in hard water.

The mysid, Mysidopsis bahia, is the only saltwater species with which a chronic test has been conducted on lead (Table 2). The most sensitive observed adverse effect was reduced spawning and the resulting chronic value was 25.08 µg/L. The 96-hr LC50 for this same species in the same study was 3,130 µg/L, producing an acute-chronic ratio of 124.8.

The range of the available acute-chronic ratios (Table 3) is small enough that all four can be used to calculate the geometric mean ratio of 51.29. When this ratio is used with the freshwater Final Acute Value and the pooled slope (Table 3), the resulting freshwater Final Chronic Value (in µg/L) = $e^{(1.273[\ln(\text{hardness})]-4.705)}$. Similarly, the saltwater Final Chronic Value is 5.603 µg/L (Table 3).

Toxicity to Plants

The effects of lead on various species of algae have been studied in tests which lasted from 4 to 10 days (Table 4). All authors except Rachlin, et al. (1982, 1983) used nominal concentrations. The adverse effect concentrations from these tests ranged from 500 to 63,800 µg/L. It would appear therefore that adverse effects of lead on freshwater plants are unlikely at concentrations protective of chronic effects on freshwater animals.

The saltwater alga, Champia parvula, is quite sensitive to lead and a diatom is only slightly less sensitive (Table 4). The saltwater alga, Dunaliella tertiolecta, is 10 times more sensitive to tetraethyl lead than to tetramethyl lead (Table 6).

Bioaccumulation

Four freshwater invertebrate species have been exposed to lead (Borgmann, et al. 1978; Spehar, et al. 1978) and the bioconcentration factors (BCFs) ranged from 499 to 1,700 (Table 5). BCFs obtained with brook trout and bluegills were 42 and 45, respectively, (Atchison, et al. 1977; Holcombe, et al. 1976).

BCFs reported for lead from tests with saltwater bivalve molluscs and diatoms range from 17.5 from a 56-day exposure of the quahog clam to 2,570 from a 130-day exposure of the blue mussel (Table 5). The difference in BCFs might be a difference between species or might be due to the difference in the duration of the tests.

~~Neither a freshwater nor a saltwater Final Residue Value can be~~ calculated because no maximum permissible tissue concentration is available for lead.

Other Data

Many of the values in Table 6 have already been discussed. Spehar, et al. (1978) found no adverse effects on a freshwater snail, stonefly, and caddisfly in 28 days at 565 $\mu\text{g}/\text{L}$. Anderson, et al. (1980) obtained a 10-day LC50 of 258 $\mu\text{g}/\text{L}$ for the midge, Tanytarsus dissimilis (Table 6), which is much lower than the 48-hr acute value of 224,000 $\mu\text{g}/\text{L}$ obtained by Call, et al. (1983) with the same species. The 10-day exposure includes most of its life cycle and several of the presumably sensitive molts, and so should probably be considered as useful as the early life-stage test with fish. Merlini and Pozzi (1977a) conducted a pH acclimation and lead bioconcentration study with bluegills collected from a lake contaminated with lead.

A variety of effects on saltwater organisms have been observed. Gray and Ventilla (1973) observed a reduction in growth rate in a ciliate protozoan after 12-hr exposures to lead concentrations of 150 and 300 $\mu\text{g/L}$. Woolery and Lewin (1976) observed a reduction in photosynthesis and respiration in the diatom, Pheodactylum tricornerum, at concentrations of lead ranging from 100 to 10,000 $\mu\text{g/L}$. However, Hannan and Patouillet (1972) obtained no inhibition of growth of the same species at a concentration of 1,000 $\mu\text{g/L}$ after 72 hours. Rivkin (1979), using growth rate to determine toxicity to the diatom, Skeletonema costatum, reported a 12-day EC50 of 5.1 $\mu\text{g/L}$. Hessler (1974) observed delayed cell division in the phytoplankton, Platymonas subcordiformis, during exposure to 2,500 $\mu\text{g/L}$ for 72 hours. At 60,000 $\mu\text{g/L}$, however, Hessler (1974) reported not only growth retardation but also death. Benijts-Claus and Benijts (1975) observed delayed larval development in the mud crab, Rhithropanopeus harrisi, during exposure to 50 $\mu\text{g/L}$. Weis and Weis (1977) observed depressed axis formation in developing embryos of Fundulus heteroclitus at lead concentrations of 100 $\mu\text{g/L}$. Raish and Carr (1978) found that 1,000 $\mu\text{g/L}$ suppressed reproduction of two polychaete species, Ctenodrilus serratus and Ophryotrocha diadema, in a 21-day test.

Unused Data

Some data on the effects of lead on aquatic organisms were not used because the studies were conducted with species that are not resident in North America. Jennett, et al. (1981) did not identify their test animals beyond common names such as "algae, crayfish, and minnows". Nehring, et al. (1979) did not identify their organisms to species, so it is not known if

Brown and Ahsanullah (1971) conducted tests with brine shrimp, which species is too atypical to be used in deriving national criteria.

Data were not used if lead was a component of a mixture (Hedcke and Puglisi, 1980; Heisey and Damman, 1982; Jana and Choudhuri, 1984; Wong, et al. 1982). Reviews by Chapman, et al. (1968), Demayo, et al. (1980, 1982), Eisler (1981), Eisler, et al. (1979), North, et al. (1972), Phillips and Russo (1978), and Thompson, et al. (1972) only contain data that have been published elsewhere.

Most studies dealing with toxicity or physiological effects could not be used because the authors did not report clearly defined endpoints (i.e., LC50, EC50, statistically significant adverse effects): Apostol (1973), Baker, et al. (1983), Behan, et al. (1979), Belding (1927), Carpenter (1925), Crandall and Goodnight (1962), Dawson (1935), Dilling, et al. (1926), Dilling and Healy (1927), Ellis (1937), Ferguson and Bubela (1974), Fujiya (1961), Jackim (1973), Jackim, et al. (1970), Johnson and Eaton (1980), Jones (1935, 1947a,b), Laube, et al. (1980), Lloyd (1961), Lu, et al. (1975), Manalis and Cooper (1973), Manalis, et al. (1984), Merlini and Pozzi (1977b), Metayer, et al. (1982), Narbonne, et al. (1973), O'Neill (1981), Overnell (1975), Phillips (1980), Rao and Subramanian (1982), Rathore and Swarup (1978), Rice, et al. (1973), Ruchven and Cairns (1973), Ryck and Whitley (1974), Schulze and Brand (1978), Stratford, et al. (1984), Thomas, et al. (1980), Tucker and Matte (1980), Van der Werff and Pruyt (1982), Varansai and Gmur (1978), Varansai, et al. (1975), Watling (1981), Westfall (1945), and Wiener and Giesy (1979).

Some results were not used because the test was either improperly designed for deriving criteria or important details were omitted from the

report: Ferard, et al. (1982), Foster (1982a,b), Gentile, et al. (1982), Marion and Denizeau (1983), Passino and Corant (1979), Say and Whitton (1983), Vighi (1981), Wehr and Whitton (1983a,b), and Whitton, et al. (1982). Dorfman and Whitworth (1969) exposed brook trout to lead only on week days and the concentrations were not measured during tests lasting up to 38 days. These authors and Carpenter (1927), Rushton (1922), and Tarzwell and Henderson (1960) conducted tests with only one or two fish at a time. Rainbow trout tested by Hodson, et al. (1973b) were not acclimated to abrupt changes in pH before stressing them with lead. Experiments reported by Hodson, et al. (1982) were designed to measure lead uptake in opercular bone and formation of black tails correlated to different growth rates of rainbow trout; however, these fish were only exposed to one sublethal concentration of lead. No data are available on the concentrations of lead in water during the studies reported by Hodson, et al. (1983a). Sicko-Goad (1982), Sicko-Goad and Lazinsky (1981, 1982), and Sicko-Goad and Scoermer (1979) exposed algae to only one sublethal concentration of lead. The 96-hr values reported by Buikema, et al. (1974a,b) were subject to error because of possible reproductive interactions (Buikema, et al. 1977). Clarke and Clarke (1974) reported that their test water was contaminated with lead leached from plastic exposure tanks. Exposure times were not reported by Brown (1976) and Haider (1964). Kariya, et al. (1969) and Turnbull (1954) failed to report the number of fish tested. High control mortalities occurred in all except one test reported by Sauter, et al. (1976). Control mortality exceeded 10 percent in two tests by Mount and Norberg (1984).

English, et al. (1963) published results based on volume dilutions instead of nominal or measured concentrations. Brown (1968), Garavini and

Martelli (1979), Pawlaczyk-Szpilowa and Slowik (1981), Rao and Saxena (1980), and Rolfe, et al. (1977) exposed algae, invertebrates, and fish to lead but failed to adequately describe their test methods. Carpenter (1926, 1930), Carter and Cameron (1973), Ellgaard and Rudner (1982), Ellis (1940), Grande and Andersen (1983), Jones (1938, 1939), Nyman (1981), Ozoh (1979), Rathore, et al. (1979), Shaw and Grushkin (1957), Shaw and Lowrance (1956), Vijaymadhavan and Iwai (1975), Wang (1959), and Weir and Hine (1970) conducted tests in distilled, deionized, chlorinated, or "tap" water.

Biegert and Valkovic (1980) expressed their acute data in hours to death and concentrations were a factor of ten apart. The concentrations of lead overlapped in the tests by Sparks, et al. (1983). Tests on the toxicity of lead to algae were not used if the medium contained too much of a complexing agent such as EDTA (Davis, 1978).

Results of laboratory bioconcentration tests were not used if the test was not flow-through (Montgomery, et al. 1978; Watling, 1983), if the test did not last long enough (Wong, et al. 1981), if no soft tissues were analyzed (Sturesson, 1978), if the concentration in water was not known (Ray, et al. 1981) or was not measured often enough (Freeman, 1978, 1980), or if control mortalities were high (Valiela, et al. 1974). Studies such as those by Ancellin, et al. (1973), Aubert, et al. (1974), and Nash, et al. (1981), which used radioactive isotopes of lead, were not used because of the possibility of isotope discrimination. Newman and McIntosh (1983b) conducted a depuration study, but not an uptake study.

A large number of reports on lead toxicity and residues in wild aquatic organisms could not be used for the calculation of bioaccumulation factors or toxicity due to an insufficient number of measurements of the concentration

of lead in the water: Anderson (1977), Badsha and Goldspink (1982), Brezina and Arnold (1977), Brezina, et al. (1974), Brown and Chow (1977), Eide and Myklesstad (1980), Enk and Mathis (1977), Evans and Lasenby (1983), Gale, et al. (1973a,b, 1982), Gordon, et al. (1980), Holm (1980), Kharkar, et al. (1976), Knowlton, et al. (1983), Leland and McNurney (1974), Lucus and Edgington (1970), Marcin and Mudre (1982), Martin, et al. (1984), Mathis and Cummings (1973), Mathis and Kevern (1975), May and McKinney (1981), Mehrle, et al. (1982), Newman and McInrosh (1983a), Pagenkopf and Newman (1974), Pakkala, et al. (1972), Pennington, et al. (1982), Popham and D'Auria (1981), Price and Knight (1978), Randall, et al. (1981), Ray (1978), Sidwell, et al. (1978), Simpson (1979), Smith, et al. (1981), Tong, et al. (1974), Trollope and Evans (1976), Tsui and McCart (1981), Uche and Bligh (1971), Vinikour, et al. (1980), Wachs (1982), Walsh, et al. (1977), Welsh and Denny (1980), Wixson and Bolter (1972), and Wren, et al. (1983).

Summary

The acute toxicity of lead to several species of freshwater animals has been shown to decrease as the hardness of water increases. At a hardness of 50 mg/L the acute sensitivities of ten species range from 142.5 $\mu\text{g/L}$ for an amphipod to 235,900 $\mu\text{g/L}$ for a midge. Data on the chronic effects of lead on freshwater animals are available for two fish and two invertebrate species. The chronic toxicity of lead also decreases as hardness increases and the lowest and highest available chronic values (12.26 and 128.1 $\mu\text{g/L}$) are both for a cladoceran, but in soft and hard water, respectively. Acute-chronic ratios are available for three species and range from 18 to 62. Freshwater algae are affected by concentrations of lead above 500 $\mu\text{g/L}$, based

on data for four species. Bioconcentration factors are available for four invertebrate and two fish species and range from 42 to 1,700.

Acute values are available for 13 saltwater animal species and range from 315 $\mu\text{g/L}$ for the mummichog to 27,000 $\mu\text{g/L}$ for the soft-shell clam. A chronic toxicity test was conducted with a mysid; unacceptable effects were observed at 37 $\mu\text{g/L}$ but not at 17 $\mu\text{g/L}$ and the acute-chronic ratio for this species is 124.8. A species of macroalgae was affected at 20 $\mu\text{g/L}$. Available bioconcentration factors range from 17.5 to 2,570.

National Criteria

The procedures described in the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" indicate that, except possibly where a locally important species is very sensitive, freshwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration (in $\mu\text{g/L}$) of lead does not exceed the numerical value given by $e^{(1.273[\ln(\text{hardness})]-4.705)}$ more than once every three years on the average and if the one-hour average concentration (in $\mu\text{g/L}$) does not exceed the numerical value given by $e^{(1.273[\ln(\text{hardness})]-1.460)}$ more than once every three years on the average. For example, at hardnesses of 50, 100, and 200 mg/L as CaCO_3 the four-day average concentrations of lead are 1.3, 3.2, and 7.7 $\mu\text{g/L}$, respectively, and the one-hour average concentrations are 34, 82, and 200 $\mu\text{g/L}$.

The procedures described in the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" indicate that, except possibly where a locally important species is very sensitive, saltwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of lead does not

exceed 5.6 $\mu\text{g}/\text{L}$ more than once every three years on the average and if the one-hour average concentration does not exceed 140 $\mu\text{g}/\text{L}$ more than once every three years on the average.

EPA believes that a measurement such as "acid-soluble" would provide a more scientifically correct basis upon which to establish criteria for metals. The criteria were developed on this basis. However, at this time, no EPA approved methods for such a measurement are available to implement the criteria through the regulatory programs of the Agency and the States. The Agency is considering development and approval of methods for a measurement such as "acid-soluble". Until available, however, EPA recommends applying the criteria using the total recoverable method. This has two impacts: (1) certain species of some metals cannot be analyzed directly because the total recoverable method does not distinguish between individual oxidation states, and (2) these criteria may be overly protective when based on the total recoverable method.

The recommended exceedence frequency of three years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to lead exceeds the criterion. Stressed systems, for example, one in which several outfalls occur in a limited area, would be expected to require more time for recovery. The resilience of ecosystems and their ability to recover differ greatly, however, and site-specific criteria may be established if adequate justification is provided.

The use of criteria in designing waste treatment facilities requires the selection of an appropriate wasteload allocation model. Dynamic models are preferred for the application of these criteria. Limited data or other

factors may make their use impractical, in which case one should rely on a steady-state model. The Agency recommends the interim use of 1Q5 or 1Q10 for Criterion Maximum Concentration (CMC) design flow and 7Q5 or 7Q10 for the Criterion Continuous Concentration (CCC) design flow in steady-state models for unstressed and stressed systems respectively. These matters are discussed in more detail in the Technical Support Document for Water Quality-Based Toxics Control (U.S. EPA, 1985).

Table 1. Acute Toxicity of Lead to Aquatic Animals

<u>Species</u>	<u>Method</u> ^a	<u>Chemical</u>	<u>Hardness</u> (mg/L as CaCO ₃)	<u>LC50</u> or <u>EC50</u> (µg/L) ^{b,c}	<u>Species Mean</u> <u>Acute Value</u> (µg/L) ^{b,c}	<u>Reference</u>
<u>FRESHWATER SPECIES</u>						
Snail, <u>Aplexa hypnorum</u>	FT, M	Lead nitrate	61	1,340	1,040	Call, et al. 1981
Cladoceran, <u>Daphnia magna</u>	S, U	Lead chloride	-	931	-	Anderson, 1948
Cladoceran, <u>Daphnia magna</u>	S, U	Lead nitrate	-	5,000 ^{****}	-	Bringmann & Kuhn, 1959a,b
Cladoceran, <u>Daphnia magna</u>	R, M	Lead nitrate	54	612	-	Chapman, et al. Manuscript
Cladoceran, <u>Daphnia magna</u>	R, M	Lead nitrate	110	952	-	Chapman, et al. Manuscript
Cladoceran, <u>Daphnia magna</u>	R, M	Lead nitrate	152	1,910	447.8	Chapman, et al. Manuscript
Cladoceran, <u>Daphnia pulex</u>	S, U	Lead nitrate	45	5,100 ^{*****}	-	Mount & Norberg, 1984
Cladoceran, <u>Simocephalus vetulus</u>	S, U	Lead nitrate	45	4,500 ^{*****}	-	Mount & Norberg, 1984
Amphipod, <u>Gammarus pseudolimnaeus</u>	FT, M	Lead nitrate	46	124	-	Spehar, et al. 1978
Amphipod, <u>Gammarus pseudolimnaeus</u>	FT, M	Lead nitrate	48	140	142.6	Call, et al. 1983
Crayfish, <u>Orconectes limosus</u>	S, M	Lead chloride	-	3,300	-	Houtet & Chalermartin, 1973
Midge, <u>Tanytarsus dissimilis</u>	FT, M	Lead nitrate	48	224,000	235,900	Call, et al. 1983
Rainbow trout (2 mos), <u>Salmo gairdneri</u>	FT, M	Lead nitrate	-	8,000	-	Hale, 1977

Table 1. (Continued)

<u>Species</u>	<u>Method^a</u>	<u>Chemical</u>	<u>Hardness (mg/L as CaCO₃)</u>	<u>LC50 or EC50 (µg/L)^{b,c}</u>	<u>Species Mean Acute Value (µg/L)^{b,c}</u>	<u>Reference^d</u>
<u>Rainbow trout, Salmo gairdneri</u>	S, U	Lead nitrate	290	542,000	-	Goettl, et al. 1972; Davies & Everhart, 1973; Davies, et al. 1976
<u>Rainbow trout, Salmo gairdneri</u>	S, U	Lead nitrate	553	471,000	-	Goettl, et al. 1972; Davies & Everhart, 1973; Davies, et al. 1976
<u>Rainbow trout, Salmo gairdneri</u>	FT, U	Lead nitrate	28	1,170	2,448†	Goettl, et al. 1972; Davies & Everhart, 1973; Davies, et al. 1976
<u>Brook trout (18 mos), Salvelinus fontinalis</u>	FT, M	Lead nitrate	44	4,100	4,820	Holcombe, et al. 1976
<u>Goldfish, Carassius auratus</u>	S, U	Lead chloride	20	51,500	101,100	Pickering & Henderson, 1966
<u>Fathead minnow, Pimephales promelas</u>	S, U	Lead chloride	20	5,580	-	Pickering & Henderson, 1966
<u>Fathead minnow, Pimephales promelas</u>	S, U	Lead chloride	20	7,330	-	Pickering & Henderson, 1966
<u>Fathead minnow, Pimephales promelas</u>	S, U	Lead chloride	360	482,000	25,440	Pickering & Henderson, 1966
<u>Mosquitofish (adult), Gambusia affinis</u>	S, U	Lead nitrate	-	240,000††	-	Wallen, et al. 1957
<u>Guppy (6 mos), Poecilia reticulata</u>	S, U	Lead chloride	20	20,600	66,140	Pickering & Henderson, 1966
<u>Bluegill, Lepomis macrochirus</u>	S, U	Lead chloride	20	25,800	-	Pickering & Henderson, 1966
<u>Bluegill, Lepomis macrochirus</u>	S, U	Lead chloride	360	442,000	52,310	Pickering & Henderson, 1966

Table 1. (Continued)

<u>Species</u>	<u>Method^a</u>	<u>Chemical</u>	<u>Hardness (mg/L as CaCO₃)</u>	<u>LC50 or EC50 (µg/L)^{b,c}</u>	<u>Species Mean Acute Value (µg/L)^{d,e}</u>	<u>Reference</u>
<u>SALTWATER SPECIES</u>						
<u>Blue mussel (larva), Mytilus edulis</u>	S, U	Lead nitrate	-	476		Martin, et al. 1981
<u>Pacific oyster (larva), Crassostrea gigas</u>	S, U	Lead nitrate	-	758	758	Martin, et al. 1981
<u>Eastern oyster (larva), Crassostrea virginica</u>	S, U	Lead nitrate	-	2,450	2,450	Calabrese, et al. 1973
<u>Quahog clam (larva), Mercenaria mercenaria</u>	S, U	Lead nitrate	-	780	780	Calabrese & Nelson, 1974
<u>Soft-shell clam (adult), Mya arenaria</u>	S, U	Lead nitrate	-	27,000	27,000	Eisler, 1977
<u>Copepod, Acartia clausi</u>	S, U	Lead nitrate	-	668	668	Gentile, 1982
<u>Mysid, Mysidopsis bahia</u>	FT, M	Lead nitrate	-	3,130	3,130	Lussler, et al. Manuscript
<u>Amphipod, Ampelisca abdita</u>	R, U	Lead nitrate	-	547	547	Scott, et al. Manuscript
<u>Dungeness crab, Cancer magister</u>	S, U	Lead nitrate	-	575	575	Martin, et al. 1981
<u>Sheepshead minnow, Cyprinodon variegatus</u>	FT, M	Lead nitrate	-	>3,140	>3,140	Cardin, 1981
<u>Mummichog, Fundulus heteroclitus</u>	S, U	Lead nitrate	-	315	315	Dortman, 1977
<u>Inland silverside, Menidia beryllina</u>	FT, M	Lead nitrate	-	>3,140	>3,140	Cardin, 1981
<u>Atlantic silverside, Menidia menidia</u>	S, U	Lead nitrate	-	>10,000	>10,000	Burry, 1981

Table 1. (Continued)

- S = static, R = renewal, FT = flow-through, M = measured, U = unmeasured.
- Results are expressed as lead, not as the chemical.
- Freshwater Species Mean Acute Values are calculated at a hardness of 50 mg/L using the pooled slope.
- In river water.
- Not used in calculations because the values in Mount and Norberg (1984) are much higher than values for other species in the same genus and family.
- † Calculated from acute value of 1,170 µg/L using pooled slope (see text).
- †† High turbidity.

Results of Covariance Analysis of Freshwater Acute Toxicity versus Hardness

<u>Species</u>	<u>n</u>	<u>Slope</u>	<u>95% Confidence Limits</u>	<u>Degrees of Freedom</u>
Daphnia magna	3	1.021	-3.592, 5.634	1
Rainbow trout	3	2.475	-0.357, 5.308	1
Fathead minnow	3	1.495	0.458, 2.533	1
Bluegill	2	1.011	(cannot be calculated)	0
All of above	11	1.608*	1.014, 2.202	6
All of above except rainbow trout	8	1.273**	0.909, 1.637	4

- * P=0.03 for equality of slopes.
- ** P=0.16 for equality of slopes.

Table 2. Chronic Toxicity of Lead to Aquatic Animals

<u>Species</u>	<u>Test*</u>	<u>Chemical</u>	<u>Hardness (mg/L as CaCO₃)</u>	<u>Limits (µg/L)**</u>	<u>Chronic Value (µg/L)**</u>	<u>Reference</u>
<u>FRESHWATER SPECIES</u>						
<u>Snail, Lymnaea palustris</u>	LC	Lead nitrate	139	12-54	75.46	Borgmann, et al. 1978
<u>Cladoceran, Daphnia magna</u>	LC	Lead nitrate	52	9-16.7	12.26	Chapman, et al. Manuscript
<u>Cladoceran, Daphnia magna</u>	LC	Lead nitrate	102	78-181	118.8	Chapman, et al. Manuscript
<u>Cladoceran, Daphnia magna</u>	LC	Lead nitrate	151	85-193	128.1	Chapman, et al. Manuscript
<u>Rainbow trout, Salmo gairdneri</u>	ELS	Lead nitrate	28	13.2-27	18.88	Goettl, et al. 1972; Davies & Everhart, 1973; Davies, et al. 1976
<u>Rainbow trout, Salmo gairdneri</u>	ELS	Lead nitrate	35	71-146	101.8	Sauter, et al. 1976
<u>Brook trout, Salvelinus fontinalis</u>	LC	Lead nitrate	44	58-119	83.08	Holcombe, et al. 1976
<u>SALTWATER SPECIES</u>						
<u>Mysid, Mysidopsis bahia</u>	LC	Lead nitrate	-	17-37	25.08	Lussler, et al. Manuscript

* LC = life cycle or partial life cycle, ELS = early life stage.

**Results are expressed as lead, not as the chemical.

Results of Regression Analysis of Freshwater Chronic Toxicity versus Hardness

<u>Species</u>	<u>n</u>	<u>Slope</u>	<u>95% Confidence Limits</u>	<u>Degrees of Freedom</u>
<u>Daphnia magna</u>	3	2.328	-0.274, 12.931	1

23

Table 2. (Continued)

<u>Species</u>	<u>Acute-Chronic Ratio</u>			<u>Ratio</u>	
	<u>Hardness (mg/L as CaCO₃)</u>	<u>Acute Value (µg/L)</u>	<u>Chronic Value (µg/L)</u>		
<u>Cladoceran, Daphnia magna</u>	52-54	612	12.26	49.92	
<u>Cladoceran, Daphnia magna</u>	102-110	952	118.8	8.013	18.13
<u>Cladoceran, Daphnia magna</u>	151-152	1,910	128.1	14.91	
<u>Rainbow trout, Salmo gairdneri</u>	28	1,170	18.88	61.97	61.97
<u>Brook trout, Salvelinus fontinalis</u>	44	4,100	83.08	49.35	49.35
<u>Mysid, Mysidopsis bahia</u>	-	3,130	25.08	124.8	124.8

Table 3. Ranked Genus Mean Acute Values with Species Mean Acute-Chronic Ratios

<u>Rank^a</u>	<u>Genus Mean Acute Value (µg/L)^{bb}</u>	<u>Species</u>	<u>Species Mean Acute Value (µg/L)^{bb}</u>	<u>Species Mean Acute-Chronic Ratio</u>
<u>FRESHWATER SPECIES</u>				
10	235,900	Midge, <u>Tanytarsus dissimilis</u>	235,900	-
9	101,100	Goldfish, <u>Carassius auratus</u>	101,100	-
8	66,140	Guppy, <u>Poecilia reticulata</u>	66,140	-
7	52,310	Bluegill, <u>Lepomis macrochirus</u>	52,310	-
6	25,440	Fathead minnow, <u>Pimephales promelas</u>	25,440	-
5	4,820	Brook trout, <u>Salvelinus fontinalis</u>	4,820	49.35
4	2,448	Rainbow trout, <u>Salmo gairdneri</u>	2,448	61.97
3	1,040	Snail, <u>Aplexa hypnorum</u>	1,040	-
2	447.8	Cladoceran, <u>Daphnia magna</u>	447.8	18.13***
1	142.6	Amphipod, <u>Gammarus pseudolimnaeus</u>	142.6	-
<u>SALTWATER SPECIES</u>				
11	27,000	Soft-shell clam, <u>Mya arenaria</u>	27,000	-

Insect

Salmonid

Plant based

Detritus

Table 3. (Continued)

<u>Rank[#]</u>	<u>Genus Mean Acute Value ($\mu\text{g/L}$)^{##}</u>	<u>Species</u>	<u>Species Mean Acute Value ($\mu\text{g/L}$)^{##}</u>	<u>Species Mean Acute-Chronic Ratio</u>
10	>5,604	Inland silverside, <u>Menidia beryllina</u>	>3,140	-
		Atlantic silverside, <u>Menidia menidia</u>	>10,000	-
9	>3,140	Sheepshead minnow, <u>Cyprinodon variegatus</u>	>3,140	-
8	3,130	Mysid, <u>Mysidopsis bahia</u>	3,130	124.8
7	1,363	Pacific oyster, <u>Crassostrea gigas</u>	758	-
		Eastern oyster, <u>Crassostrea virginica</u>	2,450	-
6	780	Quahog clam, <u>Mercenaria mercenaria</u>	780	-
5	668	Copepod, <u>Acartia clausi</u>	668	-
4	575	Dungeness crab, <u>Cancer magister</u>	575	-
3	547	Amphipod, <u>Ampelisca abdita</u>	547	-
2	476	Blue mussel, <u>Mytilus edulis</u>	476	-
1	315	Mummichog, <u>Fundulus heteroclitus</u>	315	-

Table 3. (Continued)

- * Ranked from most resistant to most sensitive based on Genus Mean Acute Value.
- ** Freshwater Genus Mean Acute Values and Species Mean Acute Values are at a hardness of 50 mg/L.
- ***Geometric mean of three values in Table 2.

Fresh water

Final Acute Value = 67.54 µg/L (at a hardness of 50 mg/L)

Criterion Maximum Concentration = $(67.54 \text{ µg/L}) / 2 = 33.77 \text{ µg/L}$ (at a hardness of 50 mg/L)

Pooled Slope = 1.273 (see Table 1)

$\ln(\text{Criterion Maximum Intercept}) = \ln(33.77) - (\text{slope} \times \ln(50))$

$= 3.520 - (1.273 \times 3.912) = -1.460$

Criterion Maximum Concentration = $e^{(1.273|\ln(\text{hardness})|-1.460)}$

Final Acute-Chronic Ratio = 51.29 (see text)

Final Chronic Value = $(67.54 \text{ µg/L}) / 51.29 = 1.317 \text{ µg/L}$ (at a hardness of 50 mg/L)

$\ln(\text{Final Chronic Intercept}) = \ln(1.317) - (\text{slope} \times \ln(50))$

$= 0.2754 - (1.27 \times 3.912) = -4.705$

Final Chronic Value = $e^{(1.273|\ln(\text{hardness})|-4.70)}$

Salt water

Final Acute Value = 287.4 µg/L

Criterion Maximum Concentration = $(287.4 \text{ µg/L}) / 2 = 143.7 \text{ µg/L}$

Final Acute-Chronic Ratio = 51.29 (see text)

Final Chronic Value = $(287.4 \text{ µg/L}) / 51.29 = 5.603 \text{ µg/L}$

Table 4. Toxicity of Lead to Aquatic Plants

<u>Species</u>	<u>Chemical</u>	<u>Hardness (mg/L as CaCO₃)</u>	<u>Effect</u>	<u>Result (µg/L)*</u>	<u>Reference</u>
<u>FRESHWATER SPECIES</u>					
Alga, <u>Ankistrodesmus</u> sp.	Lead chloride	-	24% growth inhibition	1,000	Monahan, 1976
Alga, <u>Ankistrodesmus falcatus</u>	Lead chloride	-	60% growth inhibition	2,500	Devi Prasad & Devi Prasad, 1982
Alga, <u>Chlorella</u> sp.	Lead chloride	-	53% growth inhibition	500	Monahan, 1976
Alga, <u>Chlorella saccharophila</u>	Lead chloride	-	EC50	63,800	Rachlin, et al. 1982
Alga, <u>Chlorococcum</u> sp.	Lead chloride	-	71% growth inhibition	2,500	Devi Prasad & Devi Prasad, 1982
Alga, <u>Scenedesmus</u> sp.	Lead chloride	-	35% growth inhibition	500	Monahan, 1976
Alga, <u>Scenedesmus obliquus</u>	Lead chloride	-	72% growth inhibition	2,500	Devi Prasad & Devi Prasad, 1982
Alga, <u>Selenastrum</u> sp.	Lead chloride	-	52% growth inhibition	500	Monahan, 1976
Diatom, <u>Navicula incerta</u>	Lead chloride	-	EC50	10,960	Rachlin, et al. 1983
Eurasian watermilfoil, <u>Myriophyllum spicatum</u>	-	-	32-day EC50 (root growth)	363,000	Stanley, 1974
<u>SALTWATER SPECIES</u>					
Alga, <u>Champia parvula</u>	Lead nitrate	-	Stopped sexual reproduction	20.3	Steele & Thursby, 1983
Alga, <u>Champia parvula</u>	Lead nitrate	-	Reduced female growth	20.3	Steele & Thursby, 1983

Table 4. (Continued)

<u>Species</u>	<u>Chemical</u>	<u>Hardness (mg/L as CaCO₃)</u>	<u>Effect</u>	<u>Result (µg/L)^a</u>	<u>Reference</u>
Alga, <u>Champia parvula</u>	Lead nitrate	-	Reduced tetra- sporangia production	23.3	Steele & Thursby, 1983
Alga, <u>Champia parvula</u>	Lead nitrate	-	Reduced tetra- sporophyte growth	23.3	Steele & Thursby, 1983
Alga, <u>Dunaliella salina</u>	Lead nitrate	-	65% growth reduction	900	Pace, et al. 1977
Diatom, <u>Ditylum brightwellii</u>	Lead chloride	-	EC50	40	Canterford & Canterford, 1980
Diatom, <u>Asterionella japonica</u>	Lead nitrate	-	EC50	207	Fisher & Jones, 1981

^a Results are expressed as lead, not as the chemical. All results are based on unmeasured concentrations.

Table 5. Bioaccumulation of Lead by Aquatic Organisms

<u>Species</u>	<u>Tissue</u>	<u>Chemical</u>	<u>Duration (days)</u>	<u>Bioconcentration Factor^a</u>	<u>Reference</u>
<u>FRESHWATER SPECIES</u>					
<u>Snail, Lymnaea palustris</u>	Whole body	Lead nitrate	120	1,700 ^{aa}	Dorgmann, et al. 1978
<u>Snail, Physa integra</u>	Whole body	Lead nitrate	28	738 ^{aa}	Spehar, et al. 1978
<u>Stonefly, Pteronarcys dorsata</u>	Whole body	Lead nitrate	28	1,120 ^{aa}	Spehar, et al. 1978
<u>Caddisfly, Brachycentrus sp.</u>	Whole body	Lead nitrate	28	499 ^{aa}	Spehar, et al. 1978
<u>Brook trout (embryo-3 mos), Salvelinus fontinalis</u>	Whole body	Lead nitrate	140	42 ^{aa}	Holcombe, et al. 1976
<u>Bluegill, Lepomis macrochirus</u>	Whole body	-	- ^{aaa}	45 ^{aa}	Atchison, et al. 1977
<u>SALTWATER SPECIES</u>					
<u>Diatom, Ditylum brightwellii</u>	Cells	Lead chloride	14	725 ^{aa}	Canterford, et al. 1978
<u>Blue mussel, Mytilus edulis</u>	Soft parts	Lead nitrate	40	650 ^{aa}	Schulz-Baldes, 1974
<u>Blue mussel, Mytilus edulis</u>	Soft parts	Lead chloride	37	200 ^{aa}	Talbot, et al. 1976
<u>Blue mussel, Mytilus edulis</u>	Soft parts	Lead nitrate	130	2,570 ^{aa}	Schulz-Baldes, 1972
<u>Blue mussel, Mytilus edulis</u>	Soft parts	Lead nitrate	130	2,080 ^{aa}	Schulz-Baldes, 1972
<u>Blue mussel, Mytilus edulis</u>	Soft parts	Lead nitrate	130	796 ^{aa}	Schulz-Baldes, 1972
<u>Eastern oyster, Crassostrea virginica</u>	Soft parts	Lead nitrate	140	536	Zarogyan, et al. 1979

Table 5. (Continued)

<u>Species</u>	<u>Tissue</u>	<u>Chemical</u>	<u>Duration (days)</u>	<u>Bioconcentration Factor^a</u>	<u>Reference</u>
Eastern oyster, <u>Crassostrea virginica</u>	Soft parts	Lead nitrate	49	68 ^{ab}	Pringle, et al. 1968
Eastern oyster, <u>Crassostrea virginica</u>	Soft parts	Lead nitrate	70	1,400	Shuster & Pringle, 1969
Quahog clam, <u>Mercenaria mercenaria</u>	Soft parts	Lead nitrate	56	17.5 ^{ab}	Pringle, et al. 1968

^a Results are based on lead, not the chemical.

^{ab} Bioconcentration factor was converted from dry weight to wet weight basis.

^{ab} This field study was conducted with a natural population of bluegills living in a small lake which was extensively analyzed for lead, zinc, and cadmium.

Table 6. Other Data on Effects of Lead on Aquatic Organisms

<u>Species</u>	<u>Chemical</u>	<u>Hardness (mg/L as CaCO₃)</u>	<u>Duration</u>	<u>Effect</u>	<u>Result (µg/L)*</u>	<u>Reference</u>
<u>FRESHWATER SPECIES</u>						
Green alga, <u>Scenedesmus quadricauda</u>	Lead nitrate	-	96 hrs	Incipient inhibition	2,500**	Bringmann & Kuhn, 1959a,b
Blue alga, <u>Microcystis aeruginosa</u>	Lead acetate	-	8 days	Incipient inhibition	450	Bringmann, 1975; Bringmann & Kuhn, 1976, 1978a,b
Green alga, <u>Scenedesmus quadricauda</u>	Lead acetate	-	8 days	Incipient inhibition	3,700	Bringmann & Kuhn, 1977a, 1978a,b, 1979, 1980b
Alga, <u>Anabaena sp.</u>	Lead nitrate	-	24 hrs	50% reduction of ¹⁴ C ₂ fixation	15,000 26,000 15,000	Malanchuk & Gruending, 1973
Alga, <u>Chlamydomonas sp.</u>	Lead nitrate	-	24 hrs	50% reduction of ¹⁴ C ₂ fixation	17,000 17,000	Malanchuk & Gruending, 1973
Angiosperm, <u>Potamogeton pectinatus</u>	Lead acetate	-	3 days	Reduced respiration	325,200	Jana & Choudhuri, 1982
Angiosperm, <u>Vallisneria spiralis</u>	Lead acetate	-	3 days	Reduced respiration	3,252,000	Jana & Choudhuri, 1982
Desmid, <u>Cosmarium sp.</u>	Lead nitrate	-	24 hrs	50% reduction of ¹⁴ C ₂ fixation	5,000 5,000 5,000	Malanchuk & Gruending, 1973
Diatom, <u>Navicula sp.</u>	Lead nitrate	-	24 hrs	50% reduction of ¹⁴ C ₂ fixation	17,000 28,000 17,000	Malanchuk & Gruending, 1973
Bacterium, <u>Escherichia coli</u>	Lead nitrate	-	-	Incipient inhibition	1,300	Bringmann & Kuhn, 1959a
Bacterium, <u>Pseudomonas putida</u>	Lead acetate	-	16 hrs	Incipient inhibition	1,800	Bringmann & Kuhn, 1976, 1977a, 1979, 1980b
Protozoan, <u>Entosiphon sulcatum</u>	Lead acetate	-	72 hrs	Incipient inhibition	20	Bringmann, 1978; Bringmann & Kuhn, 1979, 1980b, 1981

Table 6. (Continued)

Species	Chemical	Hardness (mg/L as CaCO ₃)	Duration	Effect	Result (µg/L) [#]	Reference
Protozoan, <u>Microregma heterostoma</u>	Lead nitrate	-	28 hrs	Incipient inhibition	1,250	Bringmann & Kuhn, 1959b
Protozoan, <u>Chilomonas paramecium</u>	Lead acetate	-	48 hrs	Incipient inhibition	220	Bringmann, et al. 1980, 1981
Protozoan, <u>Uronema parduezi</u>	Lead acetate	-	20 hrs	Incipient inhibition	70	Bringmann & Kuhn, 1980a, 1981
Tubificid worm, <u>Tubifex tubifex</u>	Lead nitrate	224	48 hrs	LC50	450,000	Qureshi, et al. 1980
Tubificid worm, <u>Tubifex sp.</u>	Lead nitrate	-	24 hrs	LC50	49,000	Whitley, 1968
Tubificid worm, <u>Tubifex sp.</u>	Lead nitrate	-	24 hrs	LC50	27,500	Whitley, 1968
Snail, <u>Goniobasis ilvoscens</u>	Lead acetate	-	48 hrs	LC50	71,000	Calrns, et al. 1976
33 Snail, <u>Lymnaea emarginata</u>	Lead acetate	-	48 hrs	LC50	14,000	Calrns, et al. 1976
Snail, <u>Physa integra</u>	Lead nitrate	46	28 days	No effect on survival	565	Spehar, et al. 1978
Cladoceran, <u>Daphnia magna</u>	Lead chloride	45	48 hrs	EC50 (fed) (immobilization)	450	Blesinger & Christensen, 1972
Cladoceran, <u>Daphnia magna</u>	Lead chloride	45	21 days	Reproductive impairment	30	Blesinger & Christensen, 1972
Cladoceran, <u>Daphnia magna</u>	Lead acetate	-	24 hrs	LC50	2,500	Bringmann & Kuhn, 1977b
Natural copepod assemblages	Lead nitrate	-	7 days	Reduced growth rate	2,320	Borgmann, 1980
Amphipod, <u>Gammarus pseudolimnaeus</u>	Lead nitrate	46	28 days	LC50	28.4	Spehar, et al. 1978
Crayfish, <u>Orconectes virilis</u>	Lead acetate	-	40 days	Increase in ventilation rate	500	Anderson, 1978

Table 6. (Continued)

<u>Species</u>	<u>Chemical</u>	<u>Hardness (mg/L as CaCO₃)</u>	<u>Duration</u>	<u>Effect</u>	<u>Result (µg/L)^a</u>	<u>Reference</u>
Mayfly, <u>EphemereUa grandis</u>	Lead nitrate	50	14 days	LC50	3,500	Nehring, 1976
Mayfly (nympn), <u>EphemereUa grandis</u>	Lead nitrate	50	14 days	BCF = 2,360	-	Nehring, 1976
Mayfly, <u>EphemereUa subvaria</u>	Lead sulfate	44	7 days	LC50	16,000	Warnick & Bell, 1969
Stonefly, <u>Pteronarcys californica</u>	Lead nitrate	50	14 days	BCF = 86	-	Nehring, 1976
Stonefly, <u>Pteronarcys dorsata</u>	Lead nitrate	46	28 days	No effect on survival	565	Spehar, et al. 1978
Caddisfly, <u>Hydropsyche betteni</u>	Lead sulfate	44	7 days	LC50	32,000	Warnick & Bell, 1969
Caddisfly, <u>Brachycentrus sp.</u>	Lead nitrate	46	28 days	No effect on survival	565	Spehar, et al. 1978
34 Midge, (embryo - 3rd instar), <u>Tanytarsus dissimilis</u>	Lead nitrate	47	10 days	LC50	258	Anderson, et al. 1980
Rainbow trout, <u>Salmo gairdneri</u>	Lead nitrate	135	28 days	Inhibition of ALA-D activity	13	Hodson, 1976
Rainbow trout (12 mos), <u>Salmo gairdneri</u>	-	135	14 days	Inhibition of ALA-D activity	10	Hodson, et al. 1977
Rainbow trout, <u>Salmo gairdneri</u>	Lead nitrate	135	21 days	LC50	2,400	Hodson, et al. 1978a
Rainbow trout, <u>Salmo gairdneri</u>	Lead nitrate	135	32 wks	Black-tails in 3 of 10 remaining fish	120	Hodson, et al. 1978a; Sippel, et al. 1983
Rainbow trout, <u>Salmo gairdneri</u>	Lead nitrate	135	32 wks	Affected RBC, Iron content, and ALA-D in blood	13	Hodson, et al. 1978a

Table 6. (Continued)

<u>Species</u>	<u>Chemical</u>	<u>Hardness (mg/L as CaCO₃)</u>	<u>Duration</u>	<u>Effect</u>	<u>Result (µg/L)*</u>	<u>Reference</u>
<u>Rainbow trout, Salmo gairdneri</u>	-	135	29 wks	All fish had black tails and decrease in ALA-D in blood	87	Hodson, et al. 1979a, 1980
<u>Rainbow trout, Salmo gairdneri</u>	Lead nitrate	135	30 wks	64% inhibition of ALA-D activity and black tails in 88% of fish	65	Hodson, et al. 1979b
<u>Rainbow trout, Salmo gairdneri</u>	Lead nitrate	135	20 days	45% inhibition of ALA-D activity	25	Hodson, et al. 1983b
<u>Rainbow trout (embryo, larva), Salmo gairdneri</u>	Lead chloride	101	28 days	EC50 (death and deformity)	220	Birge, et al. 1980
<u>Rainbow trout (embryo, larva), Salmo gairdneri</u>	Lead chloride	101	28 days	EC1 (death and deformity)	10.3	Birge, et al. 1980, 1981
<u>Rainbow trout (fingerling), Salmo gairdneri</u>	Lead nitrate	353	19 mos	Lordoscoliosis	850	Goetti, et al. 1972; Davies, et al. 1976
<u>Rainbow trout (sac fry), Salmo gairdneri</u>	Lead nitrate	28	19 mos	Lordoscoliosis	31	Goetti, et al. 1972; Davies, et al. 1976
<u>Brook trout, Salvelinus fontinalis</u>	-	-	21 days	Stamina	14	Adams, 1975
<u>Brook trout (12 mos), Salvelinus fontinalis</u>	Lead nitrate	135	14 days	Inhibition of ALA-D activity	90	Hodson, et al. 1977
<u>Brook trout (embryo - 21 day), Salvelinus fontinalis</u>	Lead chloride	44	38 days	Elevation of ALP and ACH activity	525	Christensen, 1975
<u>Brook trout (12 mos), Salvelinus fontinalis</u>	Lead chloride	44	56 days	Decrease of hemoglobin and inhibition of GOT activity	58	Christensen, et al. 1977
<u>Goldfish (embryo, larva), Carassius auratus</u>	Lead chloride	195	7 days	EC50 (death and deformity)	1,660	Birge, 1978
<u>Goldfish (<12 mos), Carassius auratus</u>	Lead nitrate	135	14 days	Inhibition of ALA-D activity	470	Hodson, et al. 1977

Table 6. (Continued)

<u>Species</u>	<u>Chemical</u>	<u>Hardness (mg/L as CaCO₃)</u>	<u>Duration</u>	<u>Effect</u>	<u>Result (µg/L)^a</u>	<u>Reference</u>
Common carp, <u>Cyprinus carpio</u>	Lead acetate	360	6 days	50% reduction in hatch	13,350	Kapur & Yadav, 1982
Red shiner, <u>Notropis lutrensis</u>	Lead nitrate	-	48 hrs	LC50 (high turbidity)	630,000	Wallen, et al. 1957
Fathead minnow, <u>Pimephales promelas</u>	Lead acetate	20	96 hrs	LC50	7,480	Pickering & Henderson, 1966
Fathead minnow, <u>Pimephales promelas</u>	Lead acetate	44	96 hrs	LC50	27,800	Curtis & Ward, 1981
Fathead minnow, <u>Pimephales promelas</u>	Lead fluoroborate	44	96 hrs	LC50	12,000	Curtis & Ward, 1981
Channel catfish (1.6 g), <u>Ictalurus punctatus</u>	Lead arsenate	45	96 hrs	LC50	>100,000	Johnson & Finley, 1980
Mosquitofish (adult), <u>Gambusia affinis</u>	Lead oxide	-	96 hrs	LC50 (high turbidity)	>56,000,000	Wallen, et al. 1957
Pumpkinseed (>12 mos), <u>Lepomis gibbosus</u>	Lead nitrate	135	14 days	Inhibition of ALA-D activity	90	Hodson, et al. 1977
Largemouth bass (embryo, larva), <u>Micropterus salmoides</u>	Lead chloride	99	8 days	EC50 (death and deformity)	240	Birge, et al. 1978
Largemouth bass, <u>Micropterus salmoides</u>	-	-	24 hrs	Affected oper- cular rhythm	1,050	Morgan, 1979
Leopard frog (adult), <u>Rana pipiens</u>	Lead nitrate	-	30 days	Death	100	Kaplan, et al. 1967
Narrow-mouthed toad (embryo, larva) <u>Gastrophryne carolinensis</u>	Lead chloride	195	7 days	EC50 (death and deformity)	40	Birge, 1978
Marbled salamander (embryo, larva), <u>Ambystoma opacum</u>	Lead chloride	99	8 days	EC50 (death and deformity)	1,460	Birge, et al. 1978

Table 6. (Continued)

<u>Species</u>	<u>Chemical</u>	<u>Hardness (mg/L as CaCO₃)</u>	<u>Duration</u>	<u>Effect</u>	<u>Result (µg/L)^a</u>	<u>Reference</u>
<u>SALTWATER SPECIES</u>						
<u>Alga, Laminaria digitata</u>	-	-	30-31 days	50-60% reduction in growth	1,000	Bryan, 1976
<u>Diatom, Phaeodactylum tricornutum</u>	Lead chloride	-	24 hrs	Completely inhibited photosynthesis	10,000	Woolery & Lewin, 1976
<u>Diatom, Phaeodactylum tricornutum</u>	Lead chloride	-	48-72 hrs	Reduced photosynthesis and respiration by 25-50%	100	Woolery & Lewin, 1976
<u>Diatom, Phaeodactylum tricornutum</u>	-	-	72 hrs	No growth inhibition	1,000	Hannan & Patouillet, 1972
<u>Diatom, Phaeodactylum tricornutum</u>	Lead chloride	-	1 hr	BCF = 1,050	-	Schulz-Baldes & Lewin, 1976
<u>Diatom, Skeletonema costatum</u>	Lead nitrate	-	12 days	EC50 (growth rate)	5.1	Rivkin, 1979
<u>Diatom, Skeletonema costatum</u>	Lead nitrate	-	12 days	EC50 (growth rate)	3.7	Rivkin, 1979
<u>Phytoplankton, Platymonas subcordiformis</u>	Lead chloride	-	72 hrs	Retarded population growth by delaying cell division	2,500	Hessler, 1974
<u>Phytoplankton, Platymonas subcordiformis</u>	Lead chloride	-	1 hr	BCF = 933	-	Schulz-Baldes & Lewin, 1976
<u>Phytoplankton, Platymonas subcordiformis</u>	Lead chloride	-	72 hrs	Death and inhibition of growth	60,000	Hessler, 1974
<u>Phytoplankton, Platymonas subcordiformis</u>	Lead chloride	-	2 days	48% of cells in culture died	2,500	Hessler, 1974
<u>Phytoplankton, Platymonas subcordiformis</u>	Lead chloride	-	6 days	98% of cells in culture died	60,000	Hessler, 1975

Table 6. (Continued)

<u>Species</u>	<u>Chemical</u>	<u>Hardness (mg/L as CaCO₃)</u>	<u>Duration</u>	<u>Effect</u>	<u>Result (µg/L)^a</u>	<u>Reference</u>
Alga, <u>Dunaliella tertiolecta</u>	Tetramethyl lead	-	96 hrs	EC50	1,650	Marchetti, 1978
Alga, <u>Dunaliella tertiolecta</u>	Tetraethyl lead	-	96 hrs	EC50	150	Marchetti, 1978
Alga, <u>Chlorella stigmatophora</u>	Lead acetate	-	21 days	50% growth inhibition	700	Christensen, et al. 1979
Natural phytoplankton populations	Lead chloride	-	5 days	Reduced chlorophyll a	207	Hollibaugh, et al. 1980
Natural phytoplankton populations	Lead chloride	-	4 days	Reduced biomass	21	Hollibaugh, et al. 1980
Macroalga, <u>Fucus serratus</u>	Lead acetate	-	-	45% growth inhibition	810	Stromgren, 1980
Ciliate protozoa, <u>Cristigera</u> sp.	Lead nitrate	-	12 hrs	Reduced growth rate by 8.5%	150	Gray & Ventilla, 1973
Ciliate protozoa, <u>Cristigera</u> sp.	Lead nitrate	-	12 hrs	Reduced growth rate by 11.7%	300	Gray & Ventilla, 1973
Polychaete worm, <u>Ophryotrocha diadema</u>	Lead acetate	-	96 hrs	LC50	14,100	Reish, et al. 1976
Polychaete worm, <u>Ophryotrocha diadema</u>	Lead acetate	-	21 days	Suppressed reproduction	1,000	Reish & Carr, 1978
Polychaete worm, <u>Ophryotrocha diadema</u>	Lead chloride	-	48 hrs	LC50	100,000	Parker, 1984
Polychaete worm, <u>Ctenodrilus serratus</u>	Lead acetate	-	21 days	Suppressed reproduction	1,000	Reish & Carr, 1978
Polychaete worm, <u>Capitella capitata</u>	Lead acetate	-	96 hrs	LC50	1,200	Reish, et al. 1976

Table 6. (Continued)

<u>Species</u>	<u>Chemical</u>	<u>Hardness (mg/L as CaCO₃)</u>	<u>Duration</u>	<u>Effect</u>	<u>Result (µg/L)*</u>	<u>Reference</u>
<u>Red abalone, Haliotis rufescens</u>	Lead chloride	-	6 mos	Accumulated 21 µg/g wet wt while exposed to a bioassay concentration of 100 µg/L with	-	Stewart & Schulz-Baldes, 1976
<u>Blue mussel, Mytilus edulis</u>	Lead chloride	-	40 days	LC50	30,000	Talbot, et al. 1976
<u>Blue mussel, Mytilus edulis</u>	Lead nitrate	-	150 days	LT50	500	Schulz-Baldes, 1972
<u>Eastern oyster, Crassostrea virginica</u>	Field study	-	1 yr	BCF = 326	-	Kopfler & Mayer, 1973
<u>Oyster, Unspecified</u>	Lead acetate	-	14 days	BCF = 1044	-	Stone, et al. 1981
<u>Soft-shell clam, Mya arenaria</u>	Lead nitrate	-	168 hrs	LC50	8,800	Eisler, 1977
<u>American lobster, Homarus americanus</u>	Lead nitrate	-	30 days	Reduced enzyme activity	50	Gould & Greig, 1983
<u>Mud crab, Rhithropanopeus harrisi</u>	Lead chloride	-	-	Delayed larval development	50	Benijts-Claus & Benijts, 1975
<u>Fiddler crab, Uca pugilator</u>	Lead nitrate	-	wks	BCF = 20	-	Wels, 1976
<u>Sea urchin, Arbacia punctulata</u>	Lead nitrate	-	-	Few gastrula developed	14	Waterman, 1937
<u>Mummichog (embryo), Fundulus heteroclitus</u>	Lead nitrate	-	-	Depressed axis formation	100	Wels & Wels, 1977
<u>Mummichog (embryo), Fundulus heteroclitus</u>	Lead nitrate	-	-	Retarded hatching	10,000	Wels & Wels, 1982

Table 6. (Continued)

<u>Species</u>	<u>Chemical</u>	<u>Hardness</u> (mg/L as CaCO ₃)	<u>Duration</u>	<u>Effect</u>	<u>Result</u> (µg/L) ^a	<u>Reference</u>
Shiner perch, <u>Cymatogaster aggregata</u>	Lead nitrate	-	-	27% inhibition of brain cholinesterase	7.8	Abou-Donia & Menzel, 1967

^a Results are expressed as lead, not as the chemical.

^{##} in river water.

REFERENCES

- Abou-Donia, M.B. and D.B. Menzel. 1967. Fish brain cholinesterase: its inhibition by carbamates and automatic assay. *Comp. Biochem. Physiol.* 21: 99.
- Adams, E.S. 1975. Effect of lead and hydrocarbons from snowmobile exhaust on brook trout (Salvelinus fontinalis). *Trans. Am. Fish. Soc.* 104: 363.
- Ancellin, J., et al. 1973. Aspects of biologiques et physico-chimiques de la contamination radioactive d'espèces et de sédiments marins. In: *Radioactive Contamination of the Marine Environment*. International Atomic Energy Agency, Vienna, Austria. p. 225.
- Anderson, B.G. 1948. The apparent thresholds of toxicity to Daphnia magna for chlorides of various metals when added to Lake Erie water. *Trans. Am. Fish. Soc.* 78: 96.
- Anderson, R.L., et al. 1980. Survival and growth of Tanytarsus dissimilis (Chironomidae) exposed to copper, cadmium, zinc, and lead. *Arch. Environ. Contam. Toxicol.* 9: 329.
- Anderson, R.V. 1977. Concentration of cadmium, copper, lead and zinc in six species of freshwater clams. *Bull. Environ. Contam. Toxicol.* 18: 492.
- Anderson, R.V. 1978. The effects of lead on oxygen uptake in the crayfish, Orconectes virilis (Hagen). *Bull. Environ. Contam. Toxicol.* 20: 394.

- Apostol, S. 1973. A bioassay of toxicity using protozoa in the study of aquatic environment pollution and its prevention. Environ. Res. 6: 365.
- Archison, G.J., et al. 1977. Trace metal contamination of bluegill (Lepomis macrochirus) from two Indiana lakes. Trans. Am. Fish. Soc. 106: 637.
- Aubert, M., et al. 1974. Utilisation d'une chaîne trophodynamique de type pélagique pour l'étude des transferts des pollutions métalliques. Rev. Int. Oceanogr. Med. 28: 27.
- Badsha, K.S. and C.R. Goldspink. 1982. Preliminary observations on the heavy metal content of four species of freshwater fish in NW England. Jour. Fish Biol. 21: 251.
- Baker, M.D., et al. 1983. Toxicity of pH, heavy metals and bisulfite to a freshwater green alga. Chemosphere 12: 35.
- Behan, M.J., et al. 1979. Lead accumulation in aquatic plants from metallic sources including shot. Jour. Wildl. Manage. 43: 240.
- Belding, D.L. 1927. Toxicity experiments with fish in reference to trade waste pollution. Trans. Am. Fish. Soc. 57: 100.
- Benijts-Claus, C. and F. Benijts. 1975. The effect of low lead and zinc concentrations on the larval development of the mud crab, Rhithropanopeus

- harrisii. In: J.H. Koeman and J.J. Strik (eds.), Sublethal Effects of Toxic Chemicals on Aquatic Animals. Elsevier, Amsterdam. p. 43.
- Berry, W.J. 1981. Memorandum to John H. Gencile. U.S. EPA, Narragansett, Rhode Island.
- Biegert, E.K. and V. Valkovic. 1980. Acute toxicity and accumulation of heavy metals in aquatic animals. Period. Biol. 82: 25.
- Biesinger, K.E. and G.M. Christensen. 1972. Effect of various metals on survival, growth, reproduction, and metabolism of Daphnia magna. Jour. Fish. Res. Board Can. 29: 1691.
- Birge, W.J. 1978. Aquatic toxicology of trace elements of coal and fly ash. In: J.H. Thorpe and J.W. Gibbons (eds.), Energy and Environmental Stress in Aquatic Systems. CONF-771114. National Technical Information Service, Springfield, Virginia. p. 219.
- Birge, W.J., et al. 1978. Embryo-larval bioassays on inorganic coal elements and in situ biomonitoring of coal-waste effluents. In: D.E. Samuel, et al. (eds.), Surface Mining and Fish/Wildlife Needs in the Eastern United States. PB 298353. National Technical Information Service, Springfield, Virginia. p. 97.
- Birge, W.J.; et al. 1980. Aquatic toxicity tests on inorganic elements occurring in oil shale. In: C. Gale (ed.), Oil Shale Symposium: Sampling,

- Analysis and Quality Assurance. EPA-600/9-80-022. National Technical Information Service, Springfield, Virginia. p. 519.
- Birge, W.J., et al. 1981. The reproductive toxicology of aquatic contaminants. In: J. Saxena and F. Fisher (eds.), Hazard Assessment of Chemicals: Current Developments. Vol. I. Academic Press, New York. p. 59.
- Boggess, W.R. (ed.). 1977. Lead in the environment. PB 278278. National Technical Information Service, Springfield, Virginia.
- Borgmann, U., et al. 1978. Rates of mortality, growth, and biomass production of Lymnaea palustris during chronic exposure to lead. Jour. Fish. Res. Board Can. 35: 1109.
- Borgmann, U. 1980. Interactive effects of metals in mixtures on biomass production kinetics of freshwater copepods. Can. Jour. Fish. Aquat. Sci. 37: 1295.
- Boutet, C. and C. Chaisemartin. 1973. Specific toxic properties of metallic salts in Austropotamobius pallipes pallipes and Orconectes limosus. C.R. Soc. Biol. 167: 1933.
- Brezina, E.R. and M.Z. Arnold. 1977. Levels of heavy metals in fishes from selected Penna. waters. Publication No. 50. Bureau of Water Quality Management, Pennsylvania Department of Environmental Resources, Harrisburg, Pennsylvania.

- Brezina, E.R., et al. 1974. Schuylkill river basin water quality. Publication No. 34. Bureau of Water Quality Management, Pennsylvania Department of Environmental Resources, Harrisburg, Pennsylvania.
- Bringmann, G. 1975. Determination of the biologically harmful effect of water pollutants by means of the retardation of cell proliferation of the blue algae Microcystis. Gesundheits-Ing. 96: 238.
- Bringmann, G. 1978. Determination of the biological toxicity of waterbound substances towards protozoa. I. bacteriovorous flagellates (model organism: Encosiphon sulcatum Stein). Z. Wasser Abwasser Forsch. 11: 210.
- Bringmann, G. and R. Kuhn. 1959a. The toxic effects of waste water on aquatic bacteria, algae, and small crustaceans. Gesundheits-Ing. 80: 115.
- Bringmann, G. and R. Kuhn. 1959b. Water toxicology studies with protozoans as test organisms. Gesundheits-Ing. 80: 239.
- Bringmann, G. and R. Kuhn. 1976. Comparative results of the damaging effects of water pollutants against bacteria (Pseudomonas putida) and blue algae (Microcystis aeruginosa). Gas-Wasserfach, Wasser-Abwasser 117: 410.
- Bringmann, G. and R. Kuhn. 1977a. Limiting values for the damaging action of water pollutants to bacteria (Pseudomonas putida) and green algae (Scenedesmus quadricauda) in the cell multiplication inhibition test. Z. Wasser Abwasser Forsch. 10: 87.

Bringmann, G. and R. Kuhn. 1977b. Results of the damaging effect of water pollutants on Daphnia magna. Z. Wasser Abwasser Forsch. 10: 161.

Bringmann, G. and R. Kuhn. 1978a. Limiting values for the noxious effects of water pollutant material to blue algae (Microcystis aeruginosa) and green algae (Scenedesmus quadricauda) in cell propagation inhibition tests. Vom Wasser 50: 45.

Bringmann, G. and R. Kuhn. 1978b. Testing of substances for their toxicity threshold: model organisms Microcystis (Diplocystis) aeruginosa and Scenedesmus quadricauda. Mitt. Int. Ver. Theor. Angew. Limnol. 21: 275.

Bringmann, G. and R. Kuhn. 1979. Comparison of toxic limiting concentrations of water contaminations toward bacteria, algae, and protozoa in the cell-growth inhibition test. Hausrech. Bauphys. Umweltrech. 100: 249.

Bringmann, G. and R. Kuhn. 1980a. Determination of the harmful biological effect of water pollutants on protozoa. II. bacteriovorous ciliates. Z. Wasser Abwasser Forsch. 13: 26.

Bringmann, G. and R. Kuhn. 1980b. Comparison of the toxicity threshold of water pollutants to bacteria, algae, and protozoa in the cell multiplication inhibition test. Water Res. 14: 231.

- Bringmann, G. and R. Kuhn. 1981. Comparison of the effects of harmful substances on flagellates as well as ciliates and on halozoic bacteriophagous and saprozoic protozoa. *Gas-Wasserfach, Wasser-Abwasser* 122: 308.
- Bringmann, G., et al. 1980. Determination of biological damage from water pollutants to protozoa. III. saprozoic flagellates. *Z. Wasser Abwasser Forsch.* 13: 170.
- Brown, B. and M. Ahsanullah. 1971. Effect of heavy metals on mortality and growth. *Mar. Pollut. Bull.* 2: 182.
- Brown, G.W. 1976. Effects of polluting substances on enzymes of aquatic organisms. *Jour. Fish. Res. Board Can.* 33: 2018.
- Brown, J.R. and L.Y. Chow. 1977. Heavy metal concentration in Ontario fish. *Bull. Environ. Contam. Toxicol.* 17: 190.
- Brown, V.M. 1968. Calculation of the acute toxicity of mixtures of poisons to rainbow trout. *Water Res.* 2: 723.
- Bryan, G.W. 1976. Heavy metal contamination in the sea. In: R. Johnson (ed.), *Marine Pollution*. Academic Press, New York. p. 185.
- Buikema, A.L., Jr., et al. 1974a. Rotifers as monitors of heavy metal pollution in water. *Bulletin 71*. Virginia Water Resources Research Center, Blacksburg, Virginia.

- Buikema, A.L., Jr., et al. 1974b. Evaluation of Philodina acuticornis (Rotifera) as a bioassay organism for heavy metals. Water Resources Bull. 10: 648.
- Buikema, A.L., Jr., et al. 1977. Rotifer sensitivity to combinations of inorganic water pollutants. Bulletin 92. Virginia Water Resources Research Center, Blacksburg, Virginia.
- Cairns, J., et al. 1976. Invertebrate response to thermal shock following exposure to acutely sub-lethal concentrations of chemicals. Arch. Hydrobiol. 77: 164.
- Calabrese, A. and D.A. Nelson. 1974. Inhibition of embryonic development of the hard clam, Mercenaria mercenaria, by heavy metals. Bull. Environ. Contam. Toxicol. 11: 92.
- Calabrese, A., et al. 1973. The toxicity of heavy metals to embryos of the American oyster Crassostrea virginica. Mar. Biol. 18: 162.
- Call, D.J., et al. 1981. Aquatic pollutant hazard assessments and development of a hazard prediction technology by quantitative structure-activity relationships. First Quarterly Report to EPA. Center for Lake Superior Environmental Studies. University of Wisconsin-Superior, Superior, Wisconsin.

- Call, D.J., et al. 1983. Toxicity and metabolism studies with EPA priority pollutants and related chemicals in freshwater organisms. PB83-263665. National Technical Information Service, Springfield, Virginia.
- Callahan, M.A., et al. 1979. Water-related environmental fate of 129 priority pollutants. Vol. I. EPA-440/4-79-029a. National Technical Information Service, Springfield, Virginia.
- Canterford, G.S. and D.R. Canterford. 1980. Toxicity of heavy metals to the marine diatom Ditylum brightwellii (West) Grunow: correlation between toxicity and metal speciation. Jour. Mar. Biol. Assoc. U.K. 60: 227.
- Canterford, G.S., et al. 1978. Accumulation of heavy metals by the marine diatom Ditylum brightwellii (West) Grunow. Aust. Jour. Mar. Freshwater Res. 29: 613.
- Cardin, J.A. 1981. Memorandum to John H. Gencile. U.S. EPA, Narragansett, Rhode Island.
- Carpenter, K.E. 1925. On the biological factors involved in the destruction of river-fisheries by pollution due to lead-mining. Ann. Appl. Biol. 12: 1.
- Carpenter, K.E. 1926. The lead mine as an active agent in river pollution. Ann. Appl. Biol. 13: 395.

- Carpenter, K.E. 1927. The lethal action of soluble metallic salts of fishes. Br. Jour. Exp. Biol. 4: 378.
- Carpenter, K.E. 1930. Further researches on the action of metallic salts on fishes. Jour. Exp. Zool. 56: 407.
- Carter, J.W. and I.L. Cameron. 1973. Toxicity bioassay of heavy metals in water using Tetrahymena pyriformis. Water Res. 7: 951.
- Chapman, G.A., et al. Manuscript. Effects of water hardness on the toxicity of metals to Daphnia magna. U.S. EPA, Corvallis, Oregon.
- Chapman, W.H., et al. 1968. Concentration factors of chemical elements in edible aquatic organisms. UCRL-50564. National Technical Information Service, Springfield, Virginia.
- Christensen, E.R., et al. 1979. Effects of manganese, copper and lead on Selenastrum capricornutum and Chlorella stigmatophora. Water Res. 13: 79.
- Christensen, G.M. 1975. Biochemical effects of methylmercuric chloride, cadmium chloride and lead nitrate on embryos and alevins of the brook trout. Toxicol. Appl. Pharmacol. 32: 191.
- Christensen, G., et al. 1977. The effect of methylmercuric chloride, cadmium chloride, and lead nitrate on six biochemical factors of the brook trout (Salvelinus fontinalis). Toxicol. Appl. Pharmacol. 42: 523.

- Clarke, A.M. and J.H. Clarke. 1974. A static monitor for lead in natural and waste waters. *Environ. Letters* 7: 251.
- Crandall, C.A. and C.J. Goodnight. 1962. Effects of sublethal concentrations of several toxicants on growth of the common guppy Lebistes reticulatus. *Limnol. Oceanogr.* 7: 233.
- Curtis, M.W. and C.H. Ward. 1981. Aquatic toxicity of forty industrial chemicals: testing in support of hazardous substance spill prevention regulation. *Jour. Hydrol.* 51: 359.
- Davies, P.H. and W.E. Everhart. 1973. Effects of chemical variations in aquatic environments: lead toxicity to rainbow trout and testing application factor concept. EPA-R3-73-011C. National Technical Information Service, Springfield, Virginia.
- Davies, P.H., et al. 1976. Acute and chronic toxicity of lead to rainbow trout (Salmo gairdneri) in hard and soft water. *Water Res.* 10: 199.
- Davis, G.A. 1978. Pollution studies with marine plankton. Part II. heavy metals. *Adv. Mar. Biol.* 15: 381.
- Dawson, A.B. 1935. The hemopoietic response in the catfish, Ameiurus nebulosus, to chronic lead poisoning. *Biol. Bull.* 68: 335.

Demayo, A., et al. 1980. Guidelines for surface water quality: inorganic chemical substances - lead. Inland Waters Directorate, Water Quality Branch, Ottawa, Canada.

Demayo, A., et al. 1982. Toxic effects of lead and lead compounds on human health, aquatic life, wildlife, plants and livestock. CRC Crit. Rev. Environ. Control 12: 257.

Devi Prasad, P.V. and P.S. Devi Prasad. 1982. Effect of cadmium, lead and nickel on three freshwater green algae. Water Air Soil Pollut. 17: 263.

Dilling, W.J. and C.W. Healy. 1927. Influence of lead on the metallic ions of Cu, Zn, thorium, beryllium and thallium on the germination of frogs' spawn and the growth of tadpoles. Ann. Appl. Biol. 13: 177.

Dilling, W.J., et al. 1926. Experiments on the effects of lead on the growth of plaice (Pleuronectes platessa). Ann. Appl. Biol. 13: 163.

Dixon, W.J. and M.B. Brown, eds. 1979. BMDP Biomedical Computer Programs, P-series. University of California, Berkeley, California. p. 521.

Dorfman, D. 1977. Tolerance of Fundulus heteroclitus to different metals in saltwaters. Bull. New Jersey Acad. Sci. 22: 21.

- Dorfman, D. and W.R. Whitworth. 1969. Effects of fluctuations of lead, temperature, and dissolved oxygen on the growth of brook trout. Jour. Fish. Res. Board Can. 26: 2493.
- Eide, I. and S. Myklesad. 1980. Long-term uptake and release of heavy metals by Ascophyllum nodosum (L.) Le Jol (Phaeophyceae) in situ. Environ. Pollut. (Series A) 23: 19.
- Eisler, R. 1977. Acute toxicities of selected heavy metals to the soft-shell clam, Mya arenaria. Bull. Environ. Contam. Toxicol. 17: 137.
- Eisler, R. 1981. Trace Metal Concentrations in Marine Organisms. Pergamon Press, New York.
- Eisler, R., et al. 1979. Fourth annotated bibliography on biological effects of metals in aquatic environments. EPA-600/3-79-084. National Technical Information Service, Springfield, Virginia.
- Ellgaard, E.G. and T.W. Rudner. 1982. Lead acetate: toxicity without effects on the locomotor activity of the bluegill sunfish. Jour. Fish Biol. 21: 411.
- Ellis, M.M. 1937. Detection and measurement of stream pollution. (Bulletin No. 22, U.S. Bureau of Fisheries) Bull. Bureau Fish. 48: 365.

- Ellis, M.M. 1940. Pollution of the Coeur d'Alene River and adjacent waters by mine wastes. Special Scientific Report No. 1. U.S. Fish and Wildlife Service. p. 61.
- English, J.N., et al. 1963. Pollutational effects of outboard motor exhaust - laboratory studies. Jour. Water Pollut. Control Fed. 35: 923.
- Enk, M.D. and B.J. Machis. 1977. Distribution of cadmium and lead in a stream ecosystem. Hydrobiologia 52: 153.
- Evans, R.D. and D.C. Lasenby. 1983. Relationship between body-lead concentration of Mysis relicta and sediment-lead concentration in Kootenary, B.C. Can. Jour. Fish. Aquat. Sci. 40: 78.
- Ferard, J.F., et al. 1982. Value of dynamic tests in acute ecotoxicity assessment in algae. In: W.C. McKay (ed.), Proceedings of the Ninth Annual Aquatic Toxicity Workshop. Can. Tech. Rept. Fish. Aquat. Sci. No. 1163. University of Alberta, Edmonton, Alberta. p. 38.
- Ferguson, J. and B. Bubela. 1974. The concentration of Cu(II), Pb(II), and Zn(II) from aqueous solution by particulate algal matter. Chem. Geol. 13: 163.
- Fisher, N.S. and G.J. Jones. 1981. Heavy metals and marine phytoplankton: correlation of toxicity and sulfhydryl-binding. Jour. Phycol. 17: 108.

- Foster, P.L. 1982a. Species association and metal contents of algae from rivers polluted by heavy metals. *Freshwater Biol.* 12: 17.
- Foster, P.L. 1982b. Metal resistances of chlorophyta from rivers polluted by heavy metals. *Freshwater Biol.* 12: 41.
- Freeman, B.J. 1978. Accumulation of cadmium, chromium, and lead by bluegill sunfish (*Lepomis macrochirus* Rafinesque) under temperature and oxygen stress. SRO-757-6. National Technical Information Service, Springfield, Virginia.
- Freeman, B.J. 1980. Accumulation of cadmium, chromium, and lead by bluegill sunfish (*Lepomis macrochirus* Rafinesque) under temperature and oxygen stress. Thesis. University of Georgia, Athens, Georgia.
- Fujiya, M. 1961. Use of electrophoretic serum separation in fish studies. *Jour. Water Pollut. Control Fed.* 33: 250.
- Gale, N.L., et al. 1973a. Transport of trace pollutants in lead mining wastewaters. In: D.D. Hemphill (ed.), *Trace Substances in Environmental Health-VI*. University of Missouri, Columbia, Missouri. p. 95.
- Gale, N.L., et al. 1973b. Aquatic organisms and heavy metals in Missouri's new lead belt. *Water Resources Bull.* 9: 673.

Gale, N.L., et al. 1982. Lead concentrations in edible fish filers collected from Missouri's old lead belt. In: D.D. Hemphill (ed.), Trace Substances in Environmental Health-XVI. University of Missouri, Columbia, Missouri. p. 12.

Garavini, C. and P. Martelli. 1979. Effect of lead acetate on erythropoiesis and ultrastructural changes of erythroblasts in the catfish. *Monitore Zool. Italy* 13: 83.

~~Gentile, J.H., et al. 1982. The use of life-tables for evaluating the chronic toxicity of pollutants to Mysidopsis bahia. *Hydrobiologia* 93: 179.~~

Gentile, S.M. 1982. Memorandum to John H. Gentile. U.S. EPA, Narragansett, Rhode Island.

Goetts, J.P., et al. 1972. Laboratory water pollution studies. *Colorado Fisheries Research Review*.

Gordon, M., et al. 1980. Mytilus californianus as a bioindicator of trace metal pollution: variability and statistical considerations. *Mar. Pollut. Bull.* 11: 193.

Gould, E. and R.A. Greig. 1983. Short-term low salinity response in lead-exposed lobsters, Homarus americanus (Milne Edwards). *Jour. Exp. Mar. Biol. Ecol.* 69: 283.

- Grande, M. and S. Andersen. 1983. Lethal effects of hexavalent chromium, lead and nickel on young stages of Atlantic salmon (Salmo salar L.) in soft water. *Vatten* 39: 405.
- Gray, J.S. and R.J. Ventilla. 1973. Growth rates of sediment-living marine protozoan as a toxicity indicator for heavy metals. *Ambio* 2: 118.
- Haider, G. 1964. Studies on the heavy metal poisoning of fishes. I: lead poisoning of rainbow trout. *Z. Angew. Zool.* 51: 347.
- Hale, J.G. 1977. Toxicity of metal mining wastes. *Bull. Environ. Contam. Toxicol.* 17: 66.
- Hannan, P.J. and C. Patouillet. 1972. Effect of mercury on algal growth rates. *Biotechnol. Bioeng.* 14: 93.
- Hedcke, S.F. and F.A. Puglisi. 1980. Effects of waste oil on the survival and reproduction of the American flagfish. *Can. Jour. Fish. Aquat. Sci.* 37: 757.
- Heisey, R.M. and A.H. Damman. 1982. Copper and lead uptake by aquatic macrophytes in eastern Connecticut, U.S.A. *Aquat. Bot.* 14: 213.
- Hessler, A. 1974. Effects of lead on algae. I. effects of Pb on viability and motility of Platymonas subcordiformis (Chlorophyta:volvocales). *Water Air Soil Pollut.* 3: 371.

- Hessler, A. 1975. Effects of lead on algae. Mutagenesis experiments on Platymonas subcordiformis (Chlorophyta:valvocales). *Mutat. Res.* 31: 43.
- Hodson, P.V. 1976. Delta-amino levulinic acid dehydratase activity of fish blood as an indicator of a harmful exposure to lead. *Jour. Fish. Res. Board Can.* 33: 268.
- Hodson, P.V., et al. 1977. Evaluation of erythrocyte delta-amino levulinic acid dehydratase activity as a short-term indicator in fish of a harmful exposure to lead. *Jour. Fish. Res. Board Can.* 34: 501.
- Hodson, P.V., et al. 1978a. Chronic toxicity of water-borne and dietary lead to rainbow trout (Salmo gairdneri) in Lake Ontario water. *Water Res.* 12: 869.
- Hodson, P.V., et al. 1978b. pH-induced changes in blood lead of lead-exposed rainbow trout. *Jour. Fish. Res. Board Can.* 35: 437.
- Hodson, P.V., et al. 1979a. Effects of increasing dietary ascorbic acid on chronic lead toxicity in rainbow trout. *Am. Jour. Clin. Nutrition* 32: R28.
- Hodson, P.V., et al. 1979b. Effect of fish age on predicted and observed chronic toxicity of lead to rainbow trout in Lake Ontario water. *Int. Assoc. Great Lakes Res.* 1: 84.

- Hodson, P.V., et al. 1980. Effects of dietary ascorbic acid on chronic lead toxicity to young rainbow trout (Salmo gairdneri). Can. Jour. Fish. Aquat. Sci. 37: 170.
- Hodson, P.V., et al. 1982. Effect of growth and size of fish on rate of intoxication of waterborne lead. Can. Jour. Fish. Aquat. Sci. 39: 1243.
- Hodson, P.V., et al. 1983a. Suitability of a biochemical method for assessing the exposure of feral fish to lead. In: W.E. Bishop, et al. (eds.), Aquatic Toxicology and Hazard Assessment. ASTM STP 802. American Society for Testing and Materials, Philadelphia, Pennsylvania. p. 389.
- Hodson, P.V., et al. 1983b. Effect of fluctuating lead exposures in lead accumulation by rainbow trout. Environ. Toxicol. Chem. 2: 225.
- Holcombe, G.W., et al. 1976. Long term effects of lead exposure on three generations of brook trout (Salvelinus fontinalis). Jour. Fish. Res. Board Can. 33: 1731.
- Hollibaugh, J.T., et al. 1980. A comparison of the acute toxicities of ten heavy metals to phytoplankton from Saanich Inlet, B. C. Canada. Estuarine Coastal Mar. Sci. 10: 93.
- Holm, J. 1980. Lead, Cd, As and Zn contents in fish from uncontaminated and contaminated inland waters. Sonderdruck aus Fleischwirtschaft 5: 1076.

- Jackim, E. 1973. Influence of lead and other metals on fish delta-aminolevulinate dehydrase activity. Jour. Fish. Res. Board Can. 30: 560.
- Jackim, E., et al. 1970. Effects of metal poisoning on five liver enzymes in the killifish (Fundulus heteroclitus). Jour. Fish. Res. Board Can. 27: 383.
- Jana, S. and M.A. Choudhuri. 1982. Senescence in submerged aquatic angiosperms: effects of heavy metals. New Phytol. 90: 477.
- Jana, S. and M.A. Choudhuri. 1984. Synergistic effects of heavy metal pollutants on senescence of submerged aquatic plants. Water Air Soil Pollut. 21: 351.
- Jennett, J.C., et al. 1981. Some effects of century old abandoned lead mining operations on streams in Missouri, U.S.A. Minerals and the Environment 3: 17.
- Johnson, M.S. and J.W. Eacon. 1980. Environmental contamination through residual trace metal dispersal from a derelict lead-zinc mine. Jour. Environ. Qual. 9: 175.
- Johnson, W.W. and M.T. Finley. 1980. Handbook of acute toxicity of chemicals to fish and aquatic invertebrates. Resource Publication 137. U.S. Fish and Wildlife Service, Washington, D.C.
- Jones, J.R.E. 1935. The toxic action of heavy metal salts on the three-spined stickleback. Jour. Exp. Biol. 12: 165.

- Jones, J.R. 1938. The relative toxicity of salts of Pb, Zn, and Cu to the stickleback and the effects of calcium on the toxicity of lead and zinc salts. Jour. Exp. Biol. 15: 394.
- Jones, J.R.E. 1939. The relation between the electrolytic solution pressures of the metals and their toxicity to the stickleback (Gasterosteus aculeatus L.). Jour. Exp. Biol. 16: 425.
- Jones, J.R.E. 1947a. The oxygen consumption of Gasterosteus aculeatus L. in toxic solutions. Jour. Exp. Biol. 23: 298.
- Jones, J.R. 1947b. A further study of the reactions of fish to toxic solutions. Jour. Exp. Biol. 24: 22.
- Kaplan, H.M., et al. 1967. Toxicity of lead nitrate solutions for frogs (Rana pipiens). Lab. Animal Care 17: 240.
- Kapur, K. and N.A. Yadav. 1982. The effect of certain heavy metal salts on the development of eggs of common carp. Acta Hydrochim. Hydrobiol. 10: 517.
- Kariya, T., et al. 1969. Studies of the post-mortem identification of the pollutant in fish killed by water pollution-X: acute poisoning with lead. Bull. Jap. Soc. Sci. Fish. 35: 1167.
- Kharhar, D.P., et al. 1976. Uranium and thorium decay series nuclides in plankton from the Caribbean. Limnol. Oceanogr. 21: 294.

- Knowlton, M.K., et al. 1983. Uptake of lead from aquatic sediment by submerged macrophytes and crayfish. Arch. Environ. Contam. Toxicol. 12: 535.
- Kopfler, F.C. and J. Mayer. 1973. Concentration of five trace metals in the waters and oysters (Crassostrea virginica) of Mobile Bay, Alabama. Proc. Natl. Shellfish Assoc. 63: 27.
- Laube, V.M., et al. 1980. Strategies of response to copper, cadmium and lead by a blue green and a green alga. Can. Jour. Microbiol. 26: 1300.
- Leland, H.V. and J.M. McNurney. 1974. Lead transport in a river ecosystem. In: Proceedings of the International Conference on Transport of Persistent Chemicals in Aquatic Ecosystems. Part III. National Research Council of Canada, Ottawa. p. 17.
- Lloyd, R. 1961. Effects of dissolved oxygen concentrations on the toxicity of several poisons to rainbow trout (Salmo gairdneri). Jour. Exp. Biol. 38: 447.
- Lu, P., et al. 1975. Model ecosystems studies of lead and cadmium of urban sewage sludge containing these elements. Jour. Environ. Qual. 4: 505.
- Lucas, H.F. and D.N. Edgington. 1970. Concentration of trace elements in Great Lakes fishes. Jour. Fish. Res. Board Can. 27: 677.

- Lussier, S.M., et al. Manuscript. Acute and chronic effects of heavy metals and cyanide on Mysidopsis bahia (Crustacea: Mysidacea). U.S. EPA, Narragansett, Rhode Island.
- Malanchuk, J.L. and G.K. Gruendling. 1973. Toxicity of lead nitrate to algae. *Water Air Soil Pollut.* 2: 181.
- Manalis, R. and G. Cooper. 1973. Presynaptic and postsynaptic effects of lead at the frog neuromuscular junction. *Nature* 243: 354.
- Manalis, R.S., et al. 1984. Effects of lead on neuromuscular transmission in the frog. *Brain Res.* 294: 95.
- Marchetti, R. 1978. Acute toxicity of alkyl lead to some marine organisms. *Mar. Pollut. Bull.* 9: 206.
- Marion, M. and F. Denizeau. 1983. Rainbow trout and human cells in culture for the evaluation of the toxicity of aquatic pollutants: a study with lead. *Aquar. Toxicol.* 3: 47.
- Martin, M.G. and J.M. Mudre. 1982. Patterns of bioaccumulation of heavy metals in stream fishes. *Virginia Jour. Sci.* 33: 116.
- Martin, M., et al. 1981. Toxicities of ten metals to Crassostrea gigas and Mytilus edulis embryos and Cancer magister larvae. *Mar. Pollut. Bull.* 12: 305.

- Martin, M., et al. 1984. Relationships between physiological stress and trace toxic substances in the bay mussel, Mytilus edulis, from San Francisco Bay, California. Mar. Environ. Res. 11: 91.
- Mathis, B.J. and T.F. Cummings. 1973. Selected metals in sediments, water, and biota in the Illinois River. Jour. Water Polluc. Control Fed. 45: 1573.
- Mathis, P.F. and N.R. Kevern. 1975. Distribution of mercury, cadmium, lead and thallium in a trophic lake. Hydrobiologia 46: 207.
- May, T.W. and G.L. McKinney. 1981. Cadmium, lead, mercury, arsenic and selenium concentrations in freshwater fish, 1976-77 - National Pesticide Monitoring Program. Pestic. Monic. Jour. 15: 14.
- Mehrle, P.M., et al. 1982. Relationship between body contaminants and bone development in East-Coast striped bass. Trans. Am. Fish. Soc. 111: 231.
- Merlini, M. and G. Pozzi. 1977a. Lead and freshwater fishes: Part I - lead accumulation and water pH. Environ. Polluc. 12: 167.
- Merlini, M. and G. Pozzi. 1977b. Lead and freshwater fishes: Part II - ionic lead accumulation. Environ. Polluc. 13: 119.
- Metayer, C., et al. 1982. Accumulation of some trace metals (cadmium, lead, copper and zinc) in sole (Solea solea) and flounder (Platichthys flesus):

- changes as a function of age and organotropism. *Rev. Int. Oceanogr. Med.* 66-67: 33.
- Monahan, T.J. 1976. Lead inhibition of chlorophycean microalgae. *Jour. Phycol.* 12: 358.
- Montgomery, J.R., et al. 1978. Biological availability of pollutants to marine organisms. EPA-600/3-78-035. National Technical Information Service, Springfield, Virginia.
- Morgan, W.S.C. 1979. Fish locomotor behavior patterns as a monitoring tool. *Jour. Water Pollut. Control Fed.* 51: 580.
- Mount, D.I. and T.J. Norberg. 1984. A seven-day life-cycle cladoceran toxicity test. *Environ. Toxicol. Chem.* 3: 425.
- Narbonne, J.F., et al. 1973. Toxicity of lead nitrate to carp - data on modifications of nucleoprotein and glucide metabolism. *Compt. Rend. Soc. Biol.* 167: 572.
- Nash, W.W., et al. 1981. The uptake and cellular distribution of lead in developing sea urchin embryos. *Comp. Biochem. Physiol.* 69C: 205.

- Nehring, R.B. 1976. Aquatic insects as biological monitors of heavy metal pollution. *Bull. Environ. Contam. Toxicol.* 15: 147.
- Nehring, R.B., et al. 1979. Reliability of aquatic insects versus water samples as measures of aquatic lead pollution. *Bull. Environ. Contam. Toxicol.* 22: 103.
- Neter, J. and W. Wasserman. 1974. *Applied Linear Statistical Models*. Irwin, Inc., Homewood, Illinois.
- Newman, M.C. and A.W. McIntosh. 1983a. Slow accumulation of lead from contaminated food sources by the freshwater gastropods, Physa integra and Campeloma decisum. *Arch. Environ. Contam. Toxicol.* 12: 685.
- Newman, M.C. and A.W. McIntosh. 1983b. Lead elimination and size effects on accumulation by two freshwater gastropods. *Arch. Environ. Contam. Toxicol.* 12: 25.
- North, W.F., et al. 1972. Marine algae and their relations to pollution problems. In: M. Ruivo (ed.), *Marine Pollution and Sea Life*. Fishing Trading News Ltd., London. p. 330.
- Nyman, H.G. 1981. Sublethal effects of lead on size selective predation by fish-application in the ecosystem level. *Verh. Int. Ver. Limnol.* 21: 1126.

- O'Neill, J.G. 1981. Effects of intraperitoneal lead and cadmium on the humoral immune response of Salmo trutta. Bull. Environ. Contam. Toxicol. 27: 42.
- Overnell, J. 1975. The effect of some heavy metal ions on photosynthesis in freshwater algae. Pestic. Biochem. Physiol. 5: 19.
- Ozoh, P.T. 1979. Studies on intraperitoneal toxicity of lead to Cichlasoma nigrofasciatum development. Bull. Environ. Contam. Toxicol. 21: 676.
- Face, F., et al. 1977. Effects of sublethal doses of copper sulphate and lead nitrate on growth and pigment composition of Dunaliella salina Teod. Bull. Environ. Contam. Toxicol. 17: 679.
- Pagenkopf, G.K. and D.R. Newman. 1974. Lead concentrations in native trout. Bull. Environ. Contam. Toxicol. 12: 70.
- Pakkala, I.S., et al. 1972. Residues in fish, wildlife and estuaries. Pestic. Monit. Jour. 5: 348.
- Parker, J.G. 1984. The effects of selected chemicals and water quality on the marine polychaete, Ophryotrocha diadema. Water Res. 18: 865.
- Passino, D.R. and C.A. Cotanc. 1979. Allantoinase in lake trout: in vitro effects of PCB, DDT and metals. Comp. Biochem. Physiol. 62C: 71.

- Pawlaczyk-Szpilowa, M. and J. Slowik. 1981. The biocumulation of copper and lead by Scenedesmus obliquus. *Acta Microbiol. Pol.* 30: 79.
- Pennington, C.H., et al. 1982. Contaminant levels in fishes from Brown's Lake, Mississippi. *Jour. Mississippi Acad. Sci.* 27: 139.
- Phillips, G.R. 1980. Accumulation of selected elements (As, Cu, Hg, Pb, Se and Zn) by northern pike reared in surface coal mine decant water. *Proc. Montana Acad. Sci.* 39: 44.
- Phillips, G.R. and R.C. Russo. 1978. Metal bioaccumulation in fishes and aquatic invertebrates: a literature review. EPA-600/3-78-103. National Technical Information Service, Springfield, Virginia.
- Pickering, Q.H. and C. Henderson. 1966. The acute toxicity of some heavy metals to different species of warmwater fishes. *Air Water Pollut. Int. Jour.* 10: 453.
- Popham, J.D. and J.M. D'Auria. 1981. Statistical models for estimating seawater metal concentrations from metal concentrations in mussels (Mytilus edulis). *Bull. Environ. Contam. Toxicol.* 27: 660.
- Price, R.E. and L.A. Knight. 1978. Hg, Cd, Pb and As in sediment, plankton and clams from Lake Washington and Sardis Reservoir, Mississippi. *Pestic. Monit. Jour.* 4: 182.

- Pringle, B.H., et al. 1968. Trace metal accumulation by estuarine mollusks. Am. Soc. Civil. Eng., Jour. Sanic. Eng. Div. 94: 455.
- Qureshi, S.A., et al. 1980. Acute toxicity of four heavy metals to benthic fish food organisms from River Khan, Vjjain. Int. Jour. Environ. Studies 15: 59.
- Rachlin, J.W., et al. 1982. The growth response of the green alga (Chlorella saccharophila) to selected concentrations of heavy metals Cd, Cu, Pb and Zn. In: D.D. Hemphill (ed.), Trace Substances in Environmental Health-XVI. University of Missouri, Columbia, Missouri. p. 145.
- Rachlin, J.W., et al. 1983. The growth response of the diatom Navicula incerta to selected concentrations of the metals cadmium, copper, lead and zinc. Bull. Torrey Bot. Club 110: 217.
- Randall, G.W., et al. 1981. The significance of heavy metals in urban runoff entering the Occoquan Reservoir. Bulletin 132. Virginia Water Resources Research Center, Blacksburg, Virginia.
- Rao, D.S. and A.B. Saxena. 1980. Acute toxicity of Hg, Zn, Pb, Cd and Mn to Chironomus sp. Int. Jour. Environ. Studies 16: 227.
- Rao, V.N.R. and S.K. Subramanian. 1982. Metal toxicity tests on growth of some diatoms. Acta Bot. Indica 10: 274.

- Rachore, H.S. and H. Swarup. 1978. A short note on the pollution ecology of Chironomus tentans larvae in a river. Natl. Acad. Sci. Letters 1: 235.
- Rachore, H.S., et al. 1979. Toxicity of cadmium chloride and lead nitrate to Chironomus tentans larvae. Environ. Pollut. 18: 173.
- Ray, S. 1978. Bioaccumulation of lead in Atlantic salmon. Bull. Environ. Contam. Toxicol. 19: 631.
- Ray, S., et al. 1981. Accumulation of copper, zinc, cadmium and lead from two contaminated sediments by three marine invertebrates--a laboratory study. Bull. Environ. Contam. Toxicol. 26: 315.
- Reish, D.J. and R.S. Carr. 1978. The effect of heavy metals on the survival, reproduction, development and life cycles for two species of polychaetous annelids. Mar. Pollut. Bull. 9: 24. (Table 3 available from author.)
- Reish, D.J., et al. 1976. The effect of heavy metals on laboratory populations of two polychaetes with comparisons to the water quality conditions and standards in Southern California marine waters. Water Res. 10: 299.
- Rice, H., et al. 1973. The effects of some trace metals on marine phytoplankton. CRC Crit. Rev. Microbiol. 3: 27.
- Rivkin, R.B. 1979. Effects of lead on growth of the marine diatom Skeletonema costatum. Mar. Biol. 50: 239.

- Rolfe, G.L., et al. 1977. Environmental contamination by lead and other heavy metals. Vol. 2: ecosystem analysis. Institute for Environmental Studies, University of Illinois at Urbana, Champaign, Illinois.
- Rushton, W. 1922. Biological notes. Salmon and Trout Magazine 28: 42.
- Ruchven, J.A. and J. Cairns, Jr. 1973. Response of fresh-water protozoan artificial communities to metals. Jour. Protozool. 20: 127.
- Ryck, F.M. and J.R. Whitley. 1974. Pollution abatement in the lead mining districts of Missouri. Missouri Department of Conservation, Columbia, Missouri.
- Sauter, S., et al. 1975. Effects of exposure to heavy metals on selected freshwater fish. Toxicity of copper, cadmium, chromium and lead to eggs and larvae of seven fish species. EPA 600/3-76-105. National Technical Information Service, Springfield, Virginia.
- Say, P.J. and B.A. Whitton. 1983. Accumulation of heavy metals by aquatic mosses. 1: Fontinalis antipyretica. Hydrobiologia 100: 245.
- Schulz-Baldes, M. 1972. Toxizität und anreicherung von blei bei der miesmuschel Mytilus edulis im laborexperiment. Mar. Biol. 16: 266.
- Schulz-Baldes, M. 1974. Lead uptake from seawater and food, and lead loss in the common mussel Mytilus edulis. Mar. Biol. 25: 177.

- Schulz-Baldes, M. and R.A. Lewin. 1976. Lead uptake in two marine phytoplankton organisms. Biol. Bull. 150: 118.
- Schulze, H. and J. Brand. 1978. Lead toxicity and phosphate deficiency in chlamydomonas. Plant Physiol. 62: 727.
- Scott, K.J., et al. Manuscript. Toxicological methods using the benthic amphipod Ampelisca abdita Mills. U.S. EPA, Narragansett, Rhode Island.
- Shaw, W.H. and B. Grushkin. 1957. The toxicity of metal ions to aquatic organisms. Arch. Biochem. Biophys. 67: 447.
- Shaw, W.H.R. and B.R. Lowrance. 1956. Bioassay for the estimation of metal ions. Anal. Chem. 28: 1164.
- Shuster, C.N. and B.H. Pringle. 1969. Trace metal accumulation by the American Eastern oyster, Crassostrea virginica. Proc. Natl. Shellfish Assoc. 59: 91.
- Sicko-Goad, L. 1982. A morphometric analysis of algal response to low dose short term heavy metal exposure. Protoplasma 110: 75.
- Sicko-Goad, L. and D. Lazinsky. 1981. Accumulation and cellular effects of heavy metals in benthic and planktonic algae. Micron Microscopia Acta 22: 289.
- Sicko-Goad, L. and D. Lazinsky. 1982. Polyphosphate body formation and degradation in Plectonema boryanum. Micron Microscopia Acta 13: 459.

- Sicko-Goad, L. and E.F. Stoermer. 1979. A morphometric study of lead and copper effects on Diatoma ceneae. Jour. Phycol. 15: 316.
- Sidwell, V.D., et al. 1978. Composition of the edible portion of raw (fresh or frozen) crustaceans, finfish and mollusks. III. microelements. Mar. Fish. Res. 40: 1.
- Simpson, R.D. 1979. Uptake and loss of zinc and lead by mussels (Mytilus edulis) and relationships with body weight and reproductive cycle. Mar. Pollut. Bull. 10: 74.
- Sippel, A.J.A., et al. 1983. Histopathological and physiological responses of rainbow trout to sublethal levels of lead. Water Res. 17: 1115.
- Smith, D.J., et al. 1981. Distribution and significance of copper, lead, zinc and cadmium in the Corio Bay ecosystem. Aust. Jour. Mar. Freshwater Res. 32: 151.
- Sparks, R.E., et al. 1983. Identification of the water quality factors which prevent fingernail clams from recolonizing the Illinois River. Research Report No. 179. Water Resources Center, University of Illinois, Urbana, Illinois.
- Spehar, R.L., et al. 1978. Toxicity and bioaccumulation of cadmium and lead in aquatic invertebrates. Environ. Pollut. 15: 195.

- Stanley, R.A. 1974. Toxicity of heavy metals and salts to Eurasian watermilfoil (Myriophyllum spicatum L.). Arch. Environ. Contam. Toxicol. 2: 331.
- Sceele, R.L. and G.B. Thursby. 1983. A toxicity test using life stages of Champia parvula (Rhodophyta). In: W.E. Bishop, et al. (eds.), Aquatic Toxicology and Hazard Assessment. ASTM STP 802. American Society for Testing and Materials, Philadelphia, Pennsylvania. p. 73.
- Stephan, C.E., et al. 1985. Guidelines for deriving numerical national water quality criteria for the protection of aquatic organisms and their uses. National Technical Information Service, Springfield, Virginia.
- Stewart, J. and M. Schulz-Baldes. 1976. Long-term lead accumulation in abalone (Haliotis sp.) fed on lead-treated brown algae (Egregia laevigata). Mar. Biol. 36: 19.
- Stone, C.L., et al. 1981. Bioavailability of lead in oysters fed to young Japanese quail. Environ. Res. 26: 409.
- Stratford, H.K., et al. 1984. Effects of heavy metals on water hyacinths. Aquat. Toxicol. 5: 117.
- Stromgren, T. 1980. The effect of lead, cadmium, and mercury on the increase in length of five intertidal fucales. Jour. Exp. Mar. Biol. Ecol. 43: 107.

- Sturesson, U. 1978. Cadmium enrichment in shells of Mytilus edulis. *Ambio* 7: 122.
- Talbot, V., et al. 1976. Lead in Port Phillip bay mussels. *Mar. Pollut. Bull.* 7: 234.
- Tarzwil, C.M. and C. Henderson. 1960. Toxicity of less common metals to fishes. *Ind. Wastes* 5: 12.
- Thomas, W. ., et al. 1980. Effects of heavy metals on the morphology of some marine phytoplankton. *Phycologia* 19: 202.
- Thompson, S.E., et al. 1972. Concentration factors of chemical elements in edible aquatic organisms. UCRL-50564. Rev. 1. National Technical Information Service, Springfield, Virginia.
- Tong, S.S.C., et al. 1974. Trace metals in Lake Cayuga lake trout (Salvelinus namaycush) in relation to age. *Jour. Fish. Res. Board Can.* 31: 238.
- Trollope, D.R. and B. Evans. 1976. Concentrations of copper, iron, lead, nickel and zinc in freshwater algae blooms. *Environ. Pollut.* 11: 109.
- Tsui, P.T.P. and P.J. McCart. 1981. Chlorinated hydrocarbon residues and heavy metals in several fish species from the Cold Lake area in Alberta, Canada. *Int Jour. Environ. Anal. Chem.* 10: 277.

- Tucker, R.K. and A. Macce. 1980. In vitro effects of cadmium and lead in ATPase in the gill of the rock crab, Cancer irroratus. Bull. Environ. Contam. Toxicol. 24: 847.
- Turnbull, H., et al. 1954. Toxicity of various refinery materials to freshwater fish. Ind. Eng. Chem. 46: 324.
- U.S. EPA. 1976. Quality criteria for water. EPA-440/9-76-023. National Technical Information Service, Springfield, Virginia.
- U.S. EPA. 1980. Ambient water quality criteria for lead. EPA-440/4-80-057. National Technical Information Service, Springfield, Virginia.
- U.S. EPA. 1983a. Methods for chemical analysis of water and wastes. EPA-600/4-79-020 (Revised March 1983). National Technical Information Service, Springfield, Virginia.
- U.S. EPA. 1983b. Water quality standards regulation. Federal Register 48: 51400. November 8.
- U.S. EPA. 1983c. Water quality standards handbook. Office of Water Regulations and Standards, Washington, D.C.
- U.S. EPA. 1985. Technical support document for water quality-based toxics control. Office of Water, Washington, D.C.

Uche, J.F. and E.G. Bligh. 1971. Preliminary survey of heavy metal contamination of Canadian freshwater fish. Jour. Fish. Res. Board Can. 28: 786.

Valiela, I., et al. 1974. Response of salt marsh bivalves to enrichment with metal-containing sewage sludge and retention of lead, zinc and cadmium by marsh sediment. Environ. Pollut. 7: 149.

Van der Werff, M. and M.J. Pruyt. 1982. Long-term effects of heavy metals on aquatic plants. Chemosphere 11: 727.

Varanasi, U. and D.J. Gmur. 1978. Influence of water-borne and dietary calcium on uptake and retention of lead by coho salmon. Toxicol. Appl. Pharmacol. 46: 65.

Varansai, U., et al. 1975. Structural alterations in fish epidermal mucus produced by water-borne lead and mercury. Nature 258: 431.

Vighi, M. 1981. Lead uptake and release in an experimental trophic chain. Ecotoxicol. Environ. Safety 5: 177.

Raymadhavan, K.T. and T. Iwai. 1975. Histochemical observations on the permeation of heavy metals into taste buds of goldfish. Bull. Jap. Soc. Sci. Fish. 41: 631.

- Vinikour, W.S., et al. 1980. Bioconcentration patterns of zinc, copper, cadmium and lead in selected fish species from the Fox River, Illinois. Bull. Environ. Contam. Toxicol. 24: 727.
- Wachs, B. 1982. Concentration of heavy metals in fishes from the river Danube. Z. Wasser Abwasser Forsch. 15: 43.
- Wallen, I.E., et al. 1957. Toxicity to Gambusia affinis of certain pure chemicals in turbid waters. Sew. Ind. Wastes 29: 695.
- Walsh, D.F., et al. 1977. Residues in fish, wildlife and estuaries. Pestic. Monic. Jour. 11: 5.
- Wang, H. 1959. Analyses of a toxic factor, lethal to paramecium present in non-glass-distilled water. Proc. Soc. Exp. Biol. Med. 101: 682.
- Warnick, S.L and H.L. Bell. 1969. The acute toxicity of some heavy metals to different species of aquatic insects. Jour. Water Pollut. Control Fed. 41: 280.
- Waterman, A.J. 1937. Effects of salts of heavy metals on development of the sea urchin, Arbacia punctulata. Biol. Bull. 73: 401.
- Watling, H.R. 1981. Effects of metals on the development of oyster embryos. South African Jour. Sci. 77: 134.

Watling, H.R. 1983. Accumulation of seven metals by Crassostrea gigas, Crassostrea margaritacea, Perna perna, and Choromytilus meridionalis. Bull. Environ. Contam. Toxicol. 30: 317.

Weber, W.J., Jr., and W. Scumm. 1963. Mechanism of hydrogen ion buffering in natural waters. Jour. Am. Water Works Assoc. 55: 1553.

Wehr, J.D. and B.A. Whitton. 1983a. Accumulation of heavy metals by aquatic mosses. 2: Rhynchosetegium riparioides. Hydrobiologia 100: 261.

Wehr, J.D. and B.A. Whitton. 1983b. Accumulation of heavy metals by aquatic mosses. 3: seasonal changes. Hydrobiologia 100: 285.

Weir, P.A. and C.H. Hine. 1970. Effects of various metals on behavior of conditioned goldfish. Arch. Environ. Health. 20: 45.

Weis, J.S. 1976. Effects of mercury, cadmium, and lead salts on regeneration and ecdysis in the fiddler crab, Uca pugilator. Fish. Bull. 74: 464.

Weis, J.S. and P. Weis. 1977. Effect of heavy metals on development of the killifish, Fundulus heteroclitus. Jour. Fish Biol. 11: 49.

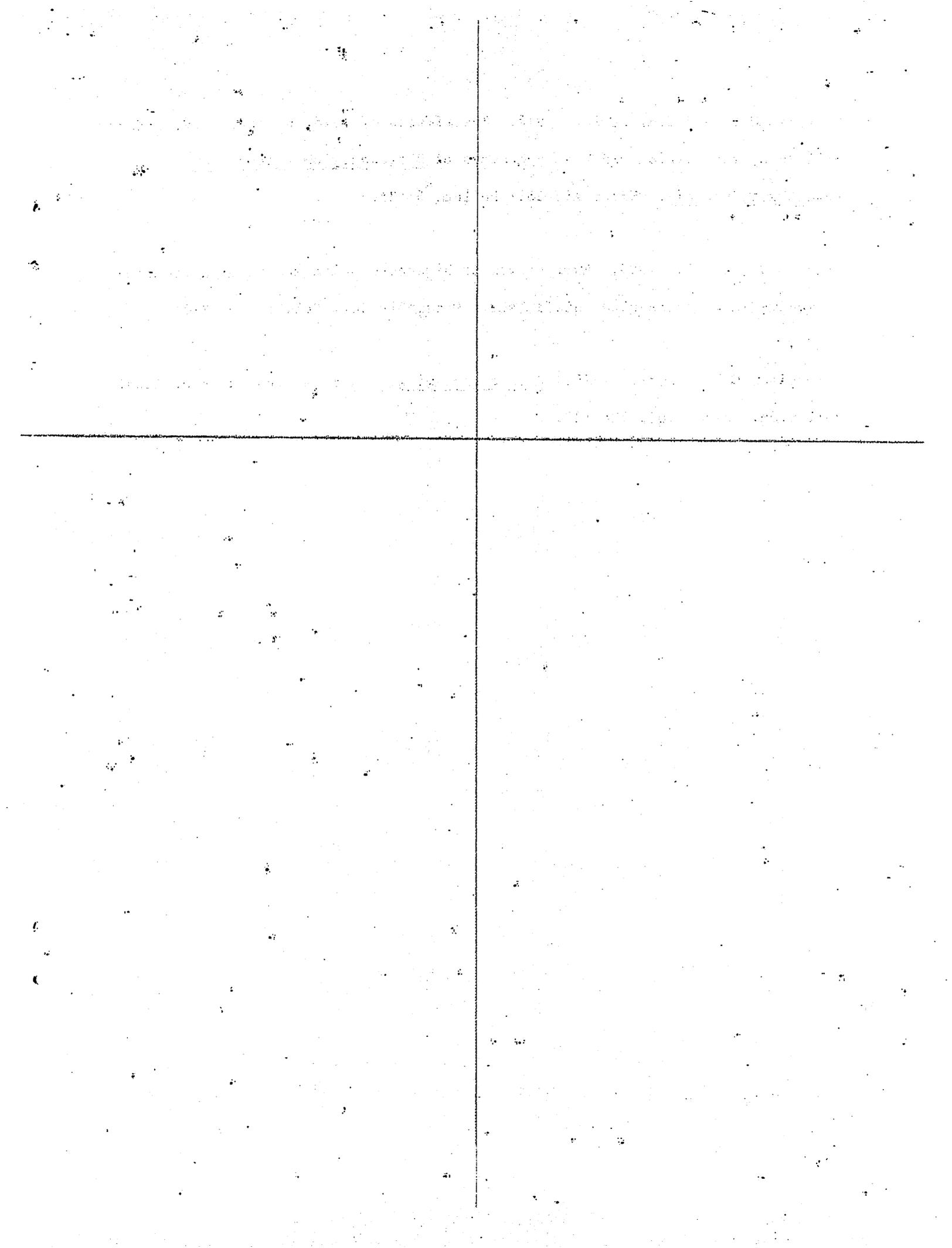
Weis, P. and J.S. Weis. 1982. Toxicity of methyl mercury, mercuric chloride, and lead in killifish (Fundulus heteroclitus) from Southampton, New York. Environ. Res. 28: 364.

- Welsh, R.P. and P. Denny. 1980. The uptake of lead and copper by submerged aquatic macrophytes in two English lakes. *Jour. Ecol.* 68: 443.
- Westfall, B.A. 1945. Coagulation film anoxia in fishes. *Ecology* 26: 283.
- Whitley, L.S. 1968. The resistance of tubificid worms to three common pollutants. *Hydrobiologia* 32: 193.
- Whitton, B.A., et al. 1982. Accumulation of zinc, cadmium, and lead by the aquatic liverwort Scapania. ~~*Environ. Pollut. (Series B)* 3: 299.~~
- Wiener, J.G. and J.P. Giesy. 1979. Concentration of Cd, Cu, Mn, Pb and Zn in fishes of a highly organic softwater pond. *Jour. Fish. Res. Board Can.* 36: 270.
- Wixson, B.G. and E. Bolter. 1972. Evaluations of stream pollution and trace substances in the new lead belt of Missouri. In: D.D. Hemphill (ed.), *Trace Substances in Environmental Health-V*. University of Missouri, Columbia, Missouri. p. 143.
- Wong, P.T.S., et al. 1981. Accumulation and depuration of tetramethyllead by rainbow trout. *Water Res.* 15: 621.
- Wong, P.T.S., et al. 1982. Physiological and biochemical responses of several freshwater algae to a mixture of metals. *Chemosphere* 11: 367.

Woolery, M.L. and R.A. Lewin. 1976. The effects of lead on algae. IV. effects of lead on respiration and photosynthesis of Phaeodactylum cricornutum (Bacillariophyceae). Water Air Soil Pollut. 6: 25.

Wren, C.D., et al. 1983. Examination of bioaccumulation and biomagnification of metals in a precambrian shield lake. Water Air Soil Pollut. 19: 277.

Zarogian, G.E., et al. 1979. Crassostrea virginica as an indicator of lead pollution. Mar. Biol. 52: 189.



**Guidelines for Deriving
Numerical National Water Quality Criteria
for the Protection Of Aquatic Organisms
and Their Uses**

by Charles E. Stephen, Donald I. Mount, David J. Hansen,
John R. Gentile, Gary A. Chapman, and William A. Brungs

Office of Research and Development
Environmental Research Laboratories
Duluth, Minnesota
Narragansett, Rhode Island
Corvallis, Oregon

Notices

This document has been reviewed in accordance with U.S. Environmental Protection Agency policy and approved for publication.

Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

This document is available the public to through the National Technical Information Service (NTIS), 5285 Port Royal Road, Springfield, VA 22161.

Special Note

This December 2010 electronic version of the 1985 Guidelines serves to meet the requirements of Section 508 of the Rehabilitation Act. While converting the 1985 Guidelines to a 508-compliant version, EPA updated the taxonomic nomenclature in the tables of Appendix 1 to reflect changes that occurred since the table were originally produced in 1985. The numbers included for Phylum, Class and Family represent those currently in use from the Integrated Taxonomic Information System, or ITIS, and reflect what is referred to in ITIS as Taxonomic Serial Numbers. ITIS replaced the National Oceanographic Data Center (NODC) taxonomic coding system which was used to create the original taxonomic tables included in the 1985 Guidelines document (NODC, Third Addition - see Introduction). For more information on the NODC taxonomic codes, see <http://www.nodc.noaa.gov/General/CDR-detdesc/taxonomic-v8.html>.

The code numbers included in the reference column of the tables have not been updated from the 1985 version. These code numbers are associated with the old NODC taxonomic referencing system and are simply replicated here for historical purposes. Footnotes may or may not still apply.

EPA is working on a more comprehensive update to the 1985 Guidelines, including new taxonomic tables which better reflect the large number of aquatic animal species known to be propagating in U.S. waters.

Table of Contents

Notices	ii
Table of Contents	iii
Executive Summary	iv
Figure 1	v
Introduction	1
I. Definition of Material of Concern	9
II. Collection of Data	11
III. Required data	11
IV. Final Acute Value	14
V. Final Acute Equation	17
VI. Final Chronic Value	19
VII. Final Chronic Equation	22
VIII. Final Plant Value	25
IX. Final Residue Value	25
X. Other Data	28
XI. Criterion	28
XII. Final Review	29
References	31
Appendix 1. Resident North American Species of Aquatic Animals Used in Toxicity and Bioconcentration Tests	33
Introduction	33
Freshwater Species Table	34
Footnotes for Freshwater Species	42
References for Freshwater Species	43
Saltwater Species Table	45
Footnotes for Saltwater Species	51
References for Saltwater Species	51
Appendix 2. Example Calculation of Final Acute Value, Computer Program, and Printouts	53
A. Example Calculation	53
B. Example Computer Program in BASIC Language for Calculating the FAV	53
C. Example Printouts from Program	54

Executive Summary

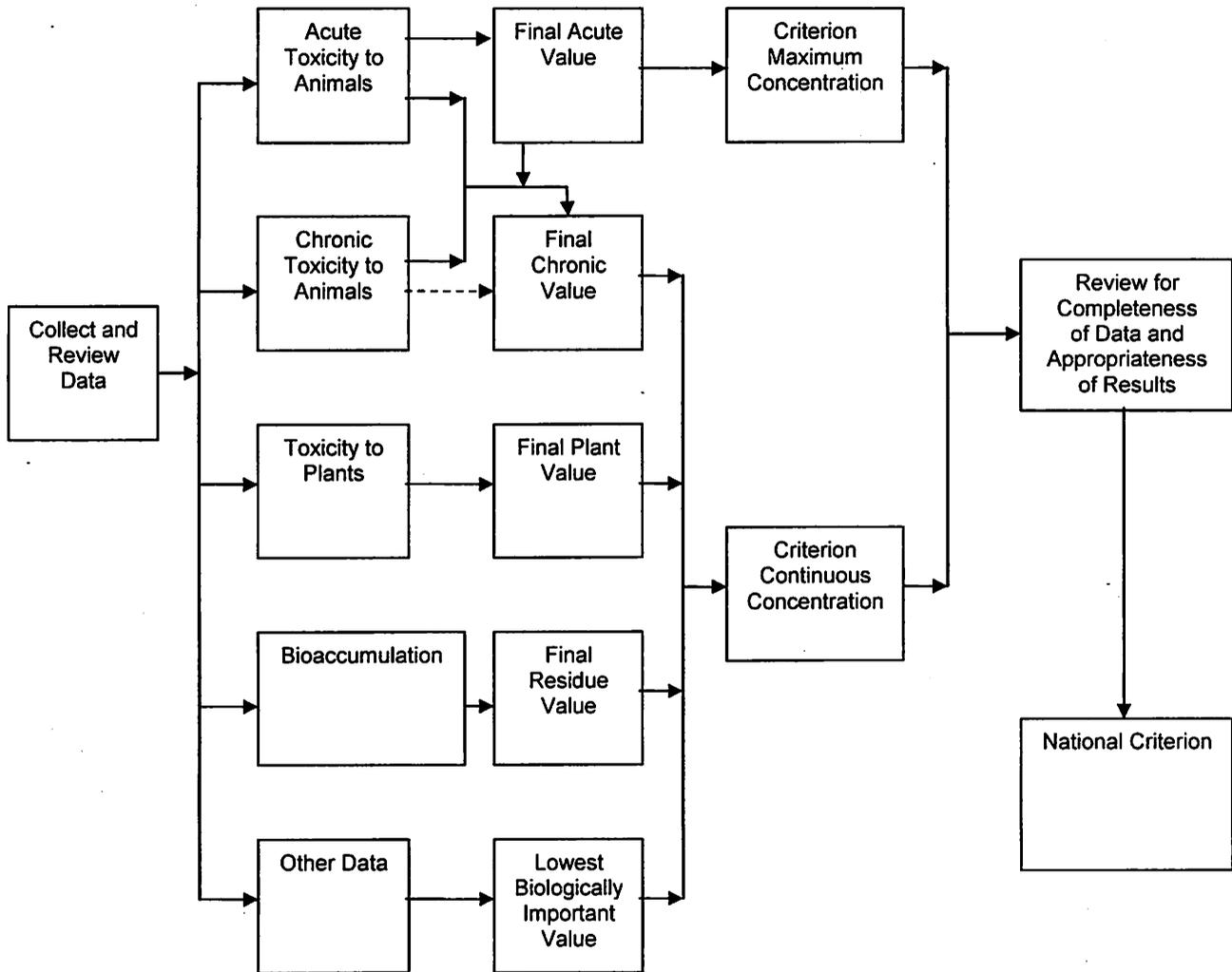
Derivation of numerical national water quality criteria for the protection of aquatic organism and their uses is a complex process (Figure 1) that uses information from many areas of aquatic toxicology. After a decision is made that a national criterion is needed for a particular material, all available information concerning toxicity to, and bioaccumulation by, aquatic organisms is collected, reviewed for acceptability, and sorted. If enough acceptable data on acute toxicity to aquatic animals are available, they are used to estimate the highest one-hour average concentration that should not result in unacceptable effects on aquatic organisms and their uses. If justified, this concentration is made a function of a water quality characteristic such as pH, salinity, or hardness. Similarly, data on the chronic toxicity of the material to aquatic animals are used to estimate the highest four-daily average concentration that should not cause unacceptable toxicity during a long-term exposure. If appropriate, this concentration is also related to a water quality characteristic.

Data on toxicity to aquatic plants are examined to determine whether plants are likely to be unacceptably affected by concentrations that should not cause unacceptable effects on animals. Data on bioaccumulation by aquatic organisms are used to determine if residues might subject edible species to restrictions by the U.S. Food and Drug Administration or if such residues might harm some wildlife consumers of aquatic life. All other available data are examined for adverse effects that might be biologically important.

If a thorough review of the pertinent information indicates that enough acceptable data are available, numerical national water quality criteria are derived for fresh water or salt water or both to protect aquatic organisms and their uses from unacceptable effects due to exposures to high concentrations for short periods of time, lower concentrations for longer periods of time, and combinations of the two.

Figure 1

Derivation of Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses



Introduction

Of the several possible forms of criteria, the numerical form is the most common, but the narrative (e.g., pollutants must not be present in harmful concentrations) and operational (e.g., concentrations of pollutants must not exceed one-tenth of the 96-hr LC50) forms can be used if numerical criteria are not possible or desirable. If it were feasible, a freshwater (or saltwater) numerical aquatic life national criterion* for a material should be determined by conducting field tests on a wide variety of unpolluted bodies of fresh (or salt) water. It would be necessary to add various amounts of the material to each body of water in order to determine the highest concentration that would not cause any unacceptable long-term or short-term effect on the aquatic organisms or their uses. The lowest of these highest concentrations would become the freshwater (or saltwater) national aquatic life water quality criterion for that material, unless one or more of the lowest concentrations were judged to be outliers. Because it is not feasible to determine national criteria by conducting such field tests, these Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses (hereafter referred to as the National Guidelines) describe an objective, internally consistent, appropriate, and feasible way of deriving national criteria, which are intended to provide the same level of protection as the infeasible field testing approach described above.

Because aquatic ecosystems can tolerate some stress and occasional adverse effects, protection of all species at all times and places is not deemed necessary. If acceptable data are available for a large number of appropriate taxa from an appropriate variety of taxonomic and functional groups, a reasonable level of protection will probably be provided if all except a small fraction of the taxa are protected, unless a commercially or recreationally important species is very sensitive. The small fraction is set at 0.05 because other fractions resulted in criteria that seemed too high or too low in comparison with the sets of data from which they were calculated. Use of 0.05 to calculate a Final Acute Value does not imply that this percentage of adversely affected taxa should be used to decide in a field situation whether a criterion is too high or too low or just right.

Determining the validity of a criterion derived for a particular body of water, possibly by modification of a national criterion to reflect local conditions^{1,2,3}, should be based on an operation definition of "protection of aquatic organisms and their uses" that takes into account the practicalities of field monitoring programs and the concerns of the public. Monitoring programs should contain sampling points at enough times and places that all unacceptable changes, whether caused directly or indirectly, will be detected. The programs should adequately monitor the kinds of species of concern to the public, i.e., fish in fresh water and fish and macroinvertebrates in salt water. If the kinds of species of concern cannot be adequately monitored at a reasonable cost, appropriate surrogate species should be monitored. The kinds of species most likely to be good surrogates are those that either (a) are a major food of the desired kinds of species or (b) utilize the same food as the desired species or (c) both. Even if a major adverse effect on appropriate surrogate species does not directly result in an unacceptable effect on the kinds of species of concern to the public, it indicates a high probability that such an effect will occur.

* The term "national criteria" is used herein because it is more descriptive than the synonymous term "section 304(a) criteria", which is used in the Water Quality Standards Regulation [1].

To be acceptable to the public and useful in field situations, protection of aquatic organisms and their uses should be defined as prevention of unacceptable long-term short-term effects on (1) commercially, recreationally, and other important species and (2) (a) fish and benthic invertebrate assemblages in rivers and streams, and (b) fish, benthic invertebrate, and zooplankton assemblages in lakes, reservoirs, estuaries, and oceans. Monitoring programs intended to be able to detect unacceptable effects should be tailored to the body of water of concern so that necessary samples are obtained at enough times and places to provide adequate data on the populations of the important species, as well as data directly related to the reasons for their being considered important. For example, for substances that are residue limited, species that are consumed should be monitored for contaminants to ensure that wildlife predators are protected, FDA action levels are not exceeded, and flavor is not impaired. Monitoring programs should also provide data on the number of taxa and number of individuals in the above-named assemblages that can be sampled at reasonable cost. The amount of decrease in the number of taxa or number of individuals in an assemblage that should be considered unacceptable should take into account appropriate features of the body of water and its aquatic community. Because most monitoring programs can only detect decreases of more than 20 percent, any statistically significant decrease should usually be considered unacceptable. The insensitivity of most monitoring programs greatly limits their usefulness for studying the validity of criteria because unacceptable changes can occur and not be detected. Therefore, although limited field studies can sometimes demonstrate that criteria are underprotective, only high quality field studies can reliably demonstrate that criteria are not underprotective.

If the purpose of water quality criteria were to protect only commercially and recreationally important species, criteria specifically derived to protect such species and their uses from the direct adverse effects of a material would probably, in most situations, also protect those species from indirect adverse effects due to effects of the material on other species in the ecosystem. For example, in most situations either the food chain would be more resistant than the important species and their uses or the important species and their food chains would be adaptable enough to overcome effects of the material on portions of the food chains.

These National Guidelines have been developed on the theory that effects which occur on a species in appropriate laboratory tests will generally occur on the same species in comparable field situations. All North American bodies of water and resident aquatic species and their uses are meant to be taken into account, except for a few that may be too atypical, such as the Great Salt Lake, brine shrimp, and the siscowet subspecies of lake trout, which occurs in Lake Superior and contains up to 67% fat in the fillets⁴. Derivation of criteria specifically for the Great Salt Lake or Lake Superior might have to take brine shrimp and siscowet, respectively, into account.

Numerical aquatic life criteria derived using these National Guidelines are expressed as two numbers, rather than the traditional one number, so that the criteria more accurately reflect toxicological and practical realities. If properly derived and used, the combination of a maximum concentration and a continuous concentration should provide an appropriate degree of protection of aquatic organisms and their uses from acute and chronic toxicity to animals, toxicity to plants, and bioaccumulation by aquatic organisms, without being as restrictive as a one-number criterion would have to be in order to provide the same degree of protection.

Criteria produced by these Guidelines are intended to be useful for developing water quality standards, mixing zone standards, effluent limitations, etc. The development of such standards

and limitations, however, might have to take into account such additional factors as social, legal, economic, and hydrological considerations, the environmental and analytical chemistry of the material, the extrapolation from laboratory data to field situations, and relationships between species for which data are available and species in the body of water of concern. As an intermediate step in the development of standards, it might be desirable to derive site-specific criteria by modification of national criteria to reflect such local conditions as water quality, temperature, or ecologically important species^{1,2,3}. In addition, with appropriate modifications these National Guidelines can be used to derive criteria for any specific geographical area, body of water (such as the Great Salt Lake), or group of similar bodies of water, if adequate information is available concerning the effects of the material of concern on appropriate species and their uses.

Criteria should attempt to provide a reasonable and adequate amount of protection with only a small possibility of considerable overprotection or underprotection. It is not enough that a national criterion be the best estimate that can be obtained using available data; it is equally important that a criterion be derived only if adequate appropriate data are available to provide reasonable confidence that it is a good estimate. Therefore, these National Guidelines specify certain data that should be available if a numerical criterion is to be derived. If all the required data are not available, usually a criterion should not be derived. On the other hand, the availability of all required data does not ensure that a criterion can be derived.

A common belief is that national criteria are based on "worst case" assumptions and that local considerations will raise, but not lower, criteria. For example, it will usually be assumed that if the concentration of a material in a body of water is lower than the national criterion, no unacceptable effects will occur and no site-specific criterion needs to be derived. If, however, the concentration of a material in a body of water is higher than the national criterion, it will usually be assumed that a site-specific criterion should be derived. In order to prevent the assumption of the "worst case" nature of national criteria from resulting in the underprotection of too many bodies of water, national criteria must be intended to protect all or almost all bodies of water. Thus, if bodies of water and the aquatic communities in them do differ substantially in their sensitivities to a material, national criteria should be at least somewhat overprotective for a majority of the bodies of water. To do otherwise would either (a) require derivation of site-specific criteria even if the site-specific concentration were substantially below the national criterion or (b) cause the "worst case" assumption to result in the underprotection of numerous bodies of water. On the other hand, national criteria are probably underprotective of some bodies of water.

The two factors that will probably cause the most difference between national and site-specific criteria are the species that will be exposed and the characteristics of the water. In order to ensure that national criteria are appropriately protective, the required data for national criteria include some species that are sensitive to many materials and national criteria are specifically based on tests conducted in water relatively low in particulate matter and organic matter. Thus, the two factors that will usually be considered in the derivation of site-specific criteria from national criteria are used to help ensure that national criteria are appropriately protective.

On the other hand, some local conditions might require that site-specific criteria be lower than national criteria. Some untested locally important species might be very sensitive to the material of concern, and local water quality might not reduce the toxicity of the material. In addition,

aquatic organisms in field situations might be stressed by diseases, parasites, predators, other pollutants, contaminated or insufficient food, and fluctuating and extreme conditions of flow, water quality, and temperature. Further, some materials might degrade to more toxic materials, or some important community functions or species interactions might be adversely affected by concentrations lower than those that affect individual species.

Criteria must be used in a manner that is consistent with the way in which they were derived if the intended level of protection is to be provided in the real world. Although derivation of water quality criteria for aquatic life is constrained by the ways toxicity and bioconcentration tests are usually conducted, there are still many different ways that criteria can be derived, expressed, and used. The means used to derive and state criteria should relate, in the best possible way, the kinds of data that are available concerning toxicity and bioconcentration and the ways criteria can be used to protect aquatic organisms and their uses.

The major problem is to determine the best way that the statement of a criterion can bridge the gap between the nearly constant concentrations used in most toxicity and bioconcentration tests and the fluctuating concentrations that usually exist in the real world. A statement of a criterion as a number that is not to be exceeded any time or place is not acceptable because few, if any, people who use criteria would take it literally and few, if any, toxicologists would defend a literal interpretation. Rather than try to reinterpret a criterion that is neither useful nor valid, it is better to develop a more appropriate way of stating criteria.

Although some materials might not exhibit thresholds, many materials probably do. For any threshold material, continuous exposure to any combination of concentrations below the threshold will not cause an unacceptable effect (as defined on pages 1 and 2) on aquatic organisms and their uses, except that the concentration of a required trace nutrient might be too low. However, it is important to note that this is a threshold of unacceptable effect, not a threshold of adverse effect. Some adverse effect, possibly even a small reduction in the survival, growth, or reproduction of a commercially or recreationally important species, will probably occur at, and possibly even below, the threshold. The Criterion Continuous Concentration (CCC) is intended to be a good estimate of this threshold of unacceptable effect. If maintained continuously, any concentration above the CCC is expected to cause an unacceptable effect. On the other hand, the concentration of a pollutant in a body of water can be above the CCC without causing an unacceptable effect if (a) the magnitudes and durations of the excursions above the CCC are appropriately limited and (b) there are compensating periods of time during which the concentration is below the CCC. The higher the concentration is above the CCC, the shorter the period of time it can be tolerated. But it is unimportant whether there is any upper limit on concentrations that can be tolerated instantaneously or even for one minute because concentrations outside mixing zones rarely change substantially in such short periods of time.

An elegant, general approach to the problem of defining conditions (a) and (b) would be to integrate the concentration over time, taking into account uptake and depuration rates, transport within the organism to a critical site, etc. Because such an approach is not currently feasible, an approximate approach is to require that the average concentration not exceed the CCC. The average concentration should probably be calculated as the arithmetic average rather than the geometric mean⁵. If a suitable averaging period is selected, the magnitudes and durations of concentrations above the CCC will be appropriately limited, and suitable compensating periods below the CCC will be required.

In the elegant approach mentioned above, the uptake and depuration rates would determine the effective averaging period, but these rates are likely to vary from species to species for any particular material. Thus the elegant approach might not provide a definitive answer to the problem of selecting an appropriate averaging period. An alternative is to consider that the purpose of the averaging period is to allow the concentration to be above the CCC only if the allowed fluctuating concentrations do not cause more adverse effect than would be caused by a continuous exposure to the CCC. For example, if the CCC caused a 10% reduction in growth of rainbow trout, or a 13% reduction in survival of oysters, or a 7% reduction in reproduction of smallmouth bass, it is the purpose of the averaging period to allow concentrations above the CCC only if the total exposure will not cause any more adverse effect than continuous exposure to the CCC would cause.

Even though only a few tests have compared the effects of a constant concentration with the effects of the same average concentration resulting from a fluctuating concentration, nearly all the available comparisons have shown that substantial fluctuations result in increased adverse effects^{5,6}. Thus if the averaging period is not to allow increased adverse effects, it must not allow substantial fluctuations. Life-cycle tests with species such as mysids and daphnids and early life-stage tests with warmwater fishes usually last for 20 to 30 days. An averaging period that is equal to the length of the test will obviously allow the worst possible fluctuations and would very likely allow increased adverse effects.

An averaging period of four days seems appropriate for use with the CCC for two reasons. First, it is substantially shorter than the 20 to 30 days that is obviously unacceptable. Second, for some species it appears that the results of chronic tests are due to the existence of a sensitive life stage at some time during the test⁷, rather than being caused by either long-term stress or long-term accumulation of the test material in the organism. The existence of a sensitive life stage is probably the cause of acute-chronic ratios that are not much greater than 1, and is also possible when the ratio is substantially greater than 1. In addition, some experimentally determined acute-chronic ratios are somewhat less than 1, possibly because prior exposure during the chronic test increased the resistance of the sensitive life stage⁸. A four-day averaging period will probably prevent increased adverse effects on sensitive life stages by limiting the durations and magnitudes of exceedences* of the CCC.

The considerations applied to interpretation of the CCC also apply to the CMC. For the CMC the averaging period should again be substantially less than the lengths of the tests it is based on, i.e., substantially less than

48 to 96 hours. One hour is probably an appropriate averaging period because high concentrations of some materials can cause death in one to three hours. Even when organisms do not die within the first hour or so, it is not known how many might have died due to delayed effects of this short of an exposure. Thus it is not appropriate to allow concentrations above the CMC to exist for as long as one hour.

The durations of the averaging periods in national criteria have been made short enough to restrict allowable fluctuations in the concentration of the pollutant in the receiving water and to restrict the length of time that the concentration in the receiving water can be continuously above

* Although "exceedence" has not been found in any dictionary, it is used here because it is not appropriate to use "violation" in conjunction with criteria, no other word seems appropriate, and all appropriate phrases are awkward.

a criterion concentrations. The statement of a criterion could specify that the four-day average should never exceed the CCC and that the one-hour average should never exceed the CMC. However, one of the most important uses of criteria is for designing waste treatment facilities. Such facilities are designed based on probabilities and it is not possible to design for a zero probability. Thus, one of the important design parameters is the probability that the four-day average or the one-hour-average will be exceeded, or, in other words, the frequency with which exceedences will be allowed.

The frequency of allowed exceedences should be based on the ability of aquatic ecosystems to recover from the exceedences, which will depend in part on the magnitudes and durations of the exceedences. It is important to realize that high concentrations caused by spills and similar major events are not what is meant by an "exceedence", because spills and other accidents are not part of the design of the normal operation of waste treatment facilities. Rather, exceedences are extreme values in the distribution of ambient concentrations and this distribution is the result of the usual variations in the flows of both the effluent and the receiving water and the usual variations in the concentrations of the material of concern in both the effluent and in the upstream receiving water. Because exceedences are the result of usual variation, most of the exceedences will be small and exceedences as large as a factor of two will be rare. In addition, because these exceedences are due to random variation, they will not be evenly spaced. In fact, because many receiving waters have both one-year and multi-year cycles and many treatment facilities have daily, weekly, and yearly cycles, exceedences will often be grouped, rather than being evenly spaced or randomly distributed. If the flow of the receiving water is usually much greater than the flow of the effluent, normal variation and the flow cycles will result in the ambient concentration usually being below the CCC, occasionally being near the CCC, and rarely being above the CCC. In addition, exceedences that do occur will be grouped. On the other hand, if the flow of the effluent is much greater than the flow of the receiving water, the concentration might be close to the CCC much of the time and rarely above the CCC, with exceedences being randomly distributed.

The abilities of ecosystems to recover differ greatly, and depend on the pollutant, the magnitude and duration of the exceedence, and the physical and biological features of the ecosystem. Documented studies of recoveries are few, but some systems recover from small stresses in six weeks whereas other systems take more than ten years to recover from severe stress³. Although most exceedences are expected to be very small, larger exceedences will occur occasionally. Most aquatic ecosystems can probably recover from most exceedences in about three years. Therefore, it does not seem reasonable to purposely design for stress above that caused by the CCC to occur more than once every three years on the average, just as it does not seem reasonable to require that these kinds of stresses only occur once every five or ten years on the average.

If the body of water is not subject to anthropogenic stress other than the exceedences of concern and if exceedences as large as a factor of two are rare, it seems reasonable that most bodies of water could tolerate exceedences once every three years on the average. In situations in which exceedences are grouped, several exceedences might occur in one or two years, but then there will be, for example, 10 to 20 years during which no exceedences will occur and the concentration will be substantially below the CCC most of the time. In situations in which the concentration is often close to the CCC and exceedences are randomly distributed, some adverse effect will occur regularly, and small additional, unacceptable effects will occur about every

third year. The relative long-term ecological consequences of evenly spaced and grouped exceedences are unknown, but because most exceedences will probably be small, the long-term consequences should be about equal over long periods of time.

The above considerations lead to a statement of a criterion in the frequency-intensity-duration format that is often used to describe rain and snow fall and stream flow, e.g., how often, on the average, does more than ten inches of rain fall in a week? The numerical values chosen for frequency (or average recurrence interval), intensity (i.e., concentration), and duration (of averaging period) are those appropriate for national criteria. Whenever adequately justified, a national criterion may be replaced by a site-specific criterion¹, which may include not only site-specific criterion concentrations², but also site-specific durations of averaging periods and site-specific frequencies of allowed exceedences³.

The concentrations, durations, and frequencies specified in criteria are based on biological, ecological, and toxicological data, and are designed to protect aquatic organisms and their uses from unacceptable effects. Use of criteria for designing waste treatment facilities requires selections of an appropriate wasteload allocation model. Dynamic models are preferred for the application of water quality criteria, but a steady-state model might have to be used instead of a dynamic model in some situations. Regardless of the model that is used, the durations of the averaging periods and the frequencies of allowed exceedences must be applied correctly if the intended level of protection is to be provided. For example, in the criterion statement frequency refers to the average frequency, over a long period of time, of rare events (i.e., exceedences). However, in some disciplines, frequency is often thought of in terms of the average frequency, over a long period of time, of the years in which rare events occur, without any consideration of how many rare events occur within each of those eventful years. The distinction between the frequency of events and the frequency of years is important for all those situations in which the rare events, e.g., exceedences, tend to occur in groups within the eventful years. The two ways of calculating frequency produce the same results in situations in which each rare event occurs in a different year because then the frequency of events is the same as the frequency of eventful years.

Because fresh water and salt water have basically different chemical compositions and because freshwater and saltwater (i.e., estuarine and true marine) species rarely inhabit the same water simultaneously, these National Guidelines provide for the derivation of separate criteria for these two kinds of water. For some materials sufficient data might not be available to allow derivation of criteria for one or both kinds of water. Even though absolute toxicities might be different in fresh and salt waters, such relative data as acute-chronic ratios and bioconcentration factors often appear to be similar in the two waters. When data are available to indicate that these ratios and factors are probably similar, they are used interchangeably.

The material for which a criterion is desired is usually defined in terms of a particular chemical compound or ion, or a group of closely related compounds or ions, but it might possibly be defined in terms of an effluent. These National Guidelines might also be useful for deriving criteria for temperature, dissolved oxygen, suspended solids, pH, etc., if the kinds of data on which the Guidelines are based are available.

Because they are meant to be applied only after a decision has been made that a national water quality criterion for aquatic organisms is needed for a material, these National Guidelines do not

address the rationale for making that decision. If the potential for adverse effects on aquatic organisms and their uses is part of the basis for deciding whether an aquatic life criterion is needed for a material, these Guidelines will probably be helpful in the collection and interpretation of relevant data. Such properties as volatility might affect the fate of a material in the aquatic environment and might be important when determining whether a criterion is needed for a material; for example, aquatic life criteria might not be needed for materials that are highly volatile or highly degradable in water. Although such properties can affect how much of the material will get from the point of discharge through any allowed mixing zone to some portion of the ambient water and can also affect the size of the zone of influence in the ambient water, such properties do not affect how much of the material aquatic organisms can tolerate in the zone of influence.

This version of the National Guidelines provides clarifications, additional details, and technical and editorial changes from the previous version ⁹. These modifications are the result of comments on the previous version and subsequent drafts ¹⁰, experience gained during the U.S. EPA's use of previous versions and drafts, and advances in aquatic toxicology and related fields. Future versions will incorporate new concepts and data as their usefulness is demonstrated. The major technical changes incorporated into this version of the National Guidelines are:

1. The requirement for acute data for freshwater animals has been changed to include more tests with invertebrate species. The taxonomic, functional, and probably the toxicological, diversities among invertebrate species are greater than those among vertebrate species and this should be reflected in the required data.
2. When available, 96-hr EC50s based on the percentage of fish immobilized plus the percentage of fish killed are used instead of 96-hr LC50s for fish; comparable EC50s are used instead of LC50s for other species. Such appropriately defined EC50s better reflect the total severe acute adverse impact of the test material on the test species than do LC50s or narrowly defined EC50s. Acute EC50s that are based on effects that are not severe, such as reduction in shell deposition and reduction in growth, are not used in calculating the Final Acute Value.
3. The Final Acute Value is now defined in terms of Genus Mean Acute Values rather than Species Mean Acute Values. A Genus Mean Acute Value is the geometric mean of all the Species Mean Acute Values available for species in the genus. On the average, species within a genus are toxicologically much more similar than species in different genera, and so the use of Genus Mean Acute Values will prevent data sets from being biased by an overabundance of species in one or a few genera.
4. The Final Acute Value is now calculated using a method ¹¹ that is not subject to the bias and anomalous behavior that the previous method was. The new method is also less influenced by one very low value because it always gives equal weight to the four values that provide the most information about the cumulative probability of 0.05. Although the four values receive the most weight, the other values do have a substantial effect on the Final Acute Value (see examples in Appendix 2).
5. The requirements for using the results of tests with aquatic plants have been made more stringent.

6. Instead of being equal to the Final Acute Value, the Criterion Maximum Concentration is now equal to one-half the Final Acute Value. The Criterion Maximum Concentration is intended to protect 95 percent of a group of diverse genera, unless a commercially or recreationally important species is very sensitive. However, a concentration that would severely harm 50 percent of the fifth percentile or 50 percent of a sensitive important species cannot be considered to be protective of that percentile or that species. Dividing the Final Acute Value by 2 is intended to result in a concentration that will not severely adversely affect too many of the organisms.
7. The lower of the two numbers in the criterion is now called the Criterion Continuous Concentration, rather than the Criterion Average Concentration, to more accurately reflect the nature of the toxicological data on which it is based.
8. The statement of a criterion has been changed (a) to include durations of averaging periods and frequencies of allowed exceedences that are based on what aquatic organisms and their uses can tolerate, and (b) to identify a specific situation in which site-specific criteria ^{1,2,3} are probably desirable.

In addition, Appendix 1 was added to aid in determining whether a species should be considered resident in North America and its taxonomic classification. Appendix 2 explains the calculation of the Final Acute Value.

The amount of guidance in these National Guidelines has been increased, but much of the guidance is necessarily qualitative rather than quantitative; much judgment will usually be required to derive a water quality criterion for aquatic organisms and their uses. In addition, although this version of the National Guidelines attempts to cover all major questions that have arisen during use of previous versions and drafts, it undoubtedly does not cover all situations that might occur in the future. All necessary decisions should be based on a thorough knowledge of aquatic toxicology and an understanding of these Guidelines and should be consistent with the spirit of these Guidelines, i.e., to make best use of the available data to derive the most appropriate criteria. These National Guidelines should be modified whenever sound scientific evidence indicates that a national criterion produced using these Guidelines would probably be substantially overprotective or underprotective of the aquatic organisms and their uses on a national basis. Derivation of numerical national water quality criteria for aquatic organisms and their uses is a complex process and requires knowledge in many areas of aquatic toxicology; any deviation from these Guidelines should be carefully considered to ensure that it is consistent with other parts of these Guidelines.

I. Definition of Material of Concern

- A. Each separate chemical that does not ionize substantially in most natural bodies of water should usually be considered a separate material, except possibly for structurally similar organic compounds that only exist in large quantities as commercial mixtures of various compounds and apparently have similar biological, chemical, physical, and toxicological properties.
- B. For chemicals that do ionize substantially in most natural bodies of water (e.g., some phenols and organic acids, some salts of phenols and organic acids, and most

inorganic salts and coordination complexes of metals), all forms that would be in chemical equilibrium should usually be considered one material. Each different oxidation state of a metal and each different nonionizable covalently bonded organometallic compound should usually be considered a separate material.

- C. The definition of the material should include an operational analytical component. Identification of a material simply, for example, as "sodium" obviously implies "total sodium", but leaves room for doubt. If "total" is meant, it should be explicitly stated. Even "total" has different operational definitions, some of which do not necessarily measure "all that is there" in all samples. Thus, it is also necessary to reference or describe the analytical method that is intended. The operational analytical component should take into account the analytical and environmental chemistry of the material, the desirability of using the same analytical method on samples from laboratory tests, ambient water, and aqueous effluents, and various practical considerations, such as labor and equipment requirements and whether the method would require measurement in the field or would allow measurement after samples are transported to a laboratory.

The primary requirements of the operational analytical component are that it be appropriate for use on samples of receiving water, that it be compatible with the available toxicity and bioaccumulation data without making extrapolations that are too hypothetical, and that it rarely result in underprotection or overprotection of aquatic organisms and their uses. Because an ideal analytical measurement will rarely be available, a compromise measurement will usually have to be used. This compromise measurement must fit with the general approach that if an ambient concentration is lower than the national criterion, unacceptable effects will probably not occur, i.e., the compromise measurement must not err on the side of underprotection when measurements are made on a surface water. Because the chemical and physical properties of an effluent are usually quite different from those of the receiving water, an analytical method that is acceptable for analyzing an effluent might not be appropriate for analyzing a receiving water, and vice versa. If the ambient concentration *calculated* from a measured concentration in an effluent is higher than the national criterion, an additional option is to *measure* the concentration after dilution of the effluent with receiving water to determine if the measured concentration is lowered by such phenomena as complexation or sorption. A further option, of course, is to derive a site-specific criterion^{1,2,3}. Thus, the criterion should be based on an appropriate analytical measurement, but the criterion is not rendered useless if an ideal measurement either is not available or is not feasible.

NOTE: The analytical chemistry of the material might have to be taken into account when defining the material or when judging the acceptability of some toxicity tests, but a criterion should not be based on the sensitivity of an analytical method. When aquatic organisms are more sensitive than routine analytical methods, the proper solution is to develop better analytical methods, not to underprotect aquatic life.

II. Collection of Data

- A. Collect all available data on the material concerning (a) toxicity to, and bioaccumulation by, aquatic animals and plants, (b) FDA action levels ¹², and (c) chronic feeding studies and long-term field studies with wildlife species that regularly consume aquatic organisms.
- B. All data that are used should be available in typed, dated, and signed hard copy (publication, manuscript, letter, memorandum, etc.) with enough supporting information to indicate that acceptable test procedures were used and that the results are probably reliable. In some cases it may be appropriate to obtain additional written information from the investigator, if possible. Information that is confidential or privileged or otherwise not available for distribution should not be used.
- C. Questionable data, whether published or unpublished, should not be used. For example, data should usually be rejected if they are from tests that did not contain a control treatment, tests in which too many organisms in the control treatment died or showed signs of stress or disease, and tests in which distilled or deionized water was used as the dilution water without addition of appropriate salts.
- D. Data on technical grade materials may be used if appropriate, but data on formulated mixtures and emulsifiable concentrates of the material of concern should not be used.
- E. For some highly volatile, hydrolyzable, or degradable materials it is probably appropriate to use only results of flow-through tests in which the concentrations of test material in the test solutions were measured often enough using acceptable analytical methods.
- F. Data should be rejected if they were obtained using:
 - 1. Brine shrimp, because they usually only occur naturally in water with salinity greater than 35 g/kg.
 - 2. Species that do not have reproducing wild populations in North America (see Appendix 1).
 - 3. Organisms that were previously exposed to substantial concentrations of the test material or other contaminants.
- G. Questionable data, data on formulated mixtures and emulsifiable concentrates, and data obtained with non-resident species in North America or previously exposed organisms may be used to provide auxiliary information but should not be used in the derivation of criteria.

III. Required data

- A. Certain data should be available to help ensure that each of the four major kinds of possible adverse effects receives adequate consideration. Results of acute and chronic toxicity tests with representative species of aquatic animals are necessary so that data available for tested species can be considered a useful indication of the sensitivities of

appropriate untested species. Fewer data concerning toxicity to aquatic plants are required because procedures for conducting tests with plants and interpreting the results of such tests are not as well developed. Data concerning bioaccumulation by aquatic organisms are only required if relevant data are available concerning the significance of residues in aquatic organisms.

- B. To derive a criterion for freshwater aquatic organisms and their uses, the following should be available:
1. Results of acceptable acute tests (see Section IV) with at least one species of freshwater animal in at least eight different families such that all of the following are included:
 - a. the family Salmonidae in the class Osteichthyes
 - b. a second family in the class Osteichthyes, preferably a commercially or recreationally important warmwater species (e.g., bluegill, channel catfish, etc.)
 - c. a third family in the phylum Chordata (may be in the class Osteichthyes or may be an amphibian, etc.)
 - d. a planktonic crustacean (e.g., cladoceran, copepod, etc.)
 - e. a benthic crustacean (e.g., ostracod, isopod, amphipod, crayfish, etc.)
 - f. an insect (e.g., mayfly, dragonfly, damselfly, stonefly, caddisfly, mosquito, midge, etc.)
 - g. a family in a phylum other than Arthropoda or Chordata (e.g., Rotifera, Annelida, Mollusca, etc.)
 - h. a family in any order of insect or any phylum not already represented.
 2. Acute-chronic ratios (see Section VI) with species of aquatic animals in at least three different families provided that one of the three species:
 - at least one is a fish
 - at least one is an invertebrate
 - at least one is an acutely sensitive freshwater species (the other two may be saltwater species).
 3. Results of at least one acceptable test with a freshwater alga or vascular plant (see Section VIII). If plants are among the aquatic organisms that are most sensitive to the material, results of a test with a plant in another phylum (division) should also be available.

4. At least one acceptable bioconcentration factor determined with an appropriate freshwater species, if a maximum permissible tissue concentration is available (see Section IX).
- C. To derive a criterion for saltwater aquatic organisms and their uses, the following should be available:
1. Results of acceptable acute tests (see Section IV) with at least one species of saltwater animal in at least eight different families such that all of the following are included:
 - a. two families in the phylum Chordata
 - b. a family in a phylum other than Arthropoda or Chordata
 - c. either the Mysidae or Penaeidae family
 - d. three other families not in the phylum Chordata (may include Mysidae or Penaeidae, whichever was not used above)
 - e. any other family.
 2. Acute-chronic ratios (see Section VI) with species of aquatic animals in at least three different families provided that of the three species:
 - at least one is a fish
 - at least one is an invertebrate
 - at least one is an acutely sensitive saltwater species (the other two may be freshwater species).
 3. Results of at least one acceptable test with a saltwater alga or vascular plant (see Section VIII). If plants are among the aquatic organisms most sensitive to the material, results of a test with a plant in another phylum (division) should also be available.
 4. At least one acceptable bioconcentration factor determined with an appropriate saltwater species, if a maximum permissible tissue concentration is available (see Section IX).
- D. If all the required data are available, a numerical criterion can usually be derived, except in special cases. For example, derivation of a criterion might not be possible if the available acute-chronic ratios vary by more than a factor of ten with no apparent pattern. Also, if a criterion is to be related to a water quality characteristic (see Sections V and VII), more data will be necessary.

Similarly, if all required data are not available, a numerical criterion should not be derived except in special cases. For example, even if not enough acute and chronic data are available, it might be possible to derive a criterion if the available data

clearly indicate that the Final Residue Value should be much lower than either the Final Chronic Value or Final Plant Value.

- E. Confidence in a criterion usually increases as the amount of available pertinent data increases. Thus, additional data are usually desirable.

IV. Final Acute Value

- A. Appropriate measures of the acute (short-term) toxicity of the material to a variety of species of aquatic animals are used to calculate the Final Acute Value. The Final Acute Value is an estimate of the concentration of the material corresponding to a cumulative probability of 0.05 in the acute toxicity values for the genera with which acceptable acute tests have been conducted on the material. However, in some cases, if the Species Mean Acute Value of a commercially or recreationally important species is lower than the calculated Final Acute Value, then that Species Mean Acute Value replaces the calculated Final Acute Value in order to provide protection for that important species.
- B. Acute toxicity tests should have been conducted using acceptable procedures ¹³.
- C. Except for test with saltwater annelids and mysids, 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.
- D. Results of acute tests conducted in unusual dilution water, e.g., dilution water in which total organic carbon or particulate matter exceeded 5 mg/L, should not be used, unless a relationship is developed between acute toxicity and organic carbon or particulate matter or unless data show that organic carbon, particulate matter, etc., do not affect toxicity.
- E. Acute values should be based on endpoints which reflect the total severe acute adverse impact of the test material on the organisms used in the test. Therefore, only the following kinds of data on acute toxicity to aquatic animals should be used:
 - 1. Tests with daphnids and other cladocerans should be started with organisms less than 24 hours old and tests with midges should be started with second- or third-instar larvae. The result should be the 48-hr EC50 based on percentage of organisms immobilized plus percentage of organisms killed. If such an EC50 is not available from a test, the 48-hr LC50 should be used in place of the desired 48-hr EC50. An EC50 or LC50 of longer than 48 hr can be used as long as the animals were not fed and the control animals were acceptable at the end of the test.
 - 2. The result of a test with embryos and larvae of barnacles, bivalve molluscs (clams, mussels, oysters, and scallops), sea urchins, lobsters, crabs, shrimp, and abalones, should be the 96-hr EC50 based on the percentage of organisms with incompletely developed shells plus the percentage of organisms killed. If such an EC50 is not available from a test, the lower of the 96-hr EC50 based on the percentage of organisms with incompletely developed shells and the 96-hr LC50

should be used in place of the desired 96-hr EC50. If the duration of the test was between 48 and 96 hr, the EC50 or LC50 at the end of the test should be used.

3. The acute values from tests with all other freshwater and saltwater animal species and older life stages of barnacles, bivalve molluscs, sea urchins, lobsters, crabs, shrimps, and abalones should be the 96-hr EC50 based on the percentage of organisms exhibiting loss of equilibrium plus the percentage of organisms immobilized plus the percentage of organisms killed. If such an EC50 is not available from a test, the 96-hr LC50 should be used in place of the desired 96-hr EC50.
 4. Tests with single-celled organisms are not considered acute tests, even if the duration was 96 hours or less.
 5. If the tests were conducted properly, acute values reported as "greater than" values and those which are above the solubility of the test material should be used, because rejection of such acute values would unnecessarily lower the Final Acute Value by eliminating acute values for resistant species.
- F. If the acute toxicity of the material to aquatic animals apparently has been shown to be related to a water quality characteristic such as hardness or particulate matter for freshwater animals or salinity or particulate matter for saltwater animals, a Final Acute Equation should be derived based on that water quality characteristic. Go to Section V.
- G. If the available data indicate that one or more life stages are at least a factor of two more resistant than one or more other life stages of the same species, the data for the more resistant life stages should not be used in the calculation of the Species Mean Acute Value because a species can only be considered protected from acute toxicity if all life stages are protected.
- H. The agreement of the data within and between species should be considered. Acute values that appear to be questionable in comparison with other acute and chronic data for the same species and for other species in the same genus probably should not be used in calculation of a Species Mean Acute Value. For example, if the acute values available for a species or genus differ by more than a factor of 10, some or all of the values probably should not be used in calculations.
- I. For each species for which at least one acute value is available, the Species Mean Acute Value (SMAV) should be calculated as the geometric mean of the results of all flow-through tests in which the concentrations of test material were measured. For a species for which no such result is available, the SMAV 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 test materials if measured concentrations are not available) of test material.

NOTE: Data reported by original investigators should not be rounded off. Results of all intermediate calculations should be rounded ¹⁴ to four significant digits.

NOTE: The geometric mean of N numbers is the Nth root of the product of the N numbers. Alternatively, the geometric mean can be calculated by adding the logarithms of the N numbers, dividing the sum by N, and taking the antilog of the quotient. The geometric mean of two numbers is the square root of the product of the two numbers, and the geometric mean of one number is that number. Either natural (base e) or common (base 10) logarithms can be used to calculate geometric means as long as they are used consistently within each set of data, i.e., the antilog used must match the logarithm used.

NOTE: Geometric means, rather than arithmetic means, are used here because the distributions of sensitivities of individual organisms in toxicity tests on most materials and the distributions of sensitivities of species within a genus are more likely to be lognormal than normal. Similarly, geometric means are used for acute-chronic ratios and bioconcentration factors because quotients are likely to be closer to lognormal than normal distributions. In addition, division of the geometric mean of a set of numerators by the geometric mean of the set of corresponding denominators will result in the geometric mean of the set of corresponding quotients.

- J. For each genus for which one or more SMAVs are available, the Genus Mean Acute Value (GMAV) should be calculated as the geometric mean of the SMAVs available for the genus.
- K. Order the GMAVs from high to low.
- L. Assign ranks, R, to the GMAVs from "1" for the lowest to "N" for the highest. If two or more GMAVs are identical, arbitrarily assign them successive ranks.
- M. Calculate the cumulative probability, P, for each GMAV as R/(N+1).
- N. Select the four GMAVs which have cumulative probabilities closest to 0.05 (if there are less than 59 GMAVs, these will always be the four lowest GMAVs).
- O. Using the selected GMAVs and Ps, calculate

$$S^2 = \frac{\sum ((\ln GMAV)^2) - ((\sum \ln GMAV))^2 / 4}{\sum (F) - ((\sum (\sqrt{P}))^2 / 4)}$$

$$L = (\sum (\ln GMAV) - S(\sum (\sqrt{P}))) / 4$$

$$A = S(\sqrt{0.05}) + L$$

$$FAV = e^A$$

(See ¹¹ for development of the calculation procedure and Appendix 2 for an example calculations and computer program.)

NOTE: Natural logarithms (logarithms to base e, denoted as ln) are used herein merely because they are easier to use on some hand calculators and computers than common (base 10) logarithms. Consistent use of either will produce the same result.

P. If for a commercially or recreationally important species the geometric mean of the acute values from ~~the~~ flow-through tests in which the concentrations of test material were measured is lower than the calculated Final Acute Value, then that geometric mean should be used as the Final Acute Value instead of the calculated Final Acute Value.

Q. Go to Section VI.

V. Final Acute Equation

A. When enough data are available to show that acute toxicity to two or more species is similarly related to a water quality characteristic, the relationship should be taken into account as described in Sections B-G below or using analysis of covariance^{15, 16}. The two methods are equivalent and produce identical results. The manual method described below provides an understanding of this application of covariance analysis, but computerized versions of covariance analysis are much more convenient for analyzing large data sets. If two or more factors affect toxicity, multiple regression analysis should be used.

B. For each species for which comparable acute toxicity values are available at two or more different values of the water quality characteristic, perform a least squares regression of the acute toxicity values on the corresponding values of the water quality characteristic to obtain the slope and its 95% confidence limits for each species.

NOTE: Because the best documented relationship is that between hardness and acute toxicity of metals in fresh water and a log-log relationship fits these data, geometric means and natural logarithms of both toxicity and water quality are used in the rest of this section. For relationships based on other water quality characteristics, such as pH, temperature, or salinity, no transformation or a different transformation might fit the data better, and appropriate changes will be necessary throughout this section.

C. Decide whether the data for each species is useful, taking into account the range and number of the tested values of the water quality characteristic and the degree of agreement within and between species. For example, a slope based on six data points might be of limited value if it is based only on data for a very narrow range of values of the water quality characteristic. A slope based on only two data points, however, might be useful if it is consistent with other information and if the two points cover a broad enough range of the water quality characteristic. In addition, acute values that appear to be questionable in comparison with other acute and chronic data available for the same species and for other species in the same genus probably should not be used. For example, if after adjustment for the water quality characteristic, the acute values available for a species or genus differ by more than a factor of 10, rejection of some or all of the values is probably appropriate. If useful slopes are not available for at least one fish and one invertebrate or if the available slopes are too dissimilar or if too few data are available to adequately define the relationship between acute toxicity and the water quality characteristic, return to Section IV.G., using the results of tests

conducted under conditions and in waters similar to those commonly used for toxicity tests with the species.

- D. Individually for each species calculate the geometric mean of the available acute values and then divide each of the acute values for a species by the mean for the species. This normalizes the acute values so that the geometric mean of the normalized values for each species individually and for any combination of species is 1.0.
- E. Similarly normalize the values of the water quality characteristic for each species individually.
- F. Individually for each species perform a least squares regression of the normalized acute toxicity values on the corresponding normalized values of the water quality characteristic. The resulting slopes and 95% confidence limits will be identical to those obtained in Section B above. Now, however, if the data are actually plotted, the line of best fit for each individual species will go through the point 1,1 in the center of the graph.
- G. Treat all the normalized data as if they were all for the same species and perform a least squares regression of all the normalized acute values on the corresponding normalized values of the water quality characteristic to obtain the pooled acute slope, V, and its 95% confidence limits. If all the normalized data are actually plotted, the line of best fit will go through the point 1,1 in the center of the graph.
- H. For each species calculate the geometric mean, W, of the acute toxicity values and the geometric mean, X, of the values of the water quality characteristic. (These were calculated in steps D and E above.)
- I. For each species calculate the logarithm, Y, of the SMAV at a selected value, Z, of the water quality characteristic using the equation:
$$Y = \ln W - V(\ln X - \ln Z).$$
- J. For each species calculate the SMAV at Z using the equation: $SMAV = e^Y$.
NOTE: Alternatively, the SMAVs at Z can be obtained by skipping step H above, using the equations in steps I and J to adjust each acute value individually to Z, and then calculating the geometric mean of the adjusted values for each species individually. This alternative procedure allows an examination of the range of the adjusted acute values for each species.
- K. Obtain the Final Acute Value at Z by using the procedure described in Section IV.J-O.
- L. If the SMAV at Z of a commercially or recreationally important species is lower than the calculated Final Acute Value at Z, then that SMAV should be used as the Final Acute Value at Z instead of the calculated Final Acute Value.
- M. The Final Acute Equation is written as: Final Acute Value = $e^{(V[\ln(\text{water quality characteristic}) + \ln A - V[\ln Z])}$, where V = pooled acute slope and A = Final Acute Value at Z. Because

V, A, and Z are known, the Final Acute Value can be calculated for any selected value of the water quality characteristic.

VI. Final Chronic Value

- A. Depending on the data that are available concerning chronic toxicity to aquatic animals, the Final Chronic Value might be calculated in the same manner as the Final Acute Value or by dividing the Final Acute Value by the Final Acute-Chronic Ratio. In some cases it may not be possible to calculate a Final Chronic Value.

NOTE: As the name implies, the acute-chronic ration (ARC) is a way of relating acute and chronic toxicities. The acute-chronic ratio is basically the inverse of the application factor, but this new name is better because it is more descriptive and should help prevent confusion between "application factors" and "safety factors". Acute-chronic ratios and application factors are ways of relating the acute and chronic toxicities of a material to aquatic organisms. Safety factors are used to provide an extra margin of safety beyond the known or estimated sensitivities of aquatic organisms. Another advantage of the acute-chronic ratio is that it will usually be greater than one; this should avoid the confusion as to whether a large application factor is one that is close to unity or one that has a denominator that is much greater than the numerator.

- B. Chronic values should be based on results of flow-through (except renewal is acceptable for daphnids) chronic tests in which the concentrations of test material in the test solutions were properly measured at appropriate times during the test.
- C. Results of chronic tests in which survival, growth, or reproduction in the control treatment was unacceptably low should not be used. The limits of acceptability will depend on the species.
- D. Results of chronic tests conducted in unusual dilution water, e.g., dilution water in which total organic carbon or particulate matter exceeded 5 mg/L, should not be used, unless a relationship is developed between chronic toxicity and organic carbon or particulate matter or unless data show that organic carbon, particulate matter, etc., do not affect toxicity.
- E. Chronic values should be based on endpoints and lengths of exposure appropriate to the species. Therefore, only results of the following kinds of chronic toxicity tests should be used:
1. Life-cycle toxicity tests consisting of exposures of each of two or more groups of individuals of a species to a different concentration of the test material throughout a life cycle. To ensure that all life stages and life processes are exposed, tests with fish should begin with embryos or newly hatched young less than 48 hours old, continue through maturation and reproduction, and should end not less than 24 days (90 days for salmonids) after the hatching of the next generation. Tests with daphnids should begin with young less than 24 hours old and last for not less than 21 days. Tests with mysids should begin with young less than 24 hours old and continue until 7 days past the median time of first brood release in the

controls. For fish, data should be obtained and analyzed on survival and growth of adults and young, maturation of males and females, eggs spawned per female, embryo viability (salmonids only), and hatchability. For daphnids, data should be obtained and analyzed on survival and young per female. For mysids, data should be obtained and analyzed on survival, growth, and young per female.

2. Partial life-cycle toxicity tests consisting of exposures of each of two or more groups of individuals of a species of fish to a different concentration of the test material through most portions of a life cycle. Partial life-cycle tests are allowed with fish species that require more than a year to reach sexual maturity, so that all major life stages can be exposed to the test material in less than 15 months. Exposure to the test material should begin with immature juveniles at least 2 months prior to active gonad development, continue through maturation and reproduction, and end not less than 24 days (90 days for salmonids) after the hatching of the next generation. Data should be obtained and analyzed on survival and growth of adults and young, maturation of males and females, eggs spawned per female, embryo viability (salmonids only), and hatchability.
3. Early life-stage toxicity tests consisting of 28- to 32-day (60 days post hatch for salmonids) exposures of the early life stages of a species of fish from shortly after fertilization through embryonic, larval, and early juvenile development. Data should be obtained and analyzed on survival and growth.

NOTE: Results of an early life-stage test are used as predictions of results of life-cycle and partial life-cycle tests with the same species. Therefore, when results of a life-cycle or partial life-cycle test are available, results of an early life-stage test with the same species should not be used. Also, results of early life-stage tests in which the incidence of mortalities or abnormalities increased substantially near the end of the test should not be used because results of such tests are possibly not good predictions of the results of comparable life-cycle or partial life-cycle tests.

- F. 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 of the 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 one or more of the specified biological measurements, and (c) above which all tested concentrations also caused such an effect.

NOTE: Because various authors have used a variety of terms and definitions to interpret and report results of chronic tests, reported results should be reviewed carefully. 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%) might be

considered acceptable even if it is statistically significantly different from the control, whereas a large percent reduction (e.g., 30%) might be considered unacceptable even if it is not statistically significant.

- G. If the chronic toxicity of the material to aquatic animals apparently has been shown to be related to a water quality characteristic such as hardness or particulate matter for freshwater animals or salinity or particulate matter for saltwater animals, a Final Chronic Equation should be derived based on that water quality characteristic. Go to Section VII.
- H. If chronic values are available for species in eight families as described in Sections III.B.1 or III.C.1, a Species Mean Chronic Value (SMCV) should be calculated for each species for which at least one chronic value is available by calculating the geometric mean of all chronic values available for the species, and appropriate Genus Mean Chronic Values should be calculated. The Final Chronic Value should then be obtained using the procedure described in Section IV.J-O. Then go to Section VI.M.
- I. For each chronic value for which at least one corresponding appropriate acute value is available, calculate an acute-chronic ratio, using for the numerator the geometric mean of the results of all acceptable flow-through (except static is acceptable for daphnids) acute tests in the same dilution water and in which the concentrations were measured. For fish, the acute test(s) should have been conducted with juveniles. The acute test(s) should have been part of the same study as the chronic test. If acute tests were not conducted as part of the same study, acute tests conducted in the same laboratory and dilution water, but in a different study, may be used. If no such acute tests are available, results of acute tests conducted in the same dilution water in a different laboratory may be used. If no such acute tests are available, an acute-chronic ratio should not be calculated.
- J. For each species, calculate the species mean acute-chronic ratio as the geometric mean of all acute-chronic ratios available for that species.
- K. For some materials the acute-chronic ratio seems to be the same for all species, but for other materials the ratio seems to increase or decrease as the Species Mean Acute Value (SMAV) increases. Thus the Final Acute-Chronic Ratio can be obtained in four ways, depending on the data available:
 - 1. If the species mean acute-chronic ratios seems to increase or decrease as the SMAV increases, the Final Acute-Chronic Ratio should be calculated as the geometric mean of the acute-chronic ratios for species whose SMAVs are close to the Final Acute Value.
 - 2. If no major trend is apparent and the acute-chronic ratios for a number of species are within a factor of ten, the Final Acute-Chronic Ratio should be calculated as the geometric mean of all the species mean acute-chronic ratios available for both freshwater and saltwater species.
 - 3. For acute tests conducted on metals and possibly other substances with embryos and larvae of barnacles, bivalve molluscs, sea urchins, lobsters,

crabs, shrimp, and abalones (see Section IV.E.2), it is probably appropriate to assume that the acute-chronic ratio is 2. Chronic tests are very difficult to conduct with most such species, but it is likely that the sensitivities of embryos and larvae would determine the results of life-cycle tests. Thus, if the lowest available SMAVs were determined with embryos and larvae of such species, the Final Acute-Chronic Ratio should probably be assumed to be 2, so that the Final Chronic Value is equal to the Criterion Maximum Concentration (see Section XI.B).

4. If the most appropriate species mean acute-chronic ratios are less than 2.0, and especially if they are less than 1.0, acclimation has probably occurred during the chronic test. Because continuous exposure and acclimation cannot be assured to provide adequate protection in field situations, the Final Acute-Chronic Ratio should be assumed to be 2, so that the Final Chronic Value is equal to the Criterion Maximum Concentration (see Section XI.B).

If the available species mean acute-chronic ratios do not fit one of these cases, a Final Acute-Chronic Ratio probably cannot be obtained, and a Final Chronic Value probably cannot be calculated.

- L. Calculate the Final Chronic Value by dividing the Final Acute Value by the Final Acute-Chronic Ratio. If there was a Final Acute Equation rather than a Final Acute Value, see also Section VII.A.
- M. If the Species Mean Chronic Value of a commercially or recreationally important species is lower than the calculated Final Chronic Value, then that Species Mean Chronic Value should be used as the Final Chronic Value instead of the calculated Final Chronic Value.
- N. Go to Section VIII.

VII. Final Chronic Equation

- A. A Final Chronic Equation can be derived in two ways. The procedure described here in Section A will result in the chronic slope being the same as the acute slope. The procedure described in Sections B-N will usually result in the chronic slope being different from the actual slope.
 1. If acute-chronic ratios are available for enough species at enough values of the water quality characteristic to indicate that the acute-chronic ratio is probably the same for all species and is probably independent of the water quality characteristic, calculate the Final Acute-Chronic Ratio as the geometric mean of the available species mean acute-chronic ratios.
 2. Calculate the Final Chronic Value at the selected value Z of the water quality characteristic by dividing the Final Acute Value at Z (see Section V.M.) by the Final Acute-Chronic Ratio.

3. Use $V =$ pooled acute slope (see section V.M.) as $L =$ pooled chronic slope.
 4. Go to Section VII.M.
- B. When enough data are available to show that chronic toxicity to at least one species is related to a water quality characteristic, the relationship should be taken into account as described in Sections B-G below or using analysis of covariance^{15, 16}. The two methods are equivalent and produce identical results. The manual method described below provides an understanding of this application of covariance analysis, but computerized versions of covariance analysis are much more convenient for analyzing large data sets. If two more factors affect toxicity, multiple regression analysis should be used.
- C. For each species for which comparable chronic toxicity values are available at two or more different values of the water quality characteristic, perform a least squares regression of the chronic toxicity values on the corresponding values of the water quality characteristic to obtain the slope and its 95% confidence limits for each species.

NOTE: Because the best documented relationship is that between hardness and acute toxicity of metals in fresh water and a log-log relationship fits these data, geometric means and natural logarithms of both toxicity and water quality are used in the rest of this section. For relationships based on other water quality characteristics, such as pH, temperature, or salinity, no transformation or a different transformation might fit the data better, and appropriate changes will be necessary throughout this section. It is probably preferable, but not necessary, to use the same transformation that was used with the acute values in Section V.

- D. Decide whether the data for each species is useful, taking into account the range and number of the tested values of the water quality characteristic and the degree of agreement within and between species. For example, a slope based on six data points might be of limited value if it is based only on data for a very narrow range of values of the water quality characteristic. A slope based on only two data points, however, might be useful if it is consistent with other information and if the two points cover a broad enough range of the water quality characteristic. In addition, chronic values that appear to be questionable in comparison with other acute and chronic data available for the same species and for other species in the same genus probably should not be used. For example, if after adjustment for the water quality characteristic, the chronic values available for a species or genus differ by more than a factor of 10, rejection of some or all of the values is probably appropriate. If a useful chronic slope is not available for at least one species or if the available slopes are too dissimilar or if too few data are available to adequately define the relationship between chronic toxicity and the water quality characteristic, it might be appropriate to assume that the chronic slope is the same as the acute slope, which is equivalent to assuming that the acute-chronic ratio is independent of the water quality characteristic. Alternatively, return to Section VI.H, using the results of tests

conducted under conditions and in waters similar to those commonly used for toxicity tests with the species.

- E. Individually for each species calculate the geometric mean of the available chronic values and then divide each chronic value for a species by the mean for the species. This normalizes the chronic values so that the geometric mean of the normalized values for each species individually and for any combination of species is 1.0.
- F. Similarly normalize the values of the water quality characteristic for each species individually.
- G. Individually for each species perform a least squares regression of the normalized chronic toxicity values on the corresponding normalized values of the water quality characteristic. The resulting slopes and the 95% confidence limits will be identical to those obtained in Section B above. Now, however, if the data are actually plotted, the line of best fit for each individual species will go through the point 1,1 in the center of the graph.
- H. Treat all the normalized data as if they were all for the same species and perform a least squares regression of all the normalized chronic values on the corresponding normalized values of the water quality characteristic to obtain the pooled chronic slope, L, and its 95% confidence limits. If all the normalized data are actually plotted, the line of best fit will go through the point 1,1 in the center of the graph.
- I. For each species calculate the geometric mean, M, of the toxicity values and the geometric mean, P, of the values of the water quality characteristic. (These were calculated in steps E and F above.)
- J. For each species calculate the logarithm, Q, of the Species Mean Chronic Value at a selected value, Z, of the water quality characteristic using the equation: $Q = \ln M - L(\ln P - \ln Z)$.

NOTE: Although it is not necessary, it will usually be best to use the same value of the water quality characteristic here as was used in Section V.I.

- K. For each species calculate a Species Mean Chronic Value at Z using the equation: $SMCV = e^Q$.

NOTE: Alternatively, the Species Mean Chronic Values at Z can be obtained by skipping step J above, using the equations in steps J and K to adjust each acute value individually to Z and then calculating the geometric means of the adjusted values for each species individually. This alternative procedure allows an examination of the range of the adjusted chronic values for each species.

- L. Obtain the Final Chronic Value at Z by using the procedure described in Section IV.J-O.
- M. If the Species Mean Chronic Value at Z of a commercially or recreationally important species is lower than the calculated Final Chronic Value at Z, then that Species Mean

- N. The Final Chronic Equation is written as: Final Chronic Value = $e^{(L[\ln(\text{water quality characteristic}) + \ln S - L (\ln Z)])}$, where L = pooled chronic slope and S = Final Chronic Value at Z. Because L, S and Z are known, the Final Chronic Value can be calculated for any selected value of the water quality characteristic.

VIII. Final Plant Value

- A. Appropriate measures of the toxicity of the material to aquatic plants are used to compare the relative sensitivities of aquatic plants and animals. Although procedures for conducting and interpreting the results of toxicity tests with plants are not well developed, results of tests with plants usually indicate that criteria which adequately protect aquatic animals and their uses will probably also protect aquatic plants and their uses.
- B. A plant value is the result of a 96-hr test conducted with an alga or a chronic test conducted with an aquatic vascular plant.

NOTE: A test of the toxicity of a metal to a plant usually should not be used if the medium contained an excessive amount of a complexing agent, such as EDTA, that might affect the toxicity of the metal. Concentrations of EDTA above about 200 µg/L should probably be considered excessive.

- C. The Final Plant Value should be obtained by selecting the lowest result from a test with an important aquatic plant species in which the concentrations of test material were measured and the endpoint was biologically important.

IX. Final Residue Value

- A. The Final Residue Value is intended to (a) prevent concentrations in commercially or recreationally important aquatic species from affecting marketability because of exceedance of applicable FDA action levels and (b) protect wildlife, including fishes and birds, that consume aquatic organisms from demonstrated unacceptable effects. The Final Residue Value is the lowest of the residue values that are obtained by dividing maximum permissible tissue concentrations by appropriate bioconcentration or bioaccumulation factors. A maximum permissible tissue concentration is either (a) an FDA action level¹² for fish oil or for the edible portion of fish or shellfish, or (b) a maximum acceptable dietary intake based on observations on survival, growth, or reproduction in a chronic wildlife feeding study or a long-term wildlife field study. If no maximum permissible tissue concentration is available, go to Section X because no Final Residue Value can be derived.
- B. Bioconcentration factors (BCFs) and bioaccumulation factors (BAFs) are quotients 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 be measured in a laboratory test. Some uptake during the bioconcentration test might not be directly from water if the food sorbs some of the test material before it is eaten by the test organisms. A BAF is intended to account for the net uptake from both food and water in a real-world situation. A BAF almost has to be measured in a field situation in which predators accumulate the material directly from water and by consuming prey that itself could have accumulated the material from both food and water. The BCF and BAF are probably similar for a material with a low BCF, but the BAF is probably higher than the BCF for materials with high BCFs. Although BCFs are not too difficult to determine, very few BAFs have been measured acceptably because it is necessary to make enough measurements of the concentration of the material in water to show that it was reasonably constant for a long enough period of time over the range of territory inhabited by the organisms. Because so few acceptable BAFs are available, only BCFs will be discussed further. However, if an acceptable BAF is available for a material, it should be used instead of any available BCFs.

- C. If a maximum permissible tissue concentration is available for a substance (e.g., parent material, parent material plus metabolites, etc.), the tissue concentration used in the calculation of the BCF should be for the same substance. Otherwise, the tissue concentration used in the calculation of the BCF should be that of the material and its metabolites which are structurally similar and are not much more soluble in water than the parent material.
- D.
1. A BCF should be used only if the test was flow-through, the BCF was calculated based on measured concentrations of the test material in tissue and in the test solution, and the exposure continued at least until either apparent steady-state or 28 days was reached. Steady-state is reached when the BCF does not change significantly over a period of time, such as two days or 16 percent of the length of the exposure, whichever is longer. The BCF used from a test should be the highest of (a) the apparent steady-state BCF, if apparent steady-state was reached, (b) the highest BCF obtained, if apparent steady-state was not reached, and (c) the projected steady-state BCF, if calculated.
 2. Whenever a BCF is determined for a lipophilic material, the percent lipids should also be determined in the tissue(s) for which the BCF was calculated.
 3. A BCF obtained from an exposure that adversely affected the test organisms may be used only if it is similar to a BCF obtained with unaffected organisms of the same species at lower concentrations that did not cause adverse effects.
 4. Because maximum permissible tissue concentrations are almost never based on dry weights, a BCF calculated using dry tissue weights must be converted to a wet tissue weight basis. If no conversion factor is reported with the BCF, multiply the dry weight BCF by 0.1 for plankton and by 0.2 for individual species of fishes and invertebrates¹⁷.

5. If more than one acceptable BCF is available for a species, the geometric mean of the available values should be used, except that if the BCFs are from different lengths of exposure and the BCF increases with length of exposure, the BCF for the longest exposure should be used.
- E. If enough pertinent data exist, several residue values can be calculated by dividing maximum permissible tissue concentrations by appropriate BCFs:
1. For each available maximum acceptable dietary intake derived from a chronic feeding study or a long-term field study with wildlife, including birds and aquatic organisms, the appropriate BCF is based on the whole body of aquatic species which constitute or represent a major portion of the diet of the tested wildlife species.
 2. For an FDA action level for fish or shellfish, the appropriate BCF is the highest geometric mean species BCF for the edible portion (muscle for decapods, muscle with or without skin for fishes, adductor muscle for scallops, and total soft tissue for other bivalve molluscs) of a consumed species. The highest species BCF is used because FDA action levels are applied on a species-by-species basis.
- F. For lipophilic materials, it might be possible to calculate additional residue values. Because the steady-state BCF for a lipophilic material seems to be proportional to percent lipids from one tissue to another and from one species to another^{18, 19, 20}, extrapolations can be made from tested tissues or species to untested tissues or species on the basis of percent lipids.
1. For each BCF for which the percent lipids is known for the same tissue for which the BCF was measured, normalize the BCF to a one percent lipid basis by dividing the BCF by the percent lipids. This adjustment to a one percent lipid basis is intended to make all the measured BCFs for a material comparable regardless of the species or tissue with which the BCF was measured.
 2. Calculate the geometric mean normalized BCF. Data for both saltwater and freshwater species should be used to determine the mean normalized BCF, unless the data show that the normalized BCFs are probably not similar.
 3. Calculate all possible residue values by dividing the available maximum permissible tissue concentrations by the mean normalized BCF and by the percent lipids values appropriate to the maximum permissible tissue concentrations, i.e.,

$$\text{Residue Value} = \frac{(\text{maximum permissible tissue concentration})}{(\text{mean normalized BCF}) (\text{appropriate percent lipids})}$$

- a. For an FDA action level for fish oil, the appropriate percent lipids value is 100.

- b. For an FDA action level for fish, the appropriate percent lipids value is 11 for freshwater criteria and 10 for saltwater criteria because FDA action levels are applied on a species-by-species basis to commonly consumed species. The highest lipid contents in the edible portions of important consumed species are about 11 percent for both the freshwater chinook salmon and lake trout and about 10 percent for the saltwater Atlantic herring ²¹.
 - c. For a maximum acceptable dietary intake derived from a chronic feeding study or a long-term field study with wildlife, the appropriate percent lipids is that of an aquatic species or group of aquatic species which constitute a major portion of the diet of the wildlife species.
- G. The Final Residue Value is obtained by selecting the lowest of the available residue values.

NOTE: In some cases the Final Residue Value will not be low enough. For example, a residue value calculated from an FDA action level will probably result in an average concentration in the edible portion of a fatty species that is at the action level. Some individual organisms, and possibly some species, will have residue concentrations higher than the mean value but no mechanism has been devised to provide appropriate additional protection. Also, some chronic feeding studies and long-term field studies with wildlife identify concentrations that cause adverse effects but do not identify concentrations which do not cause adverse effects; again no mechanism has been devised to provide appropriate additional protection. These are some of the species and uses that are not protected at all times in all places.

X. Other Data

Pertinent information that could not be used in earlier sections might be available concerning adverse effects on aquatic organisms and their uses. The most important of these are data on cumulative and delayed toxicity, flavor impairment, reduction in survival, growth, or reproduction, or any other adverse effect that has been shown to be biologically important. Especially important are data for species for which no other data are available. Data from behavioral, biochemical, physiological, microcosm, and field studies might also be available. Data might be available from tests conducted in unusual dilution water (see IV.D and VI.D), from chronic tests in which the concentrations were not measured (see VI.B), from tests with previously exposed organisms (see II.F), and from tests on formulated mixtures or emulsifiable concentrates (see II.D). Such data might affect a criterion if the data were obtained with an important species, the test concentrations were measured, and the endpoint was biologically important.

XI. Criterion

- A. A criterion consists of two concentrations: the Criterion Maximum Concentration and the Criterion Continuous Concentration.

- B. The Criterion Maximum Concentration (CMC) is equal to one-half the Final Acute Value.
- C. The Criterion Continuous Concentration (CCC) is equal to the lowest of the Final Chronic Value, the Final Plant Value, and the Final Residue Value, unless other data (see Section X) show that a lower value should be used. If toxicity is related to a water quality characteristic, the CCC is obtained from the Final Chronic Equation, the Final Plant Value, and the Final Residue Value by selecting the one, or the combination, that results in the lowest concentrations in the usual range of the water quality characteristic, unless other data (see Section X) show that a lower value should be used.
- D. Round ¹⁴ both the CMC and the CCC to two significant digits.
- E. The criterion is stated as:

The procedures described in the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" indicate that, except possibly where a locally important species is very sensitive, (1) aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of (2) does not exceed (3) $\mu\text{g/L}$ more than once every three years on the average and if the one-hour average concentration does not exceed (4) $\mu\text{g/L}$ more than once every three years on the average.

where (1) = insert "freshwater" or "saltwater"

(2) = insert name of material

(3) = insert the Criterion Continuous Concentration

(4) = insert the Criterion Maximum Concentration.

XII. Final Review

- A. The derivation of the criterion should be carefully reviewed by rechecking each step of the Guidelines. Items that should be especially checked are:
 - 1. If unpublished data are used, are they well documented?
 - 2. Are all required data available?
 - 3. Is the range of acute values for any species greater than a factor of 10?
 - 4. Is the range of Species Mean Acute Values for any genus greater than a factor of 10?
 - 5. Is there more than a factor of ten difference between the four lowest Genus Mean Acute Values?
 - 6. Are any of the four lowest Genus Mean Acute Values questionable?

7. Is the Final Acute Value reasonable in comparison with the Species Mean Acute Values and Genus Mean Acute Values?
 8. For any commercially or recreationally important species, is the geometric mean of the acute values from flow-through tests in which the concentrations of test material were measured lower than the Final Acute Value?
 9. Are any of the chronic values questionable?
 10. Are chronic values available for acutely sensitive species?
 11. Is the range of acute-chronic ratios greater than a factor of 10?
 12. Is the Final Chronic Value reasonable in comparison with the available acute and chronic data?
 13. Is the measured or predicted chronic value for any commercially or recreationally important species below the Final Chronic Value?
 14. Are any of the other data important?
 15. Do any data look like they might be outliers?
 16. Are there any deviations from the Guidelines? Are they acceptable?
- B. On the basis of all available pertinent laboratory and field information, determine if the criterion is consistent with sound scientific evidence. If it is not, another criterion, either higher or lower, should be derived using appropriate modifications of these Guidelines.

References

- ¹ U.S. EPA. 1983. Water Quality Standards Regulation. Federal Register 48: 51400-51413. November 8.
- ² U.S. EPA. 1983. Water Quality Standards Handbook. Office of Water Regulations and Standards, Washington, DC.
- ³ U.S. EPA. 1985. Technical Support Document for Water Quality-Based Toxics Control. Office of Water, Washington, DC.
- ⁴ Thurston, C.E. 1962. Physical Characteristics and Chemical Composition of Two Subspecies of Lake Trout. J. Fish. Res. Ed. Canada 19:39-44.
- ⁵ Hodson, P.V., et al. 1983. Effect of Fluctuating Lead Exposure on Lead Accumulation by Rainbow Trout (*Salmo gairdneri*). Environ. Toxicol. Chem. 2: 225-238.
- ⁶ For example, see: Ingersoll, C.G. and R.W. Winner. 1982. Effect on *Daphnia pulex* (De Geer) of Daily Pulse Exposures to Copper to Cadmium. Environ. Toxicol. Chem. 1:321-327; Seim, W.K., et al. 1984. Growth and Survival of Developing Steelhead Trout (*Salmo gairdneri*) Continuously or Intermittent Exposed to Copper. Can. J. Fish. Aquat. Sci. 41: 433-438; Buckley, J.T., et al. 1982. Chronic Exposure of Coho Salmon to Sublethal Concentrations of Copper-I. Effect on Growth, on Accumulation and Distribution of Copper, and on Copper Tolerance. Comp. Biochem. Physiol. 72C: 15-19; Brown, V.M., et al. 1969. The Acute Toxicity to Rainbow Trout of Fluctuating Concentrations and Mixtures of Ammonia Phenol and Zinc. J. Fish. Biol. 1:1-9; Thurston, R.V., et al. 1981. Effect of Fluctuating Exposures on The Acute Toxicity of Ammonia to Rainbow Trout (*Salmo gairdneri*) and Cutthroat Trout (*S. clarkii*). Water Res. 15: 911-917.
- ⁷ For example, see: Horning, W.B. and T.W. Neiheisel. 1979. Chronic Effect of Copper on the Bluntnose Minnow, *Pimephales notatus* (Rafinesque). Arch. Environ. Contam. Toxicol. 8:545-552.
- ⁸ For example, see: Chapman, G.A. 1982. Letter to Charles E. Stephan. U.S. EPA, Duluth, Minnesota. December 6; Chapman, G.A. 1975. Toxicity of Copper, Cadmium and Zinc to Pacific Northwest Salmonids. Interim Report. U.S. EPA, Corvallis, Oregon; Sephar, R.L. 1976. Cadmium and Zinc Toxicity to Flagfish, *Jordanella floridae*. J. Fish. Res. Board Can. 33: 1939-1945.
- ⁹ U.S. EPA. 1980. Water Quality Criteria Documents; Availability. Federal Register 45: 79318-79379. November 28.
- ¹⁰ U.S. EPA. 1984. Water Quality Criteria; Request for Comments. Federal Register 49: 4551-4554. February 7.
- ¹¹ Erickson, R.J. and C.E. Stephan. 1985. Calculation of the Final Acute Value for Water Quality Criteria for Aquatic Organisms. National Technical Information Service, Springfield, Virginia. PB88-214994.
- ¹² U.S. Food and Drug Administration. 1981. Compliance Policy Guide. Compliance Guidelines Branch, Washington, DC.
- ¹³ For good examples of acceptable procedures, see:
 - ASTM Standard E 729, Practice for Conducting Acute Toxicity Tests with Fishes, Macroinvertebrates, and Amphibians.
 - ASTM Standard E 724, Practice for Conducting Static Acute Toxicity Tests with Larvae of Four Species of Bivalve Molluscs.
- ¹⁴ Huth, E.J., et al. 1978. Council of Biology Editors Style Manual, 4th Ed. Council of Biology Editors, Inc., Bethesda, Maryland. p. 117.
- ¹⁵ Dixon, W.J. and M.B. Brown (eds.). 1979. BMDP Biomedical Computer Programs, P-series. University of California, Berkeley. pp. 521-539.
- ¹⁶ Neter, J. and W. Wasserman. 1974. Applied Linear Statistical Models. Irwin, Inc., Homewood Illinois.

¹⁷ The values of 0.1 and 0.2 were derived from data published in:

McDiffett, W.F. 1970. *Ecology* 51: 975-988.

Brocksen, R.W., et al. 1968. *J. Wildlife Management* 32: 52-75.

Cummins, K.W., et al. 1973. *Ecology* 54: 336-345.

Pesticide Analytical Manual, Volume I, Food and Drug Administration, 1969.

Love, R.M. 1957. In: M.E. Brown (ed.), *The Physiology of Fishes*, Vol. I. Academic Press, New York. p.411.

Ruttner, F. 1963. *Fundamentals of Limnology*, 3rd Ed. Trans. by D.G. Frey and F.E. J. Fry. University of Toronto Press, Toronto.

Some additional values can be found in:

Sculthorpe, C.D. 1967. *The Biology of Aquatic Vascular Plants*. Arnold Publishing, Ltd., London.

¹⁸ Hamelink, J.L., et al. 1971. A Proposal: Exchange Equilibria Control the Degree Chlorinated Hydrocarbons are Biologically Magnified in Lentic Environments. *Trans. Am. Fish. Soc.* 100:207-214.

¹⁹ Lunsford, C.A. and C.R. Blem. 1982. Annual Cycle of Kepone Residue in Lipid Content of the Estuarine Clam, *Rangia cuneata*. *Estuaries* 5: 121-130.

²⁰ Schnoor, J.L. 1982. Field Validation of Water Quality Criteria for Hydrophobic Pollutants. In: J.G. Pearson, et al. (eds.), *Aquatic Toxicology and Hazard Assessment*. ASTM STP 766. American Society for Testing and Materials, Philadelphia. Pp. 302-315.

²¹ Sidwell, V.D. 1981. *Chemical and Nutritional Composition of Finfishes, Whales, Crustaceans, Mollusks, and Their Products*. NOAA Technical Memorandum NMFS F/SEC-11. National Marine Fisheries Service, Southeast Fisheries Center, Charleston, South Carolina.

Appendix 1. Resident North American Species of Aquatic Animals Used in Toxicity and Bioconcentration Tests

Introduction

These lists identify species of aquatic animals which have reproducing wild populations in North America and have been used in toxicity or bioconcentration tests. "North America" includes only the 48 contiguous states, Canada, and Alaska; Hawaii and Puerto Rico are not included. Saltwater (i.e., estuarine and true marine) species are considered resident in North America if they inhabit or regularly enter shore waters on or above the continental shelf to a depth of 200 meters. Species do not have to be native to be resident. Unlisted species should be considered resident North American species if they can be similarly confirmed or if the test organisms were obtained from a wild population in North America.

The sequence for fishes is taken from A List of Common and Scientific Names of Fishes from the United States and Canada. For other species, the sequence of phyla, classes, and families is taken from the NODC Taxonomic Code, Third Edition, National Oceanographic Data Center, NOAA, Washington, DC 20235, July, 1981, and the numbers given are from that source to facilitate verification. Within a family, genera are in alphabetical order, as are species in a genus.

The references given are those used to confirm that the species is a resident North American species. (The NODC Taxonomic Code contains foreign as well as North American species.) If no such reference could be found, the species was judged to be nonresident. No reference is given for organisms not identified to species; these are considered resident only if obtained from wild North American populations. A few nonresident species are listed in brackets and noted as "nonresident" because they were mistakenly identified as resident in the past or to save other investigators from doing literature searches on the same species.

Special Note

This December 2010 electronic version of the 1985 Guidelines serves to meet the requirements of Section 508 of the Rehabilitation Act. While converting the 1985 Guidelines to a 508-compliant version, EPA updated the taxonomic nomenclature to reflect changes that occurred since the tables were originally produced in 1985. The numbers included for Phylum, Class and Family represent those currently in use from the Integrated Taxonomic Information System, or ITIS, and reflect what is referred to in ITIS as Taxonomic Serial Numbers. ITIS replaced the National Oceanographic Data Center (NODC) taxonomic coding system which was used to create the original taxonomic tables included in the 1985 Guidelines document (NODC, Third Addition - see Introduction). For more information on the NODC taxonomic codes, see <http://www.nodc.noaa.gov/General/CDR-detdesc/taxonomic-v8.html>.

The code numbers included in the reference column of the tables have not been updated from the 1985 version. These code numbers are associated with the old NODC taxonomic referencing system and are simply replicated here for historical purposes. Footnotes may or may not still apply.

EPA is working on a more comprehensive update to the 1985 Guidelines, including new taxonomic tables which better reflect the large number of aquatic animal species known to be propagating in U.S. waters.

Freshwater Species Table

Synonyms appear after the official Scientific Name and are marked with an asterisk (*).
Non-resident species are noted in the Reference column and are marked with a dagger (†)

Class	Family	Species		Reference
		Common Name	Scientific Name	
Phylum: Porifera (46861)				
Demospongiae 47528	Spongillidae 47691	Sponge	<i>Ephydatia fluviatilis</i>	P93
Phylum: Cnidaria (48738)				
Hydrozoa 48739	Hydridae 50844	Hydra	<i>Hydra oligactis</i>	E318, P112
		Hydra	<i>Hydra littoralis</i>	E321, P112
Phylum: Platyhelminthes (53963)				
Turbellaria 53964	Planariidae 54502	Planarian	<i>Dugesia dorotocephala</i>	D22
		Planarian	<i>Dugesia lugubris</i> <i>Dugesia polychroa</i> *	D24
		Planarian	<i>Planaria gonocephala</i>	
		Planarian	<i>Polycelis felina</i> [§]	nonresident
	Dendrocoelidae 54469	Planarian	<i>Procotyla fluviatilis</i> <i>Dendrocoelum lacteum</i> *	E334, P132, D63
Phylum: Gastrotricha (57597)				
Chaetonotida 57822	Chaetonotidae 57823	Gastrotrich	<i>Lepidodermella squamata</i> <i>Lepidodermella squamatum</i> *	E413
Phylum: Rotifera (58239)				
Eurotatoria (Formerly Bdelloidea) 654070	Philodinidae 58266	Rotifer	<i>Philodina acuticornis</i>	Y
		Rotifer	<i>Philodina roseola</i>	E487
Eurotatoria (Formerly Monogononta) 654070	Brachionidae 58344	Rotifer	<i>Keratella cochlearis</i>	E442, P188
		Rotifer	<i>Keratella sp.</i>	²
Phylum: Annelida (64357)				
Polychaeta (Formerly Archiannelida) 64358	Aeolosomatidae 68423	Worm	<i>Aeolosoma headleyi</i>	E528, P284
Clitellata (Formerly Oligochaeta) 568832	Lumbriculidae 68440 Tubificidae 68585	Worm	<i>Lumbriculus variegatus</i>	E533, P290
		Tubificid worm	<i>Branchiura sowerbyi</i>	E534, P289, GG
		Tubificid worm	<i>Limnodrilus hoffmeisteri</i>	E536, GG
		Tubificid worm	<i>Quistadrilus multisetosus</i> <i>Pelosclex multisetosus</i> *	E535, GG
		Tubificid worm	<i>Rhyacodrilus montanus</i>	GG
		Tubificid worm	<i>Spirosperma ferox</i> <i>Pelosclex ferox</i> *	GG
		Tubificid worm	<i>Spirosperma nikolskyi</i> <i>Pelosclex variegatus</i>	E534, GG
		Tubificid worm	<i>Stylodrilus heringianus</i>	GG

* Synonym

§ Non-resident species

Class	Family	Species		Reference	
		Common Name	Scientific Name		
		Tubificid worm	<i>Tubifex tubifex</i>		E536, P289, GG
		Tubificid worm	<i>Varichaeta pacifica</i>	GG	
	Naididae 68854	Worm	<i>Nais sp.</i>	2	
		Worm	<i>Paranais sp.</i>	2	
		Worm	<i>Pristina sp.</i>	2	
Clitellata (Formerly Hirudinea) 568832	Erpobdellidea 69438	Leech	<i>Erpobdella octoculata</i>	Formerly nonresident (BB16)	
Phylum: Mollusca (69458)					
Gastropoda 69459	Viviparidae 70304	Snail	<i>Campeloma decisum</i>	P731, M216	
	Bithyniidae (Amnicolidae) (Bulimidae) (Hydrobiidae) 70745	Snail	<i>Amnicola sp.</i>	2	
	Pleuroceridae 71541	Snail	<i>Goniobasis livescens</i>	P732	
		Snail	<i>Elimia virginica</i> <i>Goniobasis virginica</i>	E1137	
		Snail	<i>Leptoxis carinata</i> <i>Nitocris carinata</i> <i>Mudalia carinata</i>	X, E1137	
		Snail	<i>Nitocris sp.</i>	2	
	Lymnaeidae 76483	Snail	<i>Lymnaea acuminata</i> ^f	nonresident	
		Snail	<i>Lymnaea catascopium</i> <i>Lymnaea emerginata</i> <i>Stagnicola emerginata</i>	M328	
		Snail	<i>Lymnaea elodes</i> <i>Lymnaea palustris</i>	E1127, M351	
		Snail	<i>Lymnaea luteola</i> ^f	nonresident M266	
		Snail	<i>Lymnaea stagnalis</i>	E1127, P728, M296	
		Snail	<i>Lymnaea sp.</i>	2	
	Planorbidae 76591	Snail	<i>Biomphalaria glabrata</i>	Formerly nonresident (M390)	
		Snail	<i>Gyraulus circumstriatus</i>	P729, M397	
		Snail	<i>Helisoma campanulatum</i>	M445	
		Snail	<i>Helisoma trivolvis</i>	P729, M452	
	Physidae 76676	Snail	<i>Aplexa hypnorum</i>	E1126, P727, M373	
		Snail	<i>Physa fontinalis</i> ^f	nonresident M373	
		Snail	<i>Physa gyrina</i>	E1126, P727, M373	
		Snail	<i>Physa heterostropha</i>	M378	
		Snail	<i>Physa integra</i>	P727	
		Snail	<i>Physa sp.</i>	2	
	Bivalvia (Pelecypoda) 79118	Margaritiferidae 79914	Mussel	<i>Margaritifera margaritifera</i>	E1138, P748, J11
		Unionidae (Formerly Amblemidae) 79913	Mussel	<i>Amblema plicata</i>	AA122
		Unionidae	Mussel	<i>Anodonta imbecillis</i>	J72, AA122

Class	Family	Species		Reference
		Common Name	Scientific Name	
	79913	Mussel	<i>Carunculina parva</i> <i>Toxolasma texasensis</i>	J19, AA122
		Mussel	<i>Cyrtoneias tampicoensis</i>	P759, AA122
		Mussel	<i>Elliptio complanata</i>	J13
	Corbiculidae 81381	Asiatic clam	<i>Corbicula fluminea</i>	E1159
		Asiatic clam	<i>Corbicula manilensis</i>	P749
	Pisidiidae Sphaeriidae 81388	Fingernail clam	<i>Eupera cubensis</i> <i>Eupera singleyi</i>	E1158, P763, G9
		Fingernail clam	<i>Musculium transversum</i> <i>Sphaerium transversum</i>	M160, G11
		Fingernail clam	<i>Sphaerium comeum</i>	G12
	Phylum: Arthropoda (82696)			
Branchiopoda (Formerly Crustacea) 83687	Lynceidae 83769	Conchostracan	<i>Lynceus brachyurus</i>	E580, P344
	Sididae 83834	Cladoceran	<i>Diaphanosoma sp.</i>	2
	Daphniidae 83872	Cladoceran	<i>Ceriodaphnia acanthina</i>	E618
		Cladoceran	<i>Ceriodaphnia reticulata</i>	E618, P368
		Cladoceran	<i>Daphnia ambigua</i>	E607, P369
		Cladoceran	<i>Daphnia carinata</i>	3
		Cladoceran	<i>Daphnia cucullata</i> [†]	nonresident
		Cladoceran	<i>Daphnia galeata mendotae</i>	E610, P370
		Cladoceran	<i>Daphnia hyalina</i>	4
		Cladoceran	<i>Daphnia longispina</i>	5
		Cladoceran	<i>Daphnia magna</i>	E605, P367
		Cladoceran	<i>Daphnia parvula</i>	E611
		Cladoceran	<i>Daphnia pulex</i>	E613, P367
		Cladoceran	<i>Daphnia pulicaria</i>	A
		Cladoceran	<i>Daphnia similis</i>	E606, P367
		Cladoceran	<i>Simocephalus serrulatus</i>	E617, P370
	Cladoceran	<i>Simocephalus vetulus</i>	E617, P370	
	Moinidae (Formerly Daphniidae) 84162	Cladoceran	<i>Moina macrocopa</i>	E622, P372
		Cladoceran	<i>Moina rectirostris</i>	E623
	Bosminidae 83935	Cladoceran	<i>Bosmina longirostris</i>	E624, P373
Polyphemidae 83959	Cladoceran	<i>Polyphemus pediculus</i>	E599, P385	
Ostracoda (Formerly Crustacea) 84195	Cyprididae Cypridae 84462	Ostracod	<i>Cyprretta kawatai</i> [†]	nonresident U
		Ostracod	<i>Cypridopsis vidua</i>	E770, P430
Maxillopoda (Formerly Crustacea) 621145	Diaptomidae 85779	Copepod	<i>Eudiaptomus padanus</i> [†]	nonresident
	Temoridae 85855	Copepod	<i>Epischura lacustris</i>	E751, P407
	Cyclopidae 88634	Copepod	<i>Cyclops abyssorum</i> [†]	nonresident
		Copepod	<i>Cyclops bicuspidatus</i>	E807, P405
		Copepod	<i>Cyclops vernalis</i>	E804, P405
Copepod	<i>Cyclops viridis</i> <i>Acanthocyclops viridis</i>	E803, P397		

Class	Family	Species		Reference	
		Common Name	Scientific Name		
Malacostraca (Formerly Crustacea) 89787		Copepod	<i>Acanthocyclops sp.</i>	2	
		Copepod	<i>Diacyclops sp.</i>	2	
		Copepod	<i>Eucyclops agilis</i>	P403	
		Copepod	<i>Mesocyclops leuckarti</i>	E812, P403	
	Asellidae 92657	Isopod	<i>Asellus aquaticus</i> [†]	nonresident (I2)	
		Isopod	<i>Caecidotea bicrenata</i> (Formerly <i>Asellus bicrenata</i>)	HH (I1,2)	
		Isopod	<i>Asellus brevicaudus</i>	E875, P447, I	
		Isopod	<i>Asellus communis</i>	E875, P448, I	
		Isopod	<i>Asellus intermedius</i>	E875, P448, I	
		Isopod	<i>Asellus meridionalis</i> [†] <i>Asellus meridianus</i> [†]	nonresident	
		Isopod	<i>Asellus racovitzai</i>	P449, I	
		Isopod	<i>Lirceus alabamae</i>	P875, I	
		Crangonyctidae (Formerly Gammaridae) 95080	Amphipod	<i>Crangonyx pseudogracilis</i>	P459, T68, FF23
	Gammaridae 93745	Amphipod	<i>Gammarus fasciatus</i>	E877, P458, T53	
		Amphipod	<i>Gammarus lacustris</i>	E877, P458, FF23	
		Amphipod	<i>Gammarus pseudolimnaeus</i>	E877, P458, T48	
		Amphipod	<i>Gammarus pulex</i> [†]	nonresident	
		Amphipod	<i>Gammarus tigrinus</i>	L51, FF17	
		Amphipod	<i>Gammarus sp.</i>	2	
	Hyalellidae (Talitridae) 94022	Amphipod	<i>Hyalella azteca</i> <i>Hyalella knickerbockeri</i>	E876, P457, T154	
Palaemonidae 96213	Prawn	<i>Macrobrachium lamarrei</i> [†]	nonresident		
	Prawn	<i>Macrobrachium rosenbergii</i>	6		
	Prawn	<i>Palaemonetes kadiakensis</i>	E881, P484		
Cambaridae (Formerly Astacidae) 97336	Crayfish	<i>Cambarus latimanus</i>	E897		
	Crayfish	<i>Faxonella clypeata</i>	E890		
	Crayfish	<i>Orconectes immunis</i>	E894, P482		
	Crayfish	<i>Orconectes limosus</i>	E893, P482		
	Crayfish	<i>Orconectes propinquus</i>	E894, P482		
	Crayfish	<i>Orconectes nais</i>	E894		
	Crayfish	<i>Orconectes rusticus</i>	E893, P482		
	Crayfish	<i>Orconectes virilis</i>	E894, P483		
	Crayfish	<i>Pacifastacus trowbridgii</i>	E883		
	Crayfish	<i>Procambarus acutus</i>	P482		
	Crayfish	<i>Procambarus clarki</i> <i>Procambarus clarkii</i>	E885, P482		
	Crayfish	<i>Procambarus simulans</i>	E888, P482		
	Crayfish	<i>Procambarus sp.</i>	2		
	Insecta 99208	Heptageniidae 100504	Mayfly	<i>Maccaffertium ithaca</i> <i>Stenonema ithaca</i>	S173, O205
			Mayfly	<i>Maccaffertium modestum</i> <i>Stenonema rubrum</i>	S178, O205
Baetidea		Mayfly	<i>Callibaetis skokianus</i>	S116, N9	

Class	Family	Species		Reference
		Common Name	Scientific Name	
	100755	Mayfly	<i>Callibaetis sp.</i>	2
		Mayfly	<i>Cloeon dipterum</i>	O173
	Leptophlebiidae 101095	Mayfly	<i>Paraleptophlebia praepedita</i>	S89, O233
	Ephemerellidae 101232	Mayfly	<i>Drunella doddsii</i> <i>Ephemerella doddsi</i>	O245
		Mayfly	<i>Drunella grandis</i> <i>Ephemerella grandis</i>	O245
		Mayfly	<i>Ephemerella subvaria</i>	N9, O248, S71
		Mayfly	<i>Ephemerella sp.</i>	2
	Caenidea 101467	Mayfly	<i>Caenis diminuta</i>	S51, O268
	Ephemeridae 101525	Mayfly	<i>Ephemera simulans</i>	S36, N9, O283
		Mayfly	<i>Hexagenia bilineata</i>	N9, S39, O290
		Mayfly	<i>Hexagenia rigida</i>	O290, S41, N9
		Mayfly	<i>Hexagenia sp.</i>	2
	Libellulidae 101797	Dragonfly	<i>Pantala hymenaea</i> <i>Pantala hymenea</i>	N15, V603
	Coenagrionidae Agrionidae Coenagriidae 102077	Damselfly	<i>Enallagma aspersum</i>	DD
		Damselfly	<i>Ischnura elegans</i> ¹	nonresident
		Damselfly	<i>Ischnura verticalis</i>	N15, E918
		Damselfly	<i>Ischnura sp.</i>	2
	Pteronarcyidae (Formerly Pteronarcidae) Pteronarcyidae 102470	Stonefly	<i>Pteronarcella badia</i>	L172
		Stonefly	<i>Pteronarcys californica</i>	L173
		Stonefly	<i>Pteronarcys dorsata</i>	E947
		Stonefly	<i>Pteronarcys sp.</i>	2
	Nemouridae 102517	Stonefly	<i>Nemoura cinerea</i> ¹	nonresident
	Perlidae 102914	Stonefly	<i>Acroneuria lycorias</i>	N4, E953
		Stonefly	<i>Acroneuria pacifica</i>	E953, L180
		Stonefly	<i>Claassenia sabulosa</i>	E953
		Stonefly	<i>Agnatina capitata</i> <i>Neophasganophora capitata</i> <i>Phasganophora capitata</i>	E953, CC407
	Periodidae 102994	Stonefly	<i>Skwala americana</i> <i>Arcynopteryx parallela</i>	E954
	Nepidae 103747	Water Scorpion	<i>Ranatra elongate</i> ¹ (Species cannot be confirmed in ITIS)	nonresident
	Dytiscidae 111963	Beetle	-	2
	Elmidae Elminthidae 114093	Beetle	<i>Stenelmis sexlineata</i>	W21
	Hydropsychidae 115398	Caddisfly	<i>Arctopsyche grandis</i>	L251, I198
		Caddisfly	<i>Hydropsyche betteni</i>	N24
		Caddisfly	<i>Hydropsyche californica</i>	L253
		Caddisfly	<i>Hydropsyche sp.</i>	2
	Limnephilidae 115933	Caddisfly	<i>Clistronia magnifica</i>	I1206
		Caddisfly	<i>Philartus quaeris</i>	I1272

Class	Family	Species		Reference
		Common Name	Scientific Name	
	Brachycentridae 116905	Caddisfly	<i>Brachycentrus sp.</i>	2
	Tipulidae 118840	Crane fly	<i>Tipula sp.</i>	2
	Ceratopogonidae 127076	Biting midge	-	2
	Culicidae 125930	Mosquito	<i>Aedes aegypti</i>	EE3
		Mosquito	<i>Culex pipiens</i>	EE3
	Chironomidae 127917	Midge	<i>Chironomus plumosus</i> <i>Tendipas plumosus</i>	L423
		Midge	<i>Chironomus tentans</i>	Q
		Midge	<i>Chironomus thummi</i> ^f	nonresident
		Midge	<i>Chironomus sp.</i>	2
		Midge	<i>Paratanytarsus parthenogeneticus</i>	1
		Midge	<i>Paratanytarsus dissimilis</i> <i>Tanytarsus dissimilis</i>	R11
	Athericidae (Formerly Rhagionidae) Leptidae 130928	Snipe fly	<i>Atherix sp.</i>	2
Phylum: Ectoprocta (155470)				
Phylactolaemata 156688	Pectinatellidae (Formerly Pectinatelcidae) 156729	Bryozoan	<i>Pectinatella magnifica</i>	E502, P269
	Lophopodidae 156714	Bryozoan	<i>Lophopodella carteri</i>	E502, P2671
	Plumatellidae 156690	Bryozoan	<i>Plumatella emarginata</i>	E505, P272
Phylum: Chordata (158852)				
Agnatha 159693	Petromyzontidae 159697	Sea lamprey	<i>Petromyzon marinus</i>	F11
Actinopterygii (Formerly Osteichthyes) 161061	Anguillidae 161125	American eel	<i>Anguilla rostrata</i>	F15
	Salmonidae 161931	Pink salmon	<i>Oncorhynchus gorbuscha</i>	F18
		Coho salmon	<i>Oncorhynchus kisutch</i>	F18
		Sockeye salmon	<i>Oncorhynchus nerka</i>	F19
		Chinook Salmon	<i>Oncorhynchus tshawytscha</i>	F19
		Mountain whitefish	<i>Prosopium williamsoni</i>	F19
		Golden Trout	<i>Oncorhynchus aguabonita</i> (Formerly <i>Salmo aguabonita</i>)	F19
		Cutthroat trout	<i>Oncorhynchus clarki</i> (Formerly <i>Salmo clarki</i>)	F19
		Rainbow trout Steelhead trout	<i>Oncorhynchus mykiss</i> (Formerly <i>Salmo gairdneri</i>)	F19
		Atlantic salmon	<i>Salmo salar</i>	F19
		Brown trout	<i>Salmo trutta</i>	F19
		Brook trout	<i>Salvelinus fontinalis</i>	F19
		Lake trout	<i>Salvelinus namaycush</i>	F19

Class	Family	Species		Reference
		Common Name	Scientific Name	
	Esocidae 162137	Northern pike	<i>Esox lucius</i>	F20
	Cyprinidae 163342	Chiselmouth	<i>Acrocheilus alutaceus</i>	F21
		Longfin dace	<i>Agosia chrysogaster</i>	F21
		Central stoneroller	<i>Campostoma anomalum</i>	F21
		Goldfish	<i>Carassius auratus</i>	F21
		Common carp	<i>Cyprinus carpio</i>	F21
		Zebra danio Zebrafish	<i>Danio rerio</i> [†] <i>Brachydanio rerio</i> [†]	nonresident F96
		Silverjaw minnow	<i>Notropis buccatus</i> <i>Ericymba buccata</i>	F21
		Golden shiner	<i>Notemigonus crysoleucas</i>	F23
		Pugnose shiner	<i>Notropis anogenus</i>	F23
		Emerald shiner	<i>Notropis atherinoides</i>	F23
		Striped shiner	<i>Luxilus chrysocephalus</i> <i>Notropis chrysocephalus</i>	F23
		Common shiner	<i>Luxilus comutus</i> <i>Notropis comutus</i>	F23
		Pugnose minnow	<i>Opsopoeodus emiliae</i> <i>Notropis emiliae</i>	F24
		Spottail shiner	<i>Notropis hudsonius</i>	F24
		Red shiner	<i>Cyprinella lutrensis</i> <i>Notropis lutrensis</i>	F24
		Spotfin shiner	<i>Cyprinella spiloptera</i> <i>Notropis spilopterus</i>	F25
		Sand shiner	<i>Notropis stramineus</i>	F25
		Steelcolor shiner	<i>Cyprinella whipplei</i> <i>Notropis whipplei</i>	F25
		Northern redbelly dace	<i>Phoxinus eos</i>	F25
		Bluntnose minnow	<i>Pimephales notatus</i>	F25
		Fathead minnow	<i>Pimephales promelas</i>	F25
		Northern squawfish	<i>Ptychocheilus oregonensis</i>	F25
		Blacknose dace	<i>Rhinichthys atratulus</i>	F25
		Speckled dace	<i>Rhinichthys osculus</i>	F25
		Bitterling	<i>Rhodeus sericeus</i>	F26
		Rudd	<i>Scardinius erythrophthalmus</i>	F26
		Creek chub	<i>Semotilus atromaculatus</i>	F26
		Pearl dace	<i>Margariscus margarita</i> <i>Semotilus margarita</i>	F26
		Tench	<i>Tinca tinca</i>	F26
		Catostomidae 163892	White sucker	<i>Catostomus commersoni</i>
	Mountain sucker		<i>Catostomus platyrhynchus</i>	F26
	Ictaluridae 163995	Black bullhead	<i>Ameiurus melas</i> <i>Ictalurus melas</i>	F27
		Yellow bullhead	<i>Ameiurus natalis</i> <i>Ictalurus natalis</i>	F27
		Brown bullhead	<i>Ameiurus nebulosus</i> <i>Ictalurus nebulosus</i>	F27
		Channel catfish	<i>Ictalurus punctatus</i>	F27

Class	Family	Species		Reference
		Common Name	Scientific Name	
	Clariidae 164118	Walking catfish	<i>Clarias batrachus</i>	F28
	Adrianichthyidae (Formerly Oryziidae) 165623	Medaka	<i>Oryzias latipes</i>	nonresident F96
	Cyprinodontidae 165629	Banded killifish	<i>Fundulus diaphanus</i>	F33
		Flagfish	<i>Jordanella floridae</i>	F33
	Poeciliidae 165876	Mosquitofish	<i>Gambusia affinis</i>	F33
		Amazon molly	<i>Poecilia formosa</i>	F34
		Sailfin molly	<i>Poecilia latipinna</i>	F34
		Molly	<i>Poecilia sp.</i>	
		Guppy	<i>Poecilia reticulata</i> (<i>Lebistes reticulatus</i> , Obs.)	F34
		Southern platyfish	<i>Xiphophorus maculatus</i>	F34
	Gasterosteidae 166363	Brook stickleback	<i>Culaea inconstans</i>	F35
		Threespine stickleback	<i>Gasterosteus aculeatus</i>	F35
		Ninespine stickleback	<i>Pungitius pungitius</i>	F35
	Percichthyidae 170315	White perch	<i>Morone americana</i> (<i>Roccus americanus</i> , Obs.)	F36
		Striped bass	<i>Morone saxatilis</i> (<i>Roccus saxatilis</i> , Obs.)	F36
	Centrarchidae 168093	Rock bass	<i>Ambloplites rupestris</i>	F38
		Green sunfish	<i>Lepomis cyanellus</i>	F38
		Pumpkinseed	<i>Lepomis gibbosus</i>	F38
		Orangespotted sunfish	<i>Lepomis humilis</i>	F38
		Bluegill	<i>Lepomis macrochirus</i>	F38
		Longear sunfish	<i>Lepomis megalotis</i>	F38
		Redear sunfish	<i>Lepomis microlophus</i>	F38
		Smallmouth bass	<i>Micropterus dolomieu</i>	F39
		Largemouth bass	<i>Micropterus salmoides</i>	F39
		White crappie	<i>Pomoxis annularis</i>	F39
		Black crappie	<i>Pomoxis nigromaculatus</i>	F39
		Percidae 168356	Rainbow darter	<i>Etheostoma caeruleum</i>
	Johnny darter		<i>Etheostoma nigrum</i>	F40
	Orangethroat darter		<i>Etheostoma spectabile</i>	F40
	Yellow perch		<i>Perca flavescens</i>	F41
	Walleye		<i>Sander vitreus</i> <i>Stizostedion vitreum vitreum</i>	F41
	Sciaenidae 169237	Freshwater drum	<i>Aplodinotus grunniens</i>	F45
	Cichlidae 169770	Oscar	<i>Astronotus ocellatus</i>	F47
		Blue tilapia	<i>Tilapia aurea</i>	F47
		Mozambique tilapia	<i>Oreochromis mossambicus</i> <i>Tilapia mossambica</i>	F47
	Cottidae 167196	Mottled sculpin	<i>Cottus bairdi</i>	F60
Amphibia 173420	Ranidae 173433	Bullfrog	<i>Rana catesbeiana</i>	B206
		Green frog	<i>Rana clamitans</i>	B206

Class	Family	Species		Reference
		Common Name	Scientific Name	
		Pig frog	<i>Lithobates grylio</i> <i>Rana grylio</i>	B206
		River frog	<i>Rana heckscheri</i>	B206
		Leopard frog	<i>Rana pipiens</i>	B205
		Wood frog	<i>Rana sylvatica</i>	B206
		Frog	<i>Rana temporaria</i> ¹	nonresident
		Leopard frog	<i>Lithobates sphenoccephalus</i> <i>sphenoccephalus</i> (Formerly <i>Rana spenocephala</i>)	JJ
	Microhylidae 173465	Eastern narrow-mouthed toad	<i>Gastrophryne carolinensis</i>	B192
	Bufonidae 173471	American toad	<i>Anaxyrus americanus americanus</i> <i>Bufo americanus</i> ²	B196
		Toad	<i>Bufo bufo</i> ¹	nonresident
		Green toad	<i>Anaxyrus debilis debilis</i> <i>Bufo debilis</i>	B197
		Fowler's toad	<i>Anaxyrus fowleri</i> <i>Bufo fowleri</i>	B196
		Red-spotted toad	<i>Anaxyrus punctatus</i> <i>Bufo punctatus</i> ²	B198
		Woodhouse's toad	<i>Anaxyrus woodhousii woodhousii</i> <i>Bufo woodhousii</i>	B196
	Hylidae 173497	Northern cricket frog	<i>Acris crepitans</i>	B203
		Southern gray treefrog	<i>Hyla chrysoscelis</i>	B201
		Spring creeper	<i>Pseudacris crucifer</i> <i>Hyla crucifer</i>	B202
		Barking treefrog	<i>Hyla gratiosa</i>	B201
		Squirrel treefrog	<i>Hyla squirella</i>	B201
		Gray treefrog	<i>Hyla versicolor</i>	B200
		Northern chorus frog	<i>Pseudacris triseriata</i>	B202
	Pipidae 173547	African clawed frog	<i>Xenopus laevis</i>	Z16
	Ambystomatidae 173588	Spotted salamander	<i>Ambystoma maculatum</i>	B176
		Mexican axolotl	<i>Ambystoma mexicanum</i> ¹	nonresident
		Marbled salamander	<i>Ambystoma opacum</i>	B176
	Salamandridae 173613	Newt	<i>Notophthalmus viridescens</i> <i>Triturus viridescens</i>	B179

Footnotes for Freshwater Species

- ¹ Apparently this is an outdated name (D19, 20). Organisms identified as such should only be used if they were obtained from North America.
- ² Apparently this is an outdated name (D19, 20). Organisms identified as such should only be used if they were obtained from North America.
- ³ If from North America, it is resident and should be called *D. similis* (C). If not from North America, it should be considered nonresident.
- ⁴ If from North America, it is resident and may be any one of a number of species such as *D. laevis*, *D. dubia*, or *D. galeate mendoca* (C). If not from North America, it should be considered nonresident.

-
- ⁵ If from North America, it is resident and may be any one of a number of species such as *D. ambigua*, *D. longiremis*, or *D. rosea* (C). If not from North America, it should be considered nonresident.
- ⁶ This species might be established in portions of the southern United States.
- ⁷ The taxonomy of this species and this and similar genera has not been clarified, but this species should be considered resident.

References for Freshwater Species

- A) Brandlova, J., Z. Brandl, and C.H. Fernando. 1972. The Cladocera of Ontario with remarks on some species and distribution. *Can. J. Zool.* 50: 1373-1403.
- B) W. F., et al. 1968. *Vertebrates of the United States*. 2nd Ed. McGraw-Hill, New York.
- C) Brooks, J.L. 1957. *The Systematics of North American Daphnia*. *Memoirs of the Connecticut Academy of Arts and Sciences*, Vol. XIII.
- D) Kenk, R. 1972. *Freshwater Planarians (Turbellaria) of North America*. *Biota of Freshwater Ecosystems Identification Manual No. 1*. U.S. G.P.O #5501-0365.
- E) Edmondson, W.T. (ed.) 1965. *Fresh-water Biology*. 2nd Ed. Wiley, New York.
- F) Committee on Names of Fishes. 1980. *A List of Common and Scientific Names of Fishes from the United States and Canada*. 4th Ed. Special Publication No. 12. American Fisheries Society. Bethesda, MD.
- G) Burch, J.B. 1972. *Freshwater Sphaeriacean Clams (Mollusca: Pelecypoda) of North America*. *Biota of Freshwater Ecosystems Identification Manual No. 3*. U.S. G.P.O. #5501-0367.
- H) N. 1972. *Freshwater Polychaetes (Annelida) of North America*. *Biota of Freshwater Ecosystems Identification Manual No. 4*. U.S. G.P.O. #5501-0368.
- I) Williams, W. D. 1972. *Freshwater Isopods (Asellidae) of North America*. *Biota of Freshwater Ecosystems Identification Manual No. 7*. U.S. G.P.O. #5501-0390.
- J) Burch, J. B. 1973. *Freshwater Unionacean Clams (Mollusca: Pelecypoda) of North America*. *Biota of Freshwater Ecosystems Identification Manual No. 11*. 'U.S. G.P.O. #5501-00588.
- K) Kudo, R. R. 1966. *Protozoology*. 5th Ed. Thomas, Springfield, Illinois.
- L) Usinger, R. L. 1956. *Aquatic Insects of California*. University of California Press, Berkeley.
- M) Clarke, A. H. 1973. *The Freshwater Molluscs of the Canadian Interior Basin*. *Malacologia* 13: 1-509.
- N) Hilsenhoff, W.L. 1975. *Aquatic Insects of Wisconsin*. Technical Bulletin No. 89. Dept. of Natural Resources. Madison, Wisconsin.
- O) Edmunds, G. F., Jr., et al. 1976. *The Mayflies of North and Central America*. University of Minnesota Press, Minneapolis.
- P) Pennak, R. W. 1978. *Fresh-Water Invertebrates of the United States*. 2nd Ed. Wiley, New York.
- Q) Wentsell, R., et al. 1977. *Hydrobiologia* 56: 153-156.
- R) Johannsen, O.A. 1937. *Aquatic Diptera*. Part IV. Chironomidae: Subfamily Chironominae. *Memoir 210*. Cornell Univ. Agricultural Experimental Station, Ithaca, NY.
- S) Burks, B.D. 1953. *The Mayflies, or Ephemeroptera, of Illinois*. *Bulletin of the Natural History Survey Division*. Urbana, Illinois.

- T) Bousfield, E.L. 1973. Shallow-Water Gammaridean Amphipods of New England. Cornell University Press, Ithaca, New York.
- U) Sohn, I. G., and L. S. Kornicker. 1973. Morphology of *Cypretta kawatai* Sohn and Kornicker, 1972 (Crustacea, Ostracoda), with a Discussion of the Genus. Smithsonian Contributions to Zoology, No. 141.
- V) Needham, J. G., and M. J. Westfall, Jr. 1955. A Manual of the Dragonflies of North America. Univ. of California Press, Berkeley.
- W) Brown, H. P. 1972. Aquatic Dryopoid Beetles (Coleoptera) of the United States. Biota of Freshwater Ecosystems Identification Manual No. 6. U.S.G.P.O. #5501-0370.
- X) Parodiz, J.J. 1956. Notes on the Freshwater Snail *Leptoxis (Mudalia) carinata* (Bruguiere). Annals of the Carnegie Museum 33: 391-405.
- Y) Myers, F.J. 1931. The Distribution of Rotifera on Mount Desert Island. Am. Museum Novitates 494: 1-12.
- Z) National Academy of Sciences. 1974. Amphibians : Guidelines for the breeding, care, and management of laboratory animals. Washington, D.C.
- AA) Horne, F.R., and S. McIntosh. 1979. Factors Influencing Distribution of Mussels in the Blanco River in Central Texas. Nautilus 94: 119-133.
- BB) Klemm, D. J. 1972. Freshwater Leeches (Annelida: Hirudinea) of North America. Biota of Freshwater Ecosystems Identification Manual No. 8. U.S.G.P.O. #5501-0391.
- CC) Frison, T. H. 1935; The Stoneflies, or Plecoptera, of Illinois. Bull. Ill. Nat. History Survey, Vol. 20, Article 4.
- DD) White, A. M. Manuscript. John Carroll University, University Heights, Ohio.
- EE) Darsie, R.F., Jr., and R.A. Ward. 1981. Identification and Geographical Distribution of the Mosquitoes of North America, North of Mexico. American Mosquito Control Association, Fresno, California.
- FF) Holsinger, J.R. 1972. The Freshwater Amphipod Crustaceans (Gammaridae) of North America. Biota of Freshwater Ecosystems Identification Manual No. 5. U.S.G.P.O. #5501-0369.
- GG) Chapman, P. H., et al. 1982. Relative Tolerances of Selected Aquatic Oligochaetes to Individual Pollutants and Environmental Factors. Aquatic Toxicology 2: 47-67.
- HH) Bosnak, A.D., and E.L. Morgan. 1981. National Speleological Society Bull. 43: 12-18.
- II) Wiggins, G.B. 1977. Larvae of the North American Caddisfly Genera (Tricoptera). University of Toronto Press, Toronto, Canada.
- JJ) Hall, R. J. and D. Swineford. 1980. Toxic Effects of Endrin and Toxaphene on the Southern Leopard Frog *Rana sphenoccephala*. Environ. Pollut. (Series A) 23: 53-65.

Saltwater Species Table

Synonyms appear after the official Scientific Name and are marked with an asterisk (*).
 Non-resident species are noted in the Reference column and are marked with a dagger (†).

Class	Family	Species		Reference
		Common Name	Scientific Name	
Phylum: Cnidaria (Coelenterata) (48738)				
Hydroza 48739	Campanulariidae 49470	Hydroid	<i>Campanularia flexiosa</i> <i>Campanularia flexuosa</i> **	B122, E81
		Hydroid	<i>Laomedea loveni</i> ††	nonresident
		Hydromedusa	<i>Phialidium sp.</i>	(E81)
	Campanulinidae 49756	Hydroid	<i>Eirene viridula</i> †	nonresident
Phylum: Ctenophora (53856)				
Tentaculata 53858	Pleurobrachiidae 53860	Ctenophore	<i>Pleurobrachia pileus</i>	B218, E162
	Mnemiidae 53915	Ctenophore	<i>Mnemiopsis mccradyi</i>	C39, 194
Phylum: Nemertea (Rhynchocoela) (57411)				
Heteronemertea 57438	Lineidae 57443	Nemertine worm	<i>Cerebratulus fuscus</i>	B252
Phylum: Rotifera (Rotatoria) (58239)				
Monogononta 58342	Brachionidae 58344	Rotifer	<i>Brachionus plicatilis</i>	B272
Phylum: Annelida (64357)				
Polychaeta 64358	Phyllodoceidae 65228	Polychaete worm	<i>Phyllodoce maculata</i> <i>Anaitides maculata</i> <i>Nereiphylla maculata</i>	E334
	Nereididae (Nereidae) 65870	Polychaete worm	<i>Neanthes arenaceodentata</i> <i>Nereis arenaceodentata</i>	E377
		Polychaete worm	<i>Neanthes vaali</i> †	nonresident
		Polychaete worm	<i>Nereis diversicolor</i> <i>Neanthes diversicolor</i> *	E337, F527
		Sand worm	<i>Nereis virens</i> <i>Neanthes virens</i>	B317, E337, C58
		Polychaete worm	<i>Nereis sp.</i>	
	Dorvilleidae 66478	Polychaete worm	<i>Ophryotrocha diadema</i>	P23
		Polychaete worm	<i>Ophryotrocha labronica</i> † <i>Ophryotrocha labrunica</i> †	nonresident
	Spionidae 66781	Polychaete worm	<i>Polydora websteri</i>	E338
	Cirratulidae 67116	Polychaete worm	<i>Cirriformia spirabanchia</i>	G253
	Ctenodrilidae 67217	Polychaete worm	<i>Ctenodrilus serratus</i>	G275
	Capitellidae 67413	Polychaete worm	<i>Capitella capitata</i>	B358, E337
	Arenicolidae 67500	Polychaete worm	<i>Arenicola marina</i>	B369, E337

** Synonym

†† Non-resident species

Class	Family	Species		Reference
		Common Name	Scientific Name	
	Sabellidae 68076	Polycheate worm	<i>Eudistylia vancouveri</i>	DD
Oligochaeta 68422	Tubificidae 68585	Oligochaete worm	<i>Limnodriloides verrucosus</i>	Z
		Oligochaete worm	<i>Monopylephorus cuticulatus</i>	Z
		Oligochaete worm	<i>Peloscolex gabriellae</i> <i>Tubificoides gabriellae</i>	Z
Phylum: Mollusca (69458)				
Gastropoda 69459	Haliotidae 566897	Black abalone	<i>Haliotis cracherodii</i>	C88, D17
		Red abalone	<i>Haliotis rufescens</i>	D18
	Calyptraeidae 72611	Common Atlantic slippershell	<i>Crepidula fornicata</i>	C90, D141
	Muricidae 73236	Oyster drill	<i>Urosalpinx cinerea</i> <i>Urosalpinx cinereus</i>	B646, D179, E264
	Melongenidae (Neptuneidae) 74069	Channeled whelk	<i>Busycotypus canaliculatus</i> (Formerly <i>Busycon canaliculatum</i>)	B655, D223, E264
	Nassariidae (Nassidae) 74102	Mud snail	<i>Nassarius obsoletus</i> <i>Nassa obsoleta</i> <i>Icyanassa obsoleta</i>	B649, D226, E264
Bivalvia (Pelecypoda) 79118	Mytilidae 79451	Northern horse mussel	<i>Modiolus modiolus</i>	D434
		Blue mussel	<i>Mytilus edulis</i>	B566, C101, D428, E299
		Mediterranean mussel	<i>Mytilus galloprovincialis</i> [†]	nonresident
	Pectinidae 79611	Bay scallop	<i>Argopecten irradians</i>	D447
	Ostreidae 79866	Pacific oyster	<i>Crassostrea gigas</i>	C102, D456, E300
		Eastern oyster	<i>Crassostrea virginica</i>	D456, E300
		Oyster	<i>Crassostrea sp.</i>	1
		Oyster	<i>Ostrea edulis</i>	E300
	Cardiidae 80865	Cockle	<i>Cerastoderma edule</i> [†] <i>Cardium edule</i> [†]	nonresident
	Mactridae 80942	Clam	<i>Mulinia lateralis</i>	D491
		Common rangia	<i>Rangia cuneata</i>	D491, E301
		Surf clam	<i>Spisula solidissima</i>	B599, D489, E301
	Tellinidae 81032	Clam	<i>Macoma inquinata</i>	D507
		Bivalve	<i>Tellina tenuis</i> [†]	nonresident
	Veneridae 81439	Quahog clam	<i>Mercenaria mercenaria</i>	D523, E301
		Common Pacific littleneck	<i>Protothaca staminea</i>	D526
		Japanese littleneck clam	<i>Tapes philippinarum</i>	D527
	Myidae 81688	Soft-shell clam	<i>Mya arenaria</i>	B602, D536, E302
	Phylum: Arthropoda (82696)			
Merostomata 82698	Limulidae 82701	Horseshoe crab	<i>Limulus polyphemus</i>	B533, E403, H30
Branchiopoda (Formerly Crustacea) 83687	Artemiidae 83689	Brine shrimp	<i>Artemia salina</i> [†]	² nonresident
Maxillopoda (Formerly Crustacea) 621145	Calanidae 85259	Copepod	<i>Calanus helgolandicus</i>	Q25
		Copepod	<i>Undinula vulgaris</i>	Q29
	Eucalanidae 85299	Copepod	<i>Eucalanus elongatus</i>	AA
		Copepod	<i>Subeucalanus pileatus</i> <i>Eucalanus pileatus</i>	AA
	Pseudocalanidae 85351	Copepod	<i>Pseudocalanus minutus</i>	E447, I155, Q43

Class	Family	Species		Reference
		Common Name	Scientific Name	
	Euchaetidae 85524	Copepod	<i>Euchaeta marina</i>	Q63
	Metridinidae (Formerly Metridiidae) 593501	Copepod	<i>Metridia pacifica</i>	X179, Y
	Pseudodiaptomidae 85847	Copepod	<i>Pseudodiaptomus coronatus</i>	E447, I154, Q101
	Temoridae 85855	Copepod	<i>Eurytemora affinis</i>	E450, I155, Q111
	Pontellidae 86038	Copepod	<i>Labidocera scotti</i>	R157
	Acartiidae 86083	Copepod	<i>Acartia clausi</i>	E447
		Copepod	<i>Acartia tonsa</i>	E447, I154
	Harpacticidae 86329	Copepod	<i>Tigriopus californicus</i>	J78
		Copepod	<i>Tigriopus japonicus</i> ¹	nonresident
	Tisbidae 86444	Copepod	<i>Tisbe holothuriae</i>	BB
	Ameiridae (Formerly Canthocamptidae) 86999	Copepod	<i>Nitokra spinipes</i> <i>Nitocra spinipe</i>	Q240
	Archaeobalanidae (Formerly Balanidae) 89681	Barnacle	<i>Semibalanus balanoides</i> <i>Balanus balanoides</i>	B424, E457
	Balanidae 89599	Barnacle	<i>Balanus crenatus</i>	B426, E457
		Barnacle	<i>Balanus eburneus</i>	B424, E457
		Barnacle	<i>Balanus improvisus</i>	B426, E457
Malacostraca (Formerly Crustacea) 89787	Mysidae 89856	Mysid	<i>Heteromysis formosa</i>	E513, K720
		Mysid	<i>Americamysis bahia</i> <i>Mysidopsis bahia</i> ¹	U173
		Mysid	<i>Americamysis bigelowi</i> <i>Mysidopsis bigelowi</i> ¹	E513, K720
		Mysid	<i>Neomysis sp.</i>	1
	Idoteidae 92564	Isopod	<i>Idotea balthica</i> <i>Idothea baltica</i> ¹	B446, E483
		Isopod	<i>Idotea emarginata</i> ¹	nonresident
		Isopod	<i>Idotea neglecta</i> ¹	nonresident
	Janiridae 92810	Isopod	<i>Jaera albifrons</i> ¹	nonresident
		Isopod	<i>Jaera albifrons sensu</i> ¹	nonresident
		Isopod	<i>Jaera nordmanni</i> ¹	nonresident
	Ampeliscidae 93320	Amphipod	<i>Ampelisca abdita</i>	E488, L136
	Eusiridae (Pontogeneiidae) 93681	Amphipod	<i>Pontogeneia sp.</i>	1
	Gammaridae 93745	Amphipod	<i>Gammarus duebeni</i>	L56
		Amphipod	<i>Gammarus oceanicus</i>	E489, L50
		Amphipod	<i>Gammarus tigrinus</i>	L51
		Amphipod	<i>Gammarus zaddachi</i> ¹	nonresident
		Amphipod	<i>Marinogammarus obtusatus</i>	L58
	Uristidae (Formerly Lysianassidae) 621432	Amphipod	<i>Anonyx sp.</i>	1
	Euphausiidae (Thysanopodidae) 95500	Euphausiid	<i>Euphausia pacifica</i>	M15
	Penaeidae	Brown shrimp	<i>Penaeus aztecus</i>	E518, N17

Class	Family	Species		Reference
		Common Name	Scientific Name	
	95602	Pink shrimp	<i>Penaeus duorarum</i>	E518, N17
		White shrimp	<i>Penaeus setiferus</i>	E518, N17
		Blue Shrimp	<i>Penaeus stylirostris</i> ^f	nonresident
	Palaemonidae 96213	Shrimp	<i>Leander paucidens</i> ^f	nonresident
		Prawn	<i>Leander squilla</i> ^f <i>Palaemon elegans</i> ^f	nonresident
		Prawn	<i>Macrobrachium rosenbergii</i>	³
		Korean shrimp	<i>Palaemon macrodactylus</i>	T380
		Grass shrimp	<i>Palaemonetes pugio</i>	E521, N59
		Grass shrimp	<i>Palaemonetes vulgaris</i>	B500, E521, N56
	Hippolytidae 96746	Sargassum shrimp	<i>Latreutes fucorum</i>	N78
	Pandalidae 96965	Coon stripe shrimp	<i>Pandalus danae</i>	T306, W163
		Shrimp	<i>Pandalus goniurus</i>	W163
		Pink shrimp	<i>Pandalus montagui</i>	B494, E522, W163
	Crangonidae 97106	Sand shrimp	<i>Crangon crangon</i> ^f	nonresident
		Bay shrimp	<i>Crangon franciscorum</i> <i>Crago franciscorum</i>	V176, W164
		Shrimp	<i>Crangon nigricauda</i>	V176, W164
		Sand shrimp	<i>Crangon septemspinosa</i>	B500, E522
	Nephropidae (Homaridae) 97307	American lobster	<i>Homarus americanus</i>	B502, E532
		European lobster	<i>Homarus gammarus</i> ^f	nonresident
	Paguridae 97774	Hermit crab	<i>Pagurus longicarpus</i>	B514, E537, N125
	Cancridae 98670	Rock crab	<i>Cancer irroratus</i>	B518, E543, N175
		Dungeness crab	<i>Cancer magister</i>	T166, V185, W177
	Portunidae 98689	Blue crab	<i>Callinectes sapidus</i>	B521, C80, E543, N168
		Green crab	<i>Carcinus maenas</i>	C80, E543
	Xanthidae (Pilumnidae) 98748	Mud crab	<i>Eurypanopeus depressus</i>	B522, E543, N195
		Crab	<i>Leptodius floridanus</i>	S80
		Mud crab	<i>Rhithropanopeus harrisii</i>	E543, N187
Varunidae (formerly Grapsidae) 621521	Shore crab	<i>Hemigrapsus nudus</i>	CC	
	Shore crab	<i>Hemigrapsus oregonensis</i>	CC	
Sesamidae (formerly Grapsidae) 621520	Drift line crab	<i>Armases cinereum</i> (<i>Sesarma cinereum</i>)	B526, E544, N222	
	Crab	<i>Sesarma haematocheir</i> ^f	nonresident	
Ocypodidae 99080	Fiddler crab	<i>Uca pugilator</i>	B526, E544, N232	
Phylum: Echinodermata (156857)				
Asteroidea 156862	Asteriidae 157212	Starfish	<i>Asterias forbesi</i>	B728, E578, O392
Ophiuroidea 157325	Ophiothricidae 157792	Brittle star	<i>Ophiothrix spiculata</i>	O672, T526
Echinoidea 157821	Arbaciidae 157904	Sea urchin	<i>Arbacia lixula</i> ^f	nonresident
		Sea urchin	<i>Arbacia punctulata</i>	B762, E572
	Toxopneustidae 157919	Sea urchin	<i>Lytechinus pictus</i>	T253
		Sea urchin	<i>Pseudocentrotus depressus</i> ^f	nonresident
	Echinidae 157940	[chinoderm	<i>Paracentrotus lividus</i> ^f	nonresident
Echinometridae 157955	Coral reef echinoid	<i>Echinometra mathaei</i> ^f	nonresident [Hawaii only]	

Class	Family	Species		Reference
		Common Name	Scientific Name	
	Strongylocentrotidae 157965	Sea urchin	<i>Strongylocentrotus purpuratus</i>	O574, T202
	Dendrasteridae 158008	Sand dollar	<i>Dendraster excentricus</i>	O537, V363
Phylum: Chaetognatha (158650)				
Sagittoidea 158655	Sagittidae 158726	Arrow worm	<i>Ferosagitta hispida</i> <i>Sagitta hispida</i>	E218
Phylum: Chordata (158852)				
Chondrichthyes 159785	Rajidae 160845	Thornback ray	<i>Raja clavata</i> [†]	nonresident
Actinopterygii (Formerly Osteichthyes) 161061	Anguillidae 161125	American eel	<i>Anguilla rostrata</i>	A15
		Clupeidae 161700	Atlantic menhaden	<i>Brevoortia tyrannus</i>
	Gulf menhaden		<i>Brevoortia patronus</i>	A17
	Atlantic herring		<i>Clupea harengus</i> <i>Clupea harengus harengus</i>	A17
	Pacific herring		<i>Clupea pallasii</i> <i>Clupea harengus pallasii</i>	A17
	Herring		<i>Clupea harengus</i>	A17
	Engraulidae 553173	Northern anchovy	<i>Engraulis mordax</i>	A18
		Nehu	<i>Encrasicholina purpurea</i> [†] <i>tolephorus purpureus</i> [†]	nonresident [Hawaii only]
	Salmonidae 161931	Pink salmon	<i>Oncorhynchus gorbuscha</i>	A18
		Chum salmon	<i>Oncorhynchus keta</i>	A18
		Coho salmon	<i>Oncorhynchus kisutch</i>	A18
		Sockeye salmon	<i>Oncorhynchus nerka</i>	A19
		Chinook salmon	<i>Oncorhynchus tshawytscha</i>	A19
		Rainbow trout (Steelhead trout)	<i>Oncorhynchus mykiss</i> (Formerly <i>Salmo gairdneri</i>)	A19
		Atlantic salmon	<i>Salmo salar</i>	A19
	Gadidae 164701	Atlantic cod	<i>Gadus morhua</i>	A30
		Haddock	<i>Melanogrammus aeglefinus</i>	A30
	Cyprinodontidae 165629	Sheepshead minnow	<i>Cyprinodon variegatus</i>	A33
		Mummichog	<i>Fundulus heteroclitus</i>	A33
		Striped killifish	<i>Fundulus majalis</i>	A33
		Longnose killifish	<i>Fundulus similis</i>	A33
	Poeciliidae 165876	Mosquitofish	<i>Gambusia affinis</i>	A33
Sailfin molly		<i>Poecilia latipinna</i>	A34	
Atherinidae 165984	Inland silverside	<i>Menidia beryllina</i>	A34	
	Atlantic silverside	<i>Menidia menidia</i>	A34	
	Tidewater silverside	<i>Menidia peninsulae</i>	A34	
Gasterosteidae 166363	Threespine stickleback	<i>Gasterosteus aculeatus</i>	A35	
	Fourspine stickleback	<i>Apeltes quadracus</i>	A35	
Syngnathidae 166443	Northern pipefish	<i>Syngnathus fuscus</i>	A36	
Percichthyidae 170315	Striped bass	<i>Morone saxatilis</i> (<i>Roccus saxatilis</i> , Obs.)	A36	
Kuhliidae 168083	Mountain bass	<i>Kuhlia sandvicensis</i> [†]	nonresident [Hawaii only]	
Carangidae 168584	Florida Pompano	<i>Trachinotus carolinus</i>	A43	

Class	Family	Species		Reference
		Common Name	Scientific Name	
	Sparidae 169180	Pinfish	<i>Lagodon rhomboides</i>	A45
	Sciaenidae 169237	Spot	<i>Leiostomus xanthurus</i>	A46
		Atlantic croaker	<i>Micropogonias undulatus</i>	A46
		Red drum	<i>Sciaenops ocellatus</i>	A46
	Embiotocidae 169735	Shiner perch	<i>Cymatogaster aggregata</i>	A47
		Dwarf perch	<i>Micrometrus minimus</i>	A48
	Pomacentridae 170044	Blacksmith	<i>Chromis punctipinnis</i>	A48
	Labridae 170477	Cunner	<i>Tautoglabrus adspersus</i>	A49
		Bluehead	<i>Thalassoma bifasciatum</i>	A49
	Mugilidae 170333	Mullet	<i>Aldrichetta forsteri</i> ¹	nonresident
		Striped mullet	<i>Mugil cephalus</i>	A49
		White mullet	<i>Mugil curema</i>	A49
	Ammodytidae 171670	Pacific sand lance	<i>Ammodytes hexapterus</i>	A53
	Gobiidae 171746	Longjaw mudsucker	<i>Gillichthys mirabilis</i>	A54
		Naked goby	<i>Gobiosoma bosci</i>	A54
	Cottidae 167196	Tidepool sculpin	<i>Oligocottus maculosus</i>	A61
	Bothidae 172714	Speckled sanddab	<i>Citharichthys stigmaeus</i>	A64
		Summer Flounder	<i>Paralichthys dentatus</i>	A64
	Pleuronectidae 172859	Dab	<i>Limanda limanda</i> ¹	nonresident
		Plaice	<i>Pleuronectes platessa</i> ¹	nonresident
		English sole	<i>Parophrys vetulus</i>	A65
		Winter flounder	<i>Pseudopleuronectes americanus</i>	A65
	Balistidae 173128	Planehead filefish	<i>Monacanthus hispidus</i>	A66
	Tetraodontidae 173283	Northern puffer	<i>Sphoeroides maculatus</i>	A66

Footnotes for Saltwater Species

- ¹ Organisms not identified to species are considered resident only if obtained from wild populations in North America.
- ² This species should not be used because it might be too atypical.
- ³ This species might be established in portions of the southern United States.

References for Saltwater Species

- A) Committee on Names of Fishes. 1980. A List of Common and Scientific Names of Fishes from the United States and Canada. 4th Ed. Special Publication No. 12. American Fisheries Society, Bethesda, MD.
- B) Miner, R. W. 1950. Field Book of Seashore Life. Van Rees Press, New York.
- C) George, D., and J. George. 1979. Marine Life: An Illustrated Encyclopedia of Invertebrates in the Sea. Wiley-Interscience, New York.
- D) Abbott, R.T. 1974. American Seashells. 2nd Ed. Van Nostrand Reinhold Company, New York.
- E) Gosner, K.L. 1971. Guide to Identification of Marine and Estuarine Invertebrates: Cape Hatteras to the Bay of Fundy. Wiley-Interscience, New York; Gosner, K.L. 1979. A Field Guide to the Atlantic Seashore. Houghton Mifflin, Boston.
- F) Hartmann, O. 1968. Atlas of the Errantiate Polychaetous Annelids from California. Allan Hancock Foundation, University of Southern California, Los Angeles, California.
- G) Hartmann, O. 1969. Atlas of the Sedentariate Polychaetous Annelids from California. Allan Hancock Foundation, University of Southern California, Los Angeles, California.
- H) Cooley, N.R. 1978. An Inventory of the Estuarine Fauna in the Vicinity of Pensacola, Florida. Florida Marine Research Publication No. 31. Florida Department of Natural Resources, St. Petersburg, Florida.
- I) Zingmark, R.G. (ed.) 1978. An Annotated Checklist of the Biota of the Coastal Zone of South Carolina. University of South Carolina Press, Columbia, South Carolina.
- J) Monk, C.R. 1941. Marine Harpacticoid Copepods from California. Trans. Amer. Microsc. Soc. 60:75-99.
- K) Wigley, R., and B.R. Burns. 1971. Distribution and Biology of Mysids (Crustacea, Mysidacea) from the Atlantic Coast of the United States in the NMFS Woods Hole Collection. Fish. Bull. 69(4):717-746.
- L) Bousfield, E.L. 1973. Shallow-Water Gammaridean Amphipoda of New England. Cornell University Press. Ithaca, New York.
- M) Ponomareva, L.A. Euphausiids of the North Pacific, their Distribution, and Ecology. Jerusalem: Israel Program for Scientific Translations. 1966. Translated from the Russian by S. Nemchonok. TT65-50098. NTIS, Springfield, VA.
- N) Williams, A.B. 1965. Marine Decapod Crustaceans of the Carolinas. Fish. Bull. 65(1):1-298.
- O) Hyman, L.H. 1955. The Invertebrates: Echinodermata. Vol. IV. McGraw-Hill, New York.
- P) Akesson, B. 1976. Morphology and Life Cycle of *Ophryotrocha diadema*, a New Polychaete Species from California. Ophelia 15(1): 23-25.
- Q) Wilson, C.B. 1932. The Copepods of the Woods Hole Region, Massachusetts. U.S. Nat. Mus. Bull. 158: 1-635.
- R) Fleminger, A. 1956. Taxonomic and Distributional Studies on the Epiplanktonic Calanoid Copepods (Crustacea) of the Gulf of Mexico. Dissertation. Harvard University, Cambridge.
- S) Menzel, R.W. 1956. Annotated Checklist of the Marine Fauna and Flora of the St. George 's Sound - Apalachee Bay region, Florida Gulf Coast. Contrib. No. 61. Fla. State Univ. Oceanogr. Inst.

- T) Ricketts, E.F., and J. Calvin. (Revised by Joel W. Hedgpeth). 1968. *Between Pacific Tides*. Stanford University Press, Stanford, California.
- U) Price, W.W. 1978. Occurrence of *Mysidopsis almyra* Bowman, *M. bahia* Molenock and *Bowmaniella brasiliensis* Bacescu (Crustacea, Mysidacea) from the Eastern Gulf of Mexico. *Gulf Res. Reports* 6(2): 173-175.
- V) Light, S.F. (Revised by R.I. Smith, et al.). 1961. *Intertidal Invertebrates of the Central California Coast*. University of California Press, Los Angeles, California.
- W) Kozloff, E.N. 1974. *Keys to the Marine Invertebrates of Puget Sound, the San Juan Archipelago, and Adjacent Regions*. University of Washington Press, Seattle, Washington.
- X) Calcofi Atlas. No.19. California Cooperative Oceanic Fisheries Investigations, State of California Marine Research Committee. Pp. 179-185.
- Y) Brodskii, K.A. 1967. Calanoida of the Far Eastern Seas and Polar Basin of the U.S.S.R. Jerusalem Series, Keys to the Fauna of the U.S.S.R. Zoological Inst., Academy Sciences, U.S.S.R. No. 35.
- Z) Chapman, P.M., et al. 1982. Relative Tolerances of Selected Aquatic Oligochaetes to Individual Pollutants and Environmental Factors. *Aquatic Toxicology* 2: 47-67.
- AA) Venkataramiak, A., et al. 1982. Studies on Toxicity of OTEC Plant Components on *Eucalanus* sp. from the Gulf of Mexico. *Ocean Science and Engineering*.
- BB) Zingmank, R.G. (ed.). 1978. *An Annotated Checklist of the Biota of the Coastal Zone of South Carolina*. University of South Carolina Press.
- CC) Thatcher, T.O 1978. The Relative Sensitivity of Pacific Northwest Fishes and Invertebrates to Chlorinated Sea Water. In: R.L. Jolley, et al. (eds.), *Water Chlorination: Environmental Impact and Health Effects*. Vol. 2. Ann Arbor Science Publishers, Ann Arbor, Michigan. p. 341.
- DD) Young, J.S., et al. 1979. Effects of Copper on the Sabelled Polychaete, *Eudistylia vancouveri*: I. Concentration Limits for Copper Accumulation. *Arch. Environ. Contam. Toxicol.* 8: 97-106.

Appendix 2. Example Calculation of Final Acute Value, Computer Program, and Printouts

A. Example Calculation

N = total number of MAVs in data set = 8

Rank	MAV	ln(MAV)	ln(MAV) ²	P = R / (N+1)	√P
4	6.4	1.8563	3.4458	0.44444	0.66667
3	6.2	1.8245	3.3290	0.33333	0.57735
2	4.8	1.5686	2.4606	0.22222	0.47140
1	0.4	-0.9163	0.8396	0.11111	0.33333
Sum		4.3331	10.0750	1.11110	2.04875

$$S^2 = \frac{10.0750 - (4.3331)^2 / 4}{1.11110 - (2.04875)^2 / 4} = 87.134$$

$$S = 9.3346$$

$$L = [4.3331 - (9.3346)(2.04875)] / 4 = -3.6978$$

$$A = (9.3346)(\sqrt{0.05}) - 3.6978 = -1.6105$$

$$FAV = e^{-1.6105} = 0.1998$$

B. Example Computer Program in BASIC Language for Calculating the FAV

```

10   REM This program calculates the FAV when there are less than
20   REM 59 MAVs in the data set
30   X = 0
40   X2 = 0
50   Y = 0
60   Y2 = 0
70   PRINT "How many MAVs are in the data set?"
80   INPUT N
90   PRINT "What are the four lowest MAVs?"
100  FOR R = 1 TO 4
110      INPUT V

```

```

120      X = X + LOG(V)
130      X2 = X2 + (LOG(V)) * (LOG(V))
140      P = R / (N + 1)
150      Y2 = Y2 + P
160      Y = Y + SQR((X2 - X * X / 4))
170  NEXT R
180  S = SQR((X2 - X * X / 4) / (Y2 - Y * Y / 4))
190  L = (X - S * Y) / 4
200  A = S * SQR(0.05) + L
210  F = EXP(A)
220  PRINT "FAV = " F
230  END

```

C. Example Printouts from Program

How many MAVs are in the data set?

? 8

What are the four lowest MAVs?

? 6.4

? 6.2

? 4.8

? .4

FAV = 0.1998

How many MAVs are in the data set?

? 16

What are the four lowest MAVs?

? 6.4

? 6.2

? 4.8

? .4

FAV = 0.4365