United States
Environmental Protection
Agency

Office of Science and Technology
Office of Water
Washington, D.C. 20460

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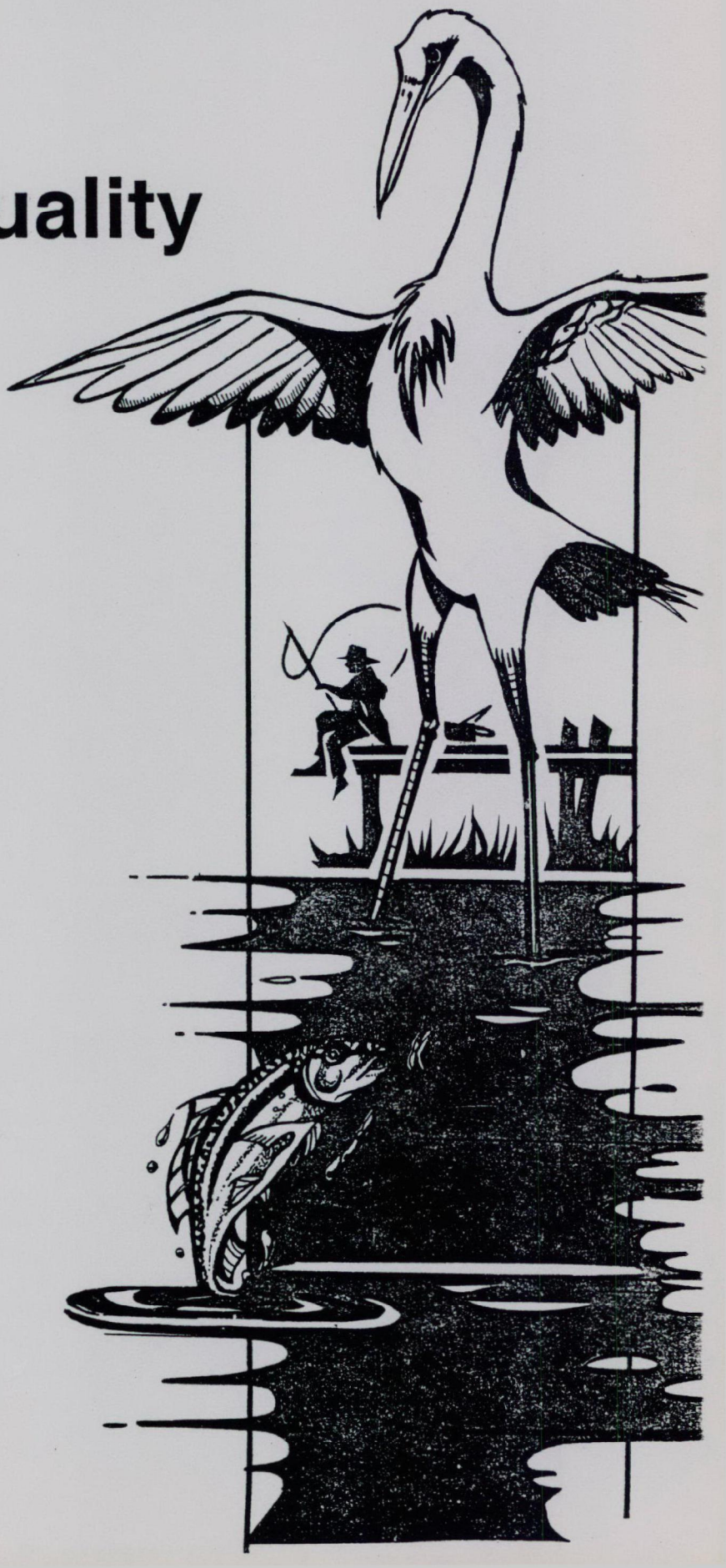
Water Quality Criteria



Quality Criteria for Water 1993

Prepared for

Health and Ecological Criteria Division
Office of Water
U.S. Environmental Protection Agency
Washington, D.C.

1993

NOTE TO USERS

Quality Criteria for Water 1993 is produced as a summary document. Individual criteria in this document are taken from previously published criteria. To obtain further information on these criteria, review the information in Appendix E. Each criteria entry lists the Federal Register notification and publication date.

This publication contains criteria issued since 1976. Different methodologies have been used to derive these criteria. Appendixes A through D provide the various available methodologies; notes at the end of each criterion indicate which methodology applies.

Individuals who use this document are encouraged to obtain a copy of the *Criteria Document for the Pollutant of Interest*. These documents contain the data sets and references upon which the criteria in this summary document were derived.

PREFACE

Section 304(a)(1) of the Clean Water Act (33 U.S.C. 1314[a][1] requires the Environmental Protection Agency (EPA) to publish and periodically update ambient water quality criteria. Intended neither as rules nor regulations, these criteria present scientific data and guidance on the environmental effects of pollutants that can be used to derive regulations based on considerations of water quality impacts.

These criteria are to accurately reflect the latest scientific knowledge (a) on the kind and extent of all identifiable effects on health and welfare including, but not limited to, plankton, fish, shellfish, wildlife, plant life, shorelines, beaches, aesthetics, and recreation that may be expected from the presence of pollutants in any body of water including groundwater; (b) on the concentration and dispersal of pollutants, or their byproducts, through biological, physical, and chemical processes; and (c) on the effects of pollutants on biological community diversity, productivity, and stability, including information on the factors affecting rates of eutrophication and organic and inorganic sedimentation for varying types of receiving waters.

The first of these publications appeared in 1968 with *The Report of the National Technical Advisory Committee to the Secretary of the Interior "Green Book." Water Quality Criteria 1972* (the "Blue Book") was published in 1973, followed three years later by the "Red Book," *Quality Criteria for Water*.

On November 28, 1980 (45 F.R. 79318), EPA announced through the *Federal Register* the publication of 64 individual ambient water quality criteria documents for pollutants listed as toxic under section 307(a)(1) of the Clean Water Act. On February 15, 1984 (49 F.R. 5831); July 29, 1985 (50 F.R. 30784); March 7, 1986 (51 F.R. 8012); June 24, 1986 (51 F.R. 22978); December 3, 1986 (51 F.R. 43665); and March 2, 1987 (52 F.R. 6213); EPA published additional water quality criteria documents, followed by *Quality Criteria for Water 1986* (the "Gold Book"). The National Toxics Rule (NTR) was promulgated on December 22, 1993 (57 F.R. 60848). The NTR was a national rulemaking that provided the most current criteria for the priority pollutants. This rulemaking recalculated the priority pollutants criteria, based on data that was available at its release. *Quality Criteria for Water 1993* draws information from its predecessors in presenting summaries of all the contaminants for which EPA has developed criteria recommendations. The rationale for these recommendations can be found in the reference identified at the end of each criteria summary.

This document is intended as a summary only. Specific data on each criterion can be found in individual criteria documents. Copies of the individual ambient water quality criteria documents containing all the data used to develop the criteria recommendations summarized here, are available from National Technical Information Service, 5285 Port Royal Road, Springfield, VA 22161, (703) 487-4650.

This book is intended for easy reference use. The Contents lists compounds by their common names.

To obtain copies of this document and supplements, contact the Government Printing Office at (202) 783-3238.

EPA's goal is to continue to develop and make available ambient water quality criteria reflecting the latest scientific information. This, we believe, constitutes a major component in our ongoing commitment to improve and protect the quality of our Nation's waters.

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WATER QUALITY CRITERIA SUMMARY
CONCENTRATIONS (in μg/L)

U.S. ENVIRONMENTAL PROTECTION AGENCY
Office of Science and Technology
Health and Ecological Criteria Division
Ecological Risk Assessment Branch (WH-586)
Human Risk Assessment Branch (WH-586)

								HUMAN HEALTH (10-6 RISK LEVEL FOR CARCINOGENS)					
								PUBLISHED CRITERA		RECALCULATED VALUES using IRIS, as of 9/90			
	CAS#	PRIORITY POLLUTANT	CARCINOGEN	FRESH ACUTE CRITERIA	FRESH CHRONIC CRITERIA	SALTWATER ACUTE CRITERIA	SALTWATER CHRONIC CRITERIA	WATER & ORGANISMS	ORGANISMS ONLY	WATER & ORGANISMS	ORGANISMS ONLY	DRINKING WATER MCL	CRITERIA FEDERAL REGISTER NOTICE
ACENAPHTHENE	83-32-9	Y	N	*1,700.	*520.	*970.	*710			1,200.	2,700		57 FR 60890
ACENAPHTHYLENE	208-96-8	Y	Y										57 FR 60913
ACROLEIN	107-02-8	Y	N	*68.	*21.	*55.		320.	780				45 FR 79324
ACRYLONITRILE	107-13-1	Y	Y	*7,550.	*2,600.			0.058	0 65	0.059	0.66		45 FR 79324
ALACHLOR	15972-60-8	N	Y									2 0	—
ALDRIN	309-00-2	Y	Y	3.0		1.3		0 000074	0 000079	0 00013	0.00014		45 FR 79325
ALKALINITY	—	N	N		20,000								RB
ALUMINUM	7429-90-5	N	N	CRITERIA ARE pH DEPENDENT — SEE DOCUMENT									53 FR 33178
AMMONIA	7664-41-7	N	N	CRITERIA ARE pH AND TEMPERATURE DEPENDENT — SEE DOCUMENT									54 FR 19227
ANTHRACENE	120-12-7	Y	Y							0.00012	0.00054		57 FR 60913
ANTIMONY	7440-36-0	Y	N	/p/88	/p/30	/p/1500	/p/500	146	45,000	14.	4,300.		45 FR 79325
ARSENIC	7440-38-2	Y	Y					0.0022	0.0175	0.018	0.14		45 FR 79325
ARSENIC(V)	17428-41-0	Y	Y	*850.		*2,319.							50 FR 30789
ARSENIC(III)	22569-72-8	Y	Y	360.	190.	69.	36.						50 FR 30786
ASBESTOS	1332-21-4	Y	Y					30k fibers/L				7 MFL	57 FR 60911
ATRAZINE	1912-24-9	N	N									3 0	—
BACTERIA	—	N	N	FOR PRIMARY RECREATION AND SHELLFISH USES — SEE DOCUMENT									51 FR 8012
BARIUM	7440-39-3	N	N					1,000				/p/2.000	RB
BENZENE	71-43-2	Y	Y	*5,300.		*5,100	*700.	0.66	40.	1 2	71.	5.0	57 FR 60911
BENZIDINE	92-87-5	Y	Y	*2,500.				0.00012	0 00053	0 00012	0 00054		57 FR 60913
BENZOFLUORANTHENE, 3,4-	205-99-2	Y	Y							0.0028	0.0311		57 FR 60913
BENZO(A) ANTHRACENE	56-55-3	Y	Y							0.0028	0.0311		57 FR 60913
BENZO(A)PYRENE	50-32-8	Y	Y							0.0028	0.0311		57 FR 60913
BENZO(GHI)PERYLENE	191-24-2	Y	Y										57 FR 60913

	CAS#	PRIORITY POLLUTANT	CARCINOGEN	FRESH ACUTE CRITERIA	FRESH CHRONIC CRITERIA	SALTWATER ACUTE CRITERIA	SALTWATER CHRONIC CRITERIA	HUMAN HEALTH (10-6 RISK LEVEL FOR CARCINOGENS) PUBLISHED CRITERA WATER & ORGANISMS	PUBLISHED CRITERA ORGANISMS ONLY	RECALCULATED VALUES using IRIS, as of 9/90 WATER & ORGANISMS	RECALCULATED VALUES using IRIS, as of 9/90 ORGANISMS ONLY	DRINKING WATER MCL	CRITERIA FEDERAL REGISTER NOTICE
BENZO(K) FLUORANTHENE	207-08-9	Y	Y							0.0028	0.0311		57 FR 60913
BERYLLIUM	7440-41-7	Y	Y	*130.	*5.3			0.0037	0.0641				57 FR 60848
BETA PARTICLE and PHOTON ACTIVITY	—	N	Y									4 mrem	—
BHC	680-73-1	N	Y	*100.		*0 34							45 FR 79335
BROMOFORM	75-25-2	Y	Y							4.3	360.		57 FR 60911
BUTYLBENZYL PHTHALATE	85-68-7	Y	N							3,000.	5,200.		57 FR 60890
CADMIUM	7440-43-9	Y	N	3.9+	1.1+	43.	9.3	10.				5 0	57 FR 60848
CARBOFURAN	1563-66-2	N	N									40.	—
CARBON TETRACHLORIDE	56-23-5	Y	Y	*35,200.		*50,000.		0 4	6.94	0.25	4.4	5.0	57 FR 60911
CHLORDANE	57-74-9	Y	Y	2.4	0.0043	0.09	0.004	0 00046	0.00048	0.00057	0.00059	2.0	45 FR 79327
CHLORIDE	16887-00-6	N	N	860,000.	230,000.								53 FR 19028
CHLORINATED BENZENES	—	Y	Y	*250.	*50	*160.	*129.			488.			45 FR 79327
CHLORINATED NAPHTHALENES	—	Y	N	*1,600.		*7.5				1,700.	4,300.		57 FR 60890
CHLORINE	7782-50-5	N	N	19.	11.	13	7.5						50 FR 30788
CHLOROALKYL ETHERS		Y	N	*238,000.									45 FR 79330
CHLOROBENZENE	108-90-7	Y	N					488		680.	21,000.	100.	57 FR 60911
CHLORODIBRO-MOMETHANE	124-48-1	Y	Y							0.41	34.		57 FR 60911
CHLOROFORM	67-66-3	Y	Y	*28,900.	*1,240.			0.19	15.7	5.7	470.		57 FR 60911
CHLOROPHENOL, 2-	95-57-8	Y	N	*4,380.						120.	400.		57 FR 60890
CHLOROPHENOL, 4-	106-48-9	N	N			*29,700.							45 FR 79329
CHLOROPHENOL, 4-, METHYL, 3-	59-50-7	Y	N	*30.									45 FR 79329
CHLOROPHENOXY HERBICIDE (2,4,5,-TP)	93-72-1	N	N					10				50	RB
CHLOROPHENOXY HERBICIDE (2,4-D)	94-75-7	N	N					100.				70.	RB
CHLORPYRIFOS	2921-88-2	N	N	0.083	0.041	0.011	0.0056						51 FR 43666
CHROMIUM (VI)	7440-47-3	Y	N	16.	11.	1,100	50.	50.		170.	3,400.	100.	50 FR 30788

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CHROMIUM (III)	1308-14-1	Y	N	1,700.-	210.+	*10,300		170,000	3,433,000	33,000.	670,000.	100.	50 FR 30788
CHRYSENE	218-01-9	Y	Y							0 0028	0.0311		57 FR 60913
COLOR	—	N	N	NARRATIVE STATEMENT — SEE DOCUMENT									RB
COPPER	7440-50-8	Y	N	18 +	12.+	2.9						/p/1300.	50 FR 30789
CYANIDE	57-12-5	Y	N	22.	5.2	1.0		200		700.	220,000.		57 FR 60911
DDT	50-29-3	Y	Y	1.1	0 001	0.13	0 001	0.000024	0 000024	0.00059	0.00059		57 FR 60914
DDT METABOLITE (DDD) (TDE)	72-54-8	Y	Y	*0.6		*3.6				0 00083	0.00084		57 FR 60914
DDT METABOLITE (DDE)	72-55-9	Y	Y	*1,050.		*14.				0 00059	0.00059		57 FR 60914
DEMETON	8065-48-3	N	N		0.1		0.1						RB
DIBENZO(A,H) ANTHRACENE	53-70-3	Y	Y							0.0028	0.0311		57 FR 60913
DIBROMOCHLORO-PROPANE	96-12-8	N	Y									0.2	—
DI-N-BUTYL PHTHALATE	84-74-2	Y	N					34,000.	154,000.	2,700.	12,000.		57 FR 60913
DICHLOROBENZENE, 1,2-	95-50-1	Y	N							2,700.	17,000.	600.	57 FR 60913
DICHLOROBENZENE, 1,3-	541-73-1	Y	N							400.	2,600.	600.	57 FR 60913
DICHLOROBENZENE, 1,4-	106-46-7	Y	N							400.	2,600.	75.	57 FR 60913
DICHLOROBENZENES	25321-22-6	N	N	*1,120.	*763.	*1,970.		400	2,600.				45 FR 79328
DICHLOROBENZIDINE, 3,3-	91-94-1	Y	Y					0 0103	0 0204	0.04	0.077		57 FR 60913
DICHLOROBROMO-METHANE	75-27-4	Y	Y							0.27	2.2		57 FR 60911
DICHLOROETHANE, 1,2-	107-06-2	Y	Y	*118,000.	*20,000.	*113,000		0 94	243	0.38	99.	5.0	57 FR 60912
DICHLOROETHYLENES	25323-30-3	N	Y	*11,600		*224,000.							45 FR 79332
DICHLOROETHYLENE, 1,1-	75-35-4	Y	Y					0 033	1 85	0 057	3.2	7.0	57 FR 60912
DICHLOROETHYLENE, cis-1,2-	—	N	N									70	—
DICHLOROETHYLENE, trans, 1,2-	156-60-5	Y	N							700		100.	57 FR 60890
DICHLOROPHENOL, 2,4-	120-83-2	Y	N	*2,020.	*365.			3090		93	790.		57 FR 60912
DICHLOROPROPANE	26638-19-7	N	N	*23,000.	*5,700.	*10,300.	*3,040.					5.0	45 FR 79333
DICHLOROPROPANE, 1,2-	78-87-5	Y	Y							0 52	39.	5 0	57 FR 60890
DICHLOROPROPENE	26952-23-8	N	N	*6,060	*244.	*790		87.	14,100				45 FR 79333
DICHLOROPROPYLENE, 1,3-	542-75-6	Y	N							10.	1,700.		57 FR 60912

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DIELDRIN	60-57-1	Y	Y	2.5	0.0019	0.71	0.0019	0.000071	0 00076	0.00014	0.00014		57 FR 60914
DIETHYL PHTHALATE	84-66-2	Y	N					350,000.	1,800,000.	23,000.	120,000.		57 FR 60913
DIMETHYL PHENOL, 2,4-	105-67-9	Y	N	*2,120.						540.	2,300.		57 FR 60890
DIMETHYL PHTHALATE	131-11-3	Y	N					313,000	2,900,000.				45 FR 79339
DINITROPHENOL, 2,4-	51-28-5	Y	N							70.	14,000.		57 FR 60912
DINITROPHENOL	25550-58-7	Y	N					70.	14,300.				45 FR 79337
DINITROTOLUENE, 2,4-	121-14-2	Y	Y	*330.	*230.	*590.	*370.	0.11	9.1				45 FR 79333
DINITRO-O-CRESOL, 2,4-**	534-52-1	Y	N					13 4	765.				45 FR 79333
DIOXIN (2,3,7,8 – TCDD)	1746-01-6	Y	Y	* <0.01	* <0.00001			0.000000013	0.000000014				49 FR 5831
DIPHENYLHYDRAZINE, 1,2-	122-66-7	Y	Y	*270.				0.042	0.56	0.041	0.54		57 FR 60913
DI-2-ETHYLHEXYL PHTHALATE	117-81-7	Y	Y	*2,100.	*160.			15,000.	50,000.	1.8	5.9		45 FR 79339
ENDOSULFAN	115-29-7	N	N	0.22	0.056	0.034	0 0087	74	159.				45 FR 79334
ENDOSULFAN SULFATE	1031-07-8	Y	N							0.93	2.0		57 FR 60915
ENDOSULFAN-ALPHA	959-98-8	Y	N	0.22	0.056	0 034	0 0087			0.93	2.0		57 FR 60914
ENDOSULFAN-BETA	33213-65-9	Y	N	0.22	0.056	0.034	0.0087			0.93	2.0		57 FR 60914
ENDRIN	72-20-8	Y	N	0.18	0.0023	0.037	0.0023	1 0		0.76	0.81		57 FR 60915
ENDRIN ALDEHYDE	7421-93-4	Y	N							0.76	0.81		57 FR 60915
ETHER, BIS (2-CHLOROETHYL)	111-44-4	Y	Y					0.03	1.36	0.031	1.4		57 FR 60913
ETHER, BIS (2-CHLOROISOPROPYL)	108-60-1	Y	N					34 7	4360.	1,400.	170,000.		57 FR 60913
ETHER, BIS (CHLOROMETHYL)	542-88-1	N	Y					0.0000038	0 00184				45 FR 79330
ETHYLBENZENE	100-41-4	Y	N	*32,000.		*430		1,400.	3280	3,100.	29,000.	700.	57 FR 60912
ETHYLENE DIBROMIDE	106-93-4	N	Y									0 05	—
FLUORANTHENE	206-44-0	Y	N	*3,980		*40.	*16.	42.	54	300.	370.		57 FR 60848
FLUORENE	86-73-7	Y	Y							0.0028	0.031		—
GASES, TOTAL DISSOLVED	—	N	N	NARRATIVE STATEMENT— SEE DOCUMENT									RB
GROSS ALPHA PARTICLE ACTIVITY	—	N	Y									15 pCi/L	—
GUTHION	86-50-0	N	N		0.01		0.01						RB
HALOETHERS	—	N	N	*360.	*122.								45 FR 79334
HALOMETHANES	—	N	Y	*11,000.		*12,000.	*6,400	0.19	15 7				45 FR 79334
HEPTACHLOR	76-44-8	Y	Y	0.52	0 0038	0 053	0.0036	0 00028	0 00029	0.00021	0.00021	0.4	45 FR 79335

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HEPTACHLOR EPOXIDE	1024-57-3	Y	Y	0.52	0.0038	0.053	0.0036			0.00010	0.00011	0.2	45 FR 79335
HEXACHLOROBENZENE	118-74-1	Y	Y	/p/6.0	/p/3.68			0.00072	0.00074	0.00075	0.00077		57 FR 60912
HEXACHLOROBUTADIENE	87-68-3	Y	Y	*90.	*9.3	*32.		0 45	50.	0.44	50.		45 FR 79335
HEXACHLOROCYCLO-HEXANE (LINDANE)	58-89-9	Y	Y	2.0	0.08	0.16		0.0186	0.0625	0.019	0.063	0 2	45 FR 79335
HEXACHLOROCYCLO-HEXANE—ALPHA	319-84-6	Y	Y					0.0092	0.031	0.0039	0.013		57 FR 60914
HEXACHLOROCYCLO-HEXANE — BETA	319-85-7	Y	Y					0.0163	0.0547	0.014	0.046		57 FR 60914
HEXACHLOROCYCLO-HEXANE — GAMMA	58-89-9	Y	Y	2.0	0.08	0.16		0.0186	0.0625	0.019	0.063	0.2	57 FR 60914
HEXACHLOROCYCLO-HEXANE—TECHNICAL	319-86-8	N	Y					0.0123	0.0414				45 FR 79335
HEXACHLOROCYCLO-PENTADIENE	77-47-4	Y	N	*7.0	*5.2	*7.0		206.		240.	17,000.		57 FR 60914
HEXACHLOROETHANE	67-72-1	Y	Y	*980.	*540.	*940.		1.9	8.74	1.9	8.9		57 FR 60914
INDENO(1,2,3-CD)PYRENE	193-39-5	Y	Y							0.0028	0.0311		57 FR 60914
IRON	7439-89-6	N	N		1,000.			300.					RB
ISOPHORONE	78-59-1	Y	N	*117,000.		*12,900.		5,200	520,000	8.4	600.		57 FR 60914
LEAD	7439-92-1	Y	N	82.+	3.2+	220.	8.5	50.				/p/5.0	57 FR 60914
MALATHION	121-75-5	N	N		0.1		0 1						RB
MANGANESE	7439-96-5	N	N					50.	100				RB
MERCURY	7439-97-6	Y	N	2.4	0.012	2.1	0.025	0.144	0.146	0.14	0.15	2.0	50 FR 30791
METHOXYCHLOR	72-43-5	N	N		0.03		0.03	100				40.	RB
METHYL BROMIDE	74-83-9	Y	N							48.	4,000.		57 FR 60912
METHYL CHLORIDE	74-87-3	Y	Y										57 FR 60848
METHYLENE CHLORIDE	75-09-2	Y	Y							4.7	1,600.		57 FR 60912
MIREX	2385-85-5	N	N		0.001		0.001						RB
NAPHTHALENE	91-20-3	Y	N	*2,300.	*620.	*2,350.							45 FR 79337
NICKEL	7440-02-0	Y	N	1,400.+	160.+	75.	8.3	13.4	100	610.	4,600.		57 FR 60911
NITRATES	14797-55-8	N	N					10,000				10,000.	RB
NITRITE	—	N	N									1,000.	—
NITROBENZENE	98-95-3	Y	N	*27,000.		*6,680		19,800.		17.	1,900.		57 FR 60914
NITROPHENOLS	—	Y	N	*230.	*150.	*4,850							45 FR 79337
NITROSAMINES	35576-91-1	Y	Y	*5,850		*3,300,000							45 FR 79337

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NITROSODIBUTYLAMINE, N-	924-16-3	N	Y					0.0064	0.587				45 FR 79338
NITROSODIETHYLAMINE, N-	55-18-5	N	Y					0.0008ng/L	0.0012				45 FR 79338
NITROSODIMETHYLAMINE, N-	62-75-9	Y	Y					0.0014	16.	0.00069	8.1		57 FR 60914
NITROSODIPHENYLAMINE, N-	86-30-6	Y	Y					4.9	16.1	5 0	16.		57 FR 60914
NITROSOPYROLIDINE, N-	930-55-2	N	Y					0.016	91.9				45 FR 79338
N-NITROSODI-N-PROPYLAMINE	621-64-7	Y	Y							0.005	1.4		57 FR 60890
OIL AND GREASE	—	N	N	NARRATIVE STATEMENT — SEE DOCUMENT									RB
OXYGEN DISSOLVED	7782-44-7	N	N	WARMWATER AND COLDWATER CRITERIA MATRIX — SEE DOCUMENT									51 FR 22978
PARATHION	56-38-2	N	N	0.065	0.013								51 FR 43667
PCBs	1336-36-3	Y	Y	2.0	0.014	10.	0 03	0.000079	0 000079			0 5	45 FR 79339
PCB-1016	12674-11-2	Y	Y							0.000044	0.000045		57 FR 60915
PCB-1221	11104-28-2	Y	Y							0.000044	0.000045		57 FR 60915
PCB-1232	11141-16-5	Y	Y							0.000044	0.000045		57 FR 60915
PCB-1242	5346-92-19	Y	Y							0.000044	0.000045		57 FR 60915
PCB-1248	12672-29-6	Y	Y							0.000044	0.000045		57 FR 60915
PCB-1254	11097-69-1	Y	Y							0.000044	0.000045		57 FR 60915
PCB-1260	11096-82-5	Y	Y							0 000044	0.000045		57 FR 60915
PENTACHLOROETHANE	76-01-7	N	N	*7,240.	*1,100.	*390.	*281.						45 FR 79328
PENTACHLOROBENZENE	608-93-5	N	N					74.	85.				45 FR 79327
PENTACHLOROPHENOL	87-86-5	Y	N	***20.	***13.	13.	7.9	1010		0.28	8.2	/p/1.0	57 FR 60912
pH	—	N	N		6.5-9		6.5-8.5	5-9					RB
PHENANTHRENE	85-01-8	Y	Y	/p/30	/p/6.3	/p/7.7	/p/4.6						57 FR 60848
PHENOL	108-95-2	Y-	N	*10,200.	*2,560.	*5,800.		3,500.		21,000.	4,600,000.		57 FR 60912
PHOSPHORUS ELEMENTAL	7723-14-0	N	N				0.1						RB
PHTHALATE ESTERS	—	Y	N										
POLYNUCLEAR AROMATIC HYDROCARBONS	—	Y	Y			*300.		0 0028	.0311				45 FR 79339
PYRENE	129-00-0	Y	Y							0.0028	0.0311		—
RADIUM 226/228	—	N	Y									5 pCi/L	—
SELENIUM	7782-49-2	Y	N	20.	5.0	300.	71.	10				50.	57 FR 60911
SILVER	7440-22-4	Y	N	4 1+/p/0.92		0.12	2.3/p/7 2	/p/0.92	50				57 FR 60911

	CAS#	PRIORITY POLLUTANT	CARCINOGEN	FRESH ACUTE CRITERIA	FRESH CHRONIC CRITERIA	SALTWATER ACUTE CRITERIA	SALTWATER CHRONIC CRITERIA	HUMAN HEALTH (10-6 RISK LEVEL FOR CARCINOGENS) PUBLISHED CRITERA WATER & ORGANISMS	PUBLISHED CRITERA ORGANISMS ONLY	RECALCULATED VALUES using IRIS, as of 9/90 WATER & ORGANISMS	RECALCULATED VALUES using IRIS, as of 9/90 ORGANISMS ONLY	DRINKING WATER MCL	CRITERIA FEDERAL REGISTER NOTICE
SOLIDS DISSOLVED AND SALINITY	—	N	N					250,000.					RB
SOLIDS SUSPENDED AND TURBIDITY	—	N	N	NARRATIVE STATEMENT — SEE DOCUMENT									RB
STYRENE	100-42-5	N	Y									100.	—
SULFIDE-HYDROGEN SULFIDE	7783-06-4	N	N		2.0		2.0						RB
TEMPERATURE	—	N	N	SPECIES DEPENDENT CRITERIA — SEE DOCUMENT									RB
TETRACHLOROBENZENE, 1,2,4,5-	95-94-3	N	N					38.	48.				45 FR 79327
TETRACHLOROETHANE, 1,1,2,2-	79-34-5	Y	Y		*2,400.	*9,020.		0.17	10.7	0.17	11.		57 FR 60912
TETRACHLOROETHANES	25322-20-7	Y	N	*9,320									45 FR 79328
TETRACHLOROETHYLENE	127-18-4	Y	Y	*5,280	*840.	*10,200.	*450.	0.8	8.85			5.0	45 FR 79340
TETRACHLOROPHENOL, 2,3,5,6-	935-95-5	N	N			*440.							45 FR 79329
THALLIUM	7440-28-0	Y	N	*1,400.	*40.	*2,130.		13.	48.	1.7	6.3		57 FR 60913
TOLUENE	108-88-3	Y	N	*17,500.		*6,300.	*5,000.	14,300.	424,000.	6,800.	200,000.	1,000.	57 FR 60912
TOXAPHENE	8001-35-2	Y	Y	0.73	0.0002	0.21	0.0002	0.00071	0.00073	0.00073	0.00075	/p/5.0	57 FR 60915
TRICHLORINATED ETHANES	25323-89-1	Y	Y	*18,000.									45 FR 79328
TRICHLOROETHANE, 1,1,1-	71-55-6	Y	N			*31,200.		18,400.	1,030,000.			200.	57 FR 60848
TRICHLOROETHANE, 1,1,2-	79-00-5	Y	Y		*9,400.			0.6	41.8	0.60	42.		57 FR 60912
TRICHLOROETHYLENE	79-01-6	Y	Y	*45,000.	*21,900.	*2,000.		2.7	80.7	2.7	81.	5.0	45 FR 79341
TRICHLOROPHENOL, 2,4,5-	95-95-4	N	N	/p/100	/p/63	/p/240	/p/11	2,600.					45 FR 79329
TRICHLOROPHENOL, 2,4,6-	88-06-2	Y	Y		*970.			1.2	3.6	2.1	6.5		57 FR 60912
VINYL CHLORIDE	75-01-4	Y	Y					2.0	525.			2.0	45 FR 79341
XYLENES	—	N	N									10,000.	—
ZINC	7440-66-6	Y	N	120.+	110.+	95.	86.						52 FR 6214

+ = Hardness dependent criteria (100 mg/L $CaCO_3$ used)

* = Insufficient data to develop criteria. Value presented is the L.O.E.L. (Lowest Observed Effect Level)

** = The preferred chemical name for 2,4 Dinitro-o-cresol listed in 45 FR 79333 is 4,6 – Dinitro-o-cresol

*** = pH dependent criteria (7.8 pH used)

MCL= Maximum contaminant level (only for listed chemicals)

RB = Red Book

/p/ = Proposed criterion

Y = Yes

N = No

Marine criteria for lead reflect values updated after 1984.

Note: This chart is for general information. Please use criteria documents or detailed summaries in Quality Criteria for Water 1992 for regulatory purposes.

CONTENTS

*Entry not listed on 1990 Water Quality Chart.

ACENAPHTHENE

83-32-9

CRITERIA

Aquatic Life

The available data for acenaphthene indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 1,700 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of acenaphthene to sensitive freshwater aquatic animals, but toxicity to freshwater algae occurs at concentrations as low as 520 µg/L.

The available data for acenaphthene indicate that acute and chronic toxicity to saltwater aquatic life occurs at concentrations as low as 970 and 710 µg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested. Toxicity to algae occurs at concentrations as low as 500 µg/L.

Human Health

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60890). Recalculated IRIS values for acenaphthene are 1,200 µg/L for ingestion of contaminated water and organisms and 2,700 µg/L for ingestion of contaminated aquatic organisms only.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60890, December 22, 1993)
See Appendix C for Human Health Methodology.

ACROLEIN

107-02-8

CRITERIA

Aquatic Life

The available data for acrolein indicate that acute and chronic toxicity to freshwater aquatic life occurs at concentrations as low as 68 and 21 μg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested.

The available data for acrolein indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 55 μg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of acrolein to sensitive saltwater aquatic life.

Human Health

For the protection of human health from the toxic properties of acrolein ingested through water and contaminated aquatic organisms, the ambient water criterion is 320 μg/L.

For the protection of human health from the toxic properties of acrolein ingested through contaminated aquatic organisms alone, the ambient water criterion is 780 μg/L.

(45 F.R. 79318, November 28, 1980)
See Appendix C for Human Health Methodology.

ACRYLONITRILE

107-13-1

CRITERIA

Aquatic Life

The available data for acrylonitrile indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 7,550 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No definitive data are available concerning the chronic toxicity of acrylonitrile to sensitive freshwater aquatic life, but mortality occurs at concentrations as low as 2,600 µg/L, with a fish species exposed for 30 days.

Only one saltwater species has been tested with acrylonitrile, therefore no statement can be made concerning acute or chronic toxicity.

Human Health

For the maximum protection of human health from the potential carcinogenic effects resulting from exposure to acrylonitrile through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentrations should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels that may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}.

Published human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92. Recalculated IRIS values for acrylonitrile are 0.059 µg/L for ingestion of contaminated water and organisms and 0.66 µg/L for ingestion of contaminated aquatic organisms only. IRIS values are based on a 10^{-6} risk level for carcinogens.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60848, December 22, 1992)
See Appendix C for Human Health Methodology.

AESTHETIC QUALITIES

CRITERIA

All waters free from substances attributable to wastewater or other discharges that

1. settle to form objectionable deposits;
2. float as debris, scum, oil, or other matter to form nuisances;
3. produce objectionable color, odor, taste, or turbidity;
4. injure or are toxic or produce adverse physiological responses in humans, animals, or plants; and
5. produce undesirable or nuisance aquatic life.

(Quality Criteria for Water, July 1976) PB-263943
See Appendix D for Methodology.

*ALDRIN

309-00-2

CRITERIA

Aquatic Life

Not to exceed 3.0 μg/L in fresh water or 1.3 μg/L in salt water.

For freshwater aquatic life, the concentration of aldrin should not exceed 3.0 μg/L at any time. No data are available concerning the chronic toxicity of aldrin to sensitive freshwater aquatic life.

For saltwater aquatic life, the concentration of aldrin should not exceed 1.3 μg/L at any time. No data are available concerning the chronic toxicity of aldrin to sensitive saltwater aquatic life.

Human Health

For the maximum protection of human health from the potential carcinogenic effects of exposure to aldrin through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels that may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}.

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60848, December 22, 1992). Recalculated IRIS values for aldrin are 0.00013 μg/L for ingestion of contaminated water and organisms and 0.00014 μg/L for ingestion of contaminated aquatic organisms only. IRIS values are based on a 10^{-6} risk level for carcinogens.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60848, December 22, 1992)

See Appendix B for Aquatic Life Methodology.
See Appendix C for Human Health Methodology.

*Indicates suspended, canceled, or restricted by U.S. EPA Office of Pesticides and Toxic Substances

ALKALINITY

CRITERIA

20 mg/L, or more as $CaCO_3$, for freshwater aquatic life except where natural concentrations are less.

Introduction

Expressed commonly as milligrams per liter of calcium carbonate, alkalinity is the sum total of components in the water that tend to elevate the pH of the water above a value of about 4.5. It is measured by titration with standardized acid to a pH value of about 4.5. Alkalinity, therefore, is a measure of water's buffering capacity, and since pH has a direct effect on organisms as well as an indirect effect on the toxicity of certain other pollutants in the water, the buffering capacity is important to water quality. Examples of commonly occurring materials in natural waters that increase the alkalinity are carbonates, bicarbonates, phosphates, and hydroxides.

Rationale

The alkalinity of water used for municipal water supplies is important because it affects the amount of chemicals that need to be added to accomplish coagulation, softening, and control of corrosion in distribution systems. The alkalinity of water assists in the neutralization of excess acid produced during the addition of such materials as aluminum sulfate during chemical coagulation. Waters having sufficient alkalinity do not have to be supplemented with artificially added materials to increase the alkalinity. Alkalinity resulting from naturally occurring materials such as carbonate and bicarbonate is not considered a health hazard in drinking water supplies, per se, and naturally occurring maximum levels up to approximately 400 mg/L as calcium carbonate are not considered a problem to human health.

Alkalinity is important for fish and other aquatic life in freshwater systems because it buffers pH changes that occur naturally as a result of photosynthetic activity of the chlorophyll-bearing vegetation. Components of alkalinity such as carbonate and bicarbonate will complex some toxic heavy metals and reduce their toxicity markedly. For these reasons, in 1968 the National Technical Advisory Committee recommended a minimum alkalinity of 20 mg/L. The subsequent 1974 National Academy of Sciences (NAS) report recommended that natural alkalinity not be reduced by more than 25 percent but did not place an absolute minimal value for it. The use of the 25 percent reduction avoids the problem of establishing standards on waters where natural alkalinity is at or below 20 mg/L. For such waters, alkalinity should not be further reduced.

The NAS Report recommends that adequate amounts of alkalinity be maintained to buffer the pH within tolerable limits for marine waters. It has been noted as a correlation that productive waterfowl habitats are above 25 mg/L with higher alkalinities resulting in better waterfowl habitats.

Excessive alkalinity can cause problems for swimmers by altering the pH of the lacrimal fluid around the eye, causing irritation.

For industrial water supplies high alkalinity can be damaging to industries involved in food production, especially those in which acidity accounts for flavor and stability, such as the carbonated beverages. In other instances, alkalinity is desirable because water with a high alkalinity is much less corrosive.

A brief summary of maximum alkalinities accepted as a source of raw water by industry is included in Table 1. The concentrations listed in the table are for water prior to treatment and thus are only desirable ranges and not critical ranges for industrial use.

Table 1.*—Maximum alkalinity in waters used as a source of supply prior to treatment.

INDUSTRY	ALKALINITY MG/L AS $CACO_3$
Steam generation boiler makeup	350
Steam generation cooling	500
Textile mill products	50–200
Paper and allied products	75–150
Chemical and allied products	500
Petroleum refining	500
Primary metals industries	200
Food canning industries	300
Bottled and canned soft drinks	85

Source: National Academy of Sciences (1974).

The effect of alkalinity in water used for irrigation may be important in some instances because it may indirectly increase the relative proportion of sodium in soil water. As an example, when bicarbonate concentrations are high, calcium and magnesium ions that are in solution precipitate as carbonates in the soil water as the water becomes more concentrated through evaporation and transpiration. As the calcium and magnesium ions decrease in concentration, the percentage of sodium increases and results in soil and plant damage. Alkalinity may also lead to chlorosis in plants because it causes the iron to precipitate as a hydroxide. Hydroxyl ions react with available iron in the soil water and make the iron unavailable to plants. Such deficiencies induce chlorosis and further plant damage. Usually alkalinity must exceed 600 mg/L before such affects are noticed, however.

(Quality Criteria for Water, July 1976) PB-263943
See Appendix D for Methodology.

ALUMINUM

7429-90-5

CRITERIA

Aquatic Life

When the pH is between 6.5 and 9.0, four-day average concentration of aluminum should not exceed 87 µg/L. One-hour average concentration of aluminum should not exceed 750 µg/L for freshwater aquatic life.

Summary

Acute tests have been conducted on aluminum at pH betwen 6.5 and 9.0 with freshwater species in 14 genera. In many tests, less than 50 percent of the organisms were affected at the highest concentration tests. Both ceriodaphnids and brook trout were affected at concentrations below 4,000 µg/L, whereas some other fish and invertebrate species were not affected by 45,000 µg/L. Some researchers found that the acute toxicity of aluminum increased with pH, whereas others found the opposite to be true.

Three studies have been conducted on the chronic toxicity of aluminum to aquatic animals. The chronic values for *Daphnia magna, Ceriodaphnia dubia,* and the fathead minnow were 742.2, 1,908, and 3,288 µg/L, respectively. The diatom, *Cyclotella meneghiniana,* and the green alga, *Selenastrum capricornutum,* were affected by concentrations of aluminum in the range of 400 to 900 µg/L. Bioconcentration factors from 50 to 231 were obtained in tests with young brook trout. At a pH of 6.5 to 6.6, 169 µg/L caused a 24 percent reduction in the growth of young brook trout and 174 µg/L killed 58 percent of the exposed striped bass.

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, when the pH is between 6.5 and 9.0, if the four-day average concentration of aluminum does not exceed 87 µg/L more than once every three years on the average and if the one-hour average concentration does not exeed 750 µg/L more than once every three years on the average.

(53 F.R. 33178, August 30, 1988)

See Appendix A for Aquatic Life Methodology.

AMMONIA

7664-41-7

SUMMARY

Fresh Water

All concentrations used herein are expressed as un-ionized ammonia (NH_3) because NH_3, not the ammonium ion (NH_4+), has been demonstrated to be the principal toxic form of ammonia. The data used in deriving criteria are predominantly from flow through tests that measured ammonia concentrations.

Ammonia was reported to be acutely toxic to freshwater organisms at concentrations (uncorrected for pH) ranging from 0.53 to 22.8 mg/L NH_3 for 19 invertebrate species representing 14 families and 16 genera and from 0.083 to 4.60 mg/L NH_3 for 29 fish species from 9 families and 18 genera. Among fish species, reported 96-hour LC50 ranged from 0.083 to 1.09 mg/L for salmonids and from 0.14 to 4.60 mg/L NH_3 for nonsalmonids. Reported data from chronic tests on ammonia with two freshwater invertebrate species, both daphnids, showed effects at concentrations (uncorrected for pH) ranging from 0.304 to 1.2 mg/L NH_3 and with 9 freshwater fish species, from 5 families and 7 genera, ranging from 0.0017 to 0.612 mg/L NH_3.

Concentrations of ammonia are acutely toxic to fishes and can cause loss of equilibrium, hyperexcitability, increased breathing, cardiac output and oxygen uptake, and, in extreme cases, convulsions, coma, and death. At lower concentrations, ammonia has many effects on fishes, including a reduction in hatching success, reduction in growth rate and morphological development, and pathologic changes in tissues of gills, livers, and kidneys.

Several factors have been shown to modify acute NH_3 toxicity in fresh water. Some factors alter the concentration of un-ionized ammonia in the water by affecting the aqueous ammonia equilibrium, and some factors affect the toxicity of un-ionized ammonia itself, either by ameliorating or exacerbating the effects of ammonia. Factors that have been shown to affect ammonia toxicity include dissolved oxygen concentration, temperature, pH, previous acclimation to ammonia, fluctuating or intermittent exposures, carbon dioxide concentration, salinity, and the presence of other toxicants.

The most well-studied of these factors is pH: the acute toxicity of NH_3 has been shown to increase as pH decreases. Sufficient data exist from toxicity tests conducted at different pH values to formulate a mathematical expression to describe pH-dependent, acute NH_3 toxicity. The very limited amount of data regarding effects of pH on chronic NH_3 toxicity also indicates increasing NH_3 toxicity with decreasing pH, but the data are insufficient to derive a broadly applicable toxicity/pH relationship. Data on temperature effects on acute NH_3 toxicity are limited and somewhat variable, but indications are that NH_3 toxicity to fish is greater as temperature decreases. No information is available regarding temperature effects on chronic NH_3 toxicity.

Examination of pH and temperature-corrected acute NH_3 toxicity values among species and genera of freshwater organisms showed that invertebrates are generally more tolerant than fishes, a notable exception being the fingernail clam. No clear trend exists among groups of fish; the several most sensitive tested species and genera include representatives from diverse families (Salmonidae, Cyprinidae, Percidae, and Centrarchidae). Available chronic toxicity data for freshwater organisms also indicate invertebrates (cladocerans, one insect species) to be more tolerant than fishes, again with the exception of the fingernail clam. When corrected for the presumed effects of temperature and pH, chronic toxicity values also show no clear trend among groups of fish, the most sensitive species including representatives from five families (Salmonidae, Cyprinidae, Ictaluridae, Centrarchidae, and Catostomidae) and having chronic values ranging by not much more than a factor or two. The range of acute-chronic ratios for 10 species from 6 families was 3 to 43; acute-chronic ratios were higher for the species having chronic tolerance below the median.

Available data indicate that differences in sensitivities between warmwater and coldwater families of aquatic organisms are inadequate to warrant discrimination in the national ammonia criterion between bodies of water with "warmwater" and "coldwater" fishes; rather, effects of organism sensitivities on the criterion are most appropriately handled by site-specific criteria derivation procedures.

Data for concentrations of NH_3 toxic to freshwater phytoplankton and vascular plants, although limited, indicate that freshwater plant species are appreciably more tolerant to NH_3 than are invertebrates or fishes. The ammonia criterion appropriate for the protection of aquatic animals, therefore, will probably sufficiently protect plant life.

Salt Water

In aqueous solutions, the ammonium ion dissociates to un-ionized ammonia and the hydrogen ion. The equilibrium equation can be written

Equation 1

$$H_2O + NH_4+ \Leftrightarrow NH_3 + H_3O^+$$

The total ammonia concentration is the sum of NH_3 and NH_4+.

The toxicity of aqueous ammonia solutions to aquatic organisms is primarily attributable to the un-ionized form, the ammonium ion being less toxic. It is necessary, therefore, to know the percentage of total ammonia in the un-ionized form in order to establish the corresponding total ammonia concentration toxic to aquatic life. The percentage of un-ionized ammonia (UIA) can be calculated from the solution pH and pK_a, the negative log of stoichiometric dissociation,

Equation 2

$$\% \text{ UIA} = 100\,[1 + 10^{(pK_a - pH)}]^{-1}$$

The stoichiometric dissociation constant is defined

Equation 3

$$k_a = \frac{[NH_3]\,[H^+]}{[NH_4^+]}$$

where the brackets represent molal concentrations. K_a is a function of the temperature and ionic strength of the solution.

Whitfield (1974) developed theoretical models to determine the pK_a of the ammonium ion in seawater. He combined his models with the infinite dilution data of Bates and Pinching (1949) to define general equations for the PK_a of ammonium ion as a function of salinity and temperature.

Whitfield's models allow reasonable approximations of the percent of un-ionized ammonia in sea water and have been substantiated experimentally. Hampson's (1977) program for Whitfield's full seawater model has been used to calculate the un-ionized ammonia fraction of measured total ammonia concentrations in toxicity studies conducted by EPA and also in deriving most other acute and chronic ammonia values that contribute to the criteria. The equations for this model are

Equation 4

$$\% \text{ UIA} = 100\,[1 + 10\,(X + 0.0324\,(298\text{-}T) + 0.0415\,P/T - pH]^{-1}$$

where

P = 1 AIM for all toxicity testing reported to date
T = temperature (K)
X = pk^S_a or the stoichiometric acid hydrolysis constant of ammonium ions in saline water based on I,

Equation 5

$$I = 19.9273\,S\,(1000\text{-}1.005109\,S)^{-1}$$

where

I = molal ionic strength of the sea water
S = salinity (g/kg)

The Hampson program calculates the value for I for the test salinity (Eq. 5), finds the corresponding pk^S_a, then calculates % UIA (Eq. 4).

The major factors influencing the degree of ammonia dissociation are pH and temperature. Both correlate positively with un-ionized ammonia. Salinity, the least influential of the three water quality factors that control the fraction of un-ionized ammonia, is inversely correlated.

NATIONAL CRITERIA

Fresh Water

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

(1) the one-hour* average concentration of un-ionized ammonia (in mg/L NH_3) (see Tables 1 and 2) does not exceed more often than once every three years on the average the numerical value given by 0.52/FT/FPH/2

where:

$$FT = \begin{cases} 10^{0.03(20\text{-}TCAP)} & ; TCAP \le T \le 30 \\ 10^{0.03(20\text{-}T)} & ; 0 \le T \le TCAP \end{cases}$$

*An averaging period of one hour may not be appropriate if excursions of concentrations to greater than 1.5 times the average occur during the hour; in such cases, a shorter averaging period may be needed.

Table 1.—One-hour average concentrations for ammonia with salmonids or other sensitive coldwater species present.*

pH	Temperature (°C) 0	5	10	15	20	25	30
	Un-ionized Ammonia (mg/L NH_3)						
6.50	0.0091	0.0129	0.0182	0.026	0.036	0.036	0.036
6.75	0.0149	0.021	0.030	0.042	0.059	0.059	0.059
7.00	0.023	0.033	0.046	0.066	0.093	0.093	0.093
7.25	0.034	0.048	0.068	0.095	0.135	0.135	0.135
7.50	0.045	0.064	0.091	0.128	0.181	0.181	0.181
7.75	0.056	0.080	0.113	0.159	0.22	0.22	0.22
8.00	0.065	0.092	0.130	0.184	0.26	0.26	0.26
8.25	0.065	0.092	0.130	0.184	0.26	0.26	0.26
8.50	0.065	0.092	0.130	0.184	0.26	0.26	0.26
8.75	0.065	0.092	0 130	0.184	0.26	0.26	0.26
9 00	0.065	0.092	0.130	0.184	0.26	0.26	0.26
	Total Ammonia (mg/L NH_3)						
6.50	35	33	31	30	29	20	14.3
6.75	32	30	28	27	27	18.6	13.2
7.00	28	26	25	24	23	16.4	11.6
7.25	23	22	20	19.7	19.2	13 4	9.5
7.50	17.4	16.3	15.5	14.9	14.6	10.2	7.3
7.75	12.2	11.4	10.9	10.5	10.3	7.2	5.2
8.00	8 0	7.5	7.1	6.9	6.8	4.8	3.5
8.25	4.5	4.2	4.1	4.0	3.9	2.8	2.1
8.50	2.6	2 4	2.3	2.3	2.3	1.71	1.28
8.75	1 47	1 40	1.37	1.38	1.42	1.07	0.83
9.00	0.86	0.83	0.83	0.86	0.91	0.72	0.58

*To convert these values to mg/L N, multiply by 0.822.

$$FPH = \begin{cases} 1 & 8 \le pH \le 9 \\ \dfrac{1 + 10^{7.4 - pH}}{1.25} & 6.5 \le pH \le 8 \end{cases}$$

TCAP = 20°C; salmonids or other sensitive coldwater species present

TCAP = 25°C; salmonids and other sensitive coldwater species absent

(2) the four-day average concentration of un-ionized ammonia (in mg/L NH_3) (see Tables 3 and 4) does not exceed, more often than once every three years on the average, the average** numerical value given by 0.80/FT/FPH/RATIO, where FT and FPH are as above and

$$RATIO = \begin{cases} 13.5 & 7.7 \le pH \le 9 \\ 20 \cdot \dfrac{10^{7.7 - pH}}{1 + 10^{7.4 - pH}} & 6.5 \le pH \le 7.7 \end{cases}$$

TCAP = 15°C; salmonids or other sensitive coldwater species present

TCAP = 20°C; salmonids and other sensitive coldwater species absent

**Because these formulas are nonlinear in pH and temperature, the criterion should be the average of separate evaluations of the formulas reflective of the fluctuations of flow, pH, and temperature within the averaging period; it is not appropriate in general to simply apply the formula to average pH, temperature, and flow.

Table 2.—One-hour average concentrations for ammonia with salmonids or other sensitive coldwater species absent.*

pH	Temperature (°C) 0	5	10	15	20	25	30
	Un-ionized Ammonia (mg/L NH_3)						
6.50	0.0091	0.0129	0.0182	0.026	0.036	0.051	0.051
6.75	0.0149	0.021	0.030	0.042	0.059	0.084	0.084
7.00	0.023	0.033	0.046	0.066	0.093	0.131	0.131
7.25	0.034	0.048	0.068	0.095	0.135	0.190	0.190
7.50	0.045	0.064	0.091	0.128	0.181	0.26	0.26
7.75	0.056	0.080	0.113	0.159	0.22	0.32	0.32
8.00	0.065	0.092	0.130	0.184	0.26	0.37	0.37
8.25	0.065	0.092	0.130	0.184	0.26	0.37	0.37
8.50	0.065	0.092	0.130	0.184	0.26	0.37	0.37
8.75	0.065	0.092	0.130	0 184	0.26	0.37	0.37
9.00	0.065	0.092	0.130	0.184	0.26	0.37	0.37
	Total Ammonia (mg/L NH_3)						
6.50	35	33	31	30	29	29	20
6.75	32	30	28	27	27	26	18.6
7.00	28	26	25	24	23	23	16.4
7.25	23	22	20	19.7	19.2	19.0	13.5
7.50	17.4	16.3	15.5	14.9	14.6	14.5	10.3
7.75	12.2	11.4	10.9	10.5	10.3	10.2	7.3
8.00	8.0	7.5	7.1	6.9	6.8	6.8	4.9
8.25	4.5	4.2	4.1	4.0	3.9	4.0	2.9
8 50	2.6	2.4	2.3	2.3	2.3	2.4	1 81
8.75	1.47	1 40	1.37	1.38	1 42	1.52	1.18
9.00	0.86	0.83	0 83	0.86	0.91	1.01	0.82

*To convert these values to mg/L N, multiply by 0.822.

The extremes for temperature (0°C and 30°C) and pH (6.5, 9) given in the above formulas are absolute. It is not permissible with current data to conduct any extrapolations beyond these limits. In particular, there is reason to believe that appropriate criteria at pH > 9 will be lower than the plateau between pH 8 and 9 given above.

Limited data exists on the effect of temperature on chronic toxicity. EPA will be conducting additional research on the effects of temperature on ammonia toxicity to fill perceived data gaps. Because of this uncertainty, additional site-specific information should be developed before these criteria are used in wasteload allocation modeling. For example, the chronic criteria tabulated for sites lacking salmonids are less certain at temperatures much below 20°C than those tabulated at temperatures near 20°C. Where the treatment levels needed to meet these criteria below 20°C may be substantial, use of site-specific criteria is strongly suggested. Development of such criteria should be based upon site-specific toxicity tests.

The recommended exceedence frequency of three years is the Agency's best scientific judgment of the average amount of time for an unstressed system to recover from a pollution event in which exposure to ammonia exceeds the criterion. A stressed system — for example, one in which several outfalls occur in a limited area — would be expected to require more

Table 3.—Four-day average concentrations for ammonia with salmonids or other sensitive coldwater species present.*

pH	Temperature (°C) 0	5	10	15	20	25	30
	Un-ionized Ammonia (mg/L NH_3)						
6.50	0.0008	0.0011	0.0016	0.0022	0.0022	0.0022	0.0022
6.75	0.0014	0.0020	0.0028	0.0039	0.0039	0.0039	0.0039
7.00	0.0025	0.0035	0.0049	0.0070	0.0070	0.0070	0.0070
7.25	0.0044	0.0062	0.0088	0.0124	0.0124	0.0124	0.0124
7.50	0.0018	0.0111	0.0156	0.022	0.022	0.022	0.022
7.75	0.0129	0.0182	0.026	0.036	0.036	0.036	0.036
8.00	0.0149	0.021	0.030	0.042	0.042	0.042	0.042
8.25	0.0149	0.021	0.030	0.042	0.042	0.042	0.042
8.50	0.0149	0.021	0.030	0.042	0.042	0.042	0.042
8.75	0.0149	0.021	0.030	0.042	0.042	0.042	0.042
9 00	0.0149	0.021	0.030	0.042	0 042	0.042	0 042
	Total Ammonia (mg/L NH_3)						
6.50	3.0	2.8	2 7	2.5	1 76	1.23	0.87
6 75	3.0	2.8	2.7	2.6	1 76	1.23	0.87
7.00	3 0	2.8	2 7	2.6	1.76	1.23	0.87
7.25	3.0	2.8	2.7	2.6	1.77	1.24	0.88
7.50	3.0	2.8	2.7	2.6	1.78	1.25	0.89
7.75	2.8	2.6	2.5	2 4	1.66	1.17	0.84
8.00	1.82	1.70	1.62	1.57	1.10	0.78	0.56
8.25	1.03	0.97	0.93	0.90	0.64	0.46	0.33
8.50	0.58	0.55	0.53	0.53	0.38	0.28	0.21
8.75	0.34	0.32	0.31	0.31	0.23	0.173	0.135
9.00	0.195	0.189	0.189	0.195	0.148	0.116	0.094

*To convert these values to mg/L N, multiply by 0.822.

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 selecting 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 design flow and 7Q5 or 7Q10 for the Criterion Continuous Concentration design flow in steady-state models for unstressed and stressed systems, respectively. The Agency acknowledges that the Criterion Continuous Concentration stream flow averaging period used for steady-state wasteload allocation modeling may be as long as 30 days in situations involving POTWs designed to remove ammonia where limited variability of effluent pollutant concentration and resultant concentrations in receiving waters can be demonstrated. In cases where low variability can be demonstrated, longer averaging periods for the ammonia Criterion Continuous Concentration (e.g., 30-day averaging periods) would be acceptable because the magnitude and duration of exceedences above the Criterion Continuous Concentration would be sufficiently limited.

Table 4.—Four-day average concentrations for ammonia with salmonids or other sensitive coldwater species absent.* **

pH	Temperature (°C) 0	5	10	15	20	25	30
	Un-ionized Ammonia (mg/L NH_3)						
6.50	0.0008	0.0011	0.0016	0.0022	0.0031	0.0031	0.0031
6.75	0.0014	0.0020	0.0028	0.0039	0.0055	0.0055	0.0055
7.00	0.0025	0.0035	0.0049	0.0070	0.0099	0.0099	0.0099
7.25	0.0044	0.0062	0.0088	0.0124	0.0175	0.0175	0.0175
7.50	0.0078	0.0111	0.0156	0.022	0.031	0.031	0.031
7.75	0.0129	0.0182	0.026	0.036	0.051	0.051	0.051
8.00	0.0149	0.021	0.030	0.042	0.059	0.059	0.059
8.25	0.0149	0.021	0.030	0.042	0.059	0.059	0.059
8.50	0.0149	0.021	0.030	0.042	0.059	0.059	0.059
8.75	0.0149	0.021	0.030	0.042	0.059	0.059	0.059
9.00	0.0149	0.021	0.030	0.042	0.059	0.059	0.059
	Total Ammonia (mg/L NH_3)						
6.50	3.0	2.8	2.7	2.5	2.5	1.73	1.23
6.75	3.0	2.8	2.7	2.6	2.5	1.74	1.23
7.00	3.0	2.8	2.7	2.6	2.5	1.74	1.23
7.25	3.0	2.8	2.7	2.6	2.5	1.75	1.24
7.50	3.0	2.8	2.7	2.6	2.5	1.76	1.25
7.75	2.8	2.6	2.5	2.4	2.3	1.65	1.18
8.00	1.82	1.70	1.62	1.57	1.55	1.10	0.79
8.25	1.03	0.97	0.93	0.90	0.90	0.64	0.47
8.50	0.58	0.55	0.53	0.53	0.53	0.39	0.29
8.75	0.34	0.32	0.31	0.31	0.32	0.24	0.190
9.00	0.195	0.189	0.189	0.195	0.21	0.163	0.133

***To convert these values to mg/L N, multiply by 0.822.**

****These values may be conservative; however, if a more refined criterion is desired, EPA recommends a site-specific criteria modification.**

These matters are discussed in more detail in EPA's "Technical Support Document for Water Quality-Based Toxics Control."

Salt Water

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 should not be affected unacceptably if the four-day average concentration of un-ionized ammonia does not exceed 0.035 mg/L more than once every three years on the average and if the one-hour average concentration does not exceed 0.233 mg/L more than once every three years on the average. Because sensitive saltwater animals appear to have narrow range of acute susceptibilities to ammonia, this criterion will probably be as protective as intended only when the magnitudes and/or durations of excursions are appropriately small.

Criteria concentrations based on total ammonia for the pH range of 7.0 to 9.0, temperature range of 0 to 35°C, and salinities of 10, 20 and 30 g/kg are provided in Tables 5 and 6. These values were calculated by Hampson's (1977) program of Whitfield's (1974) model for hydrolysis of ammonium ions in sea water (see original document).

Table 5.—Water quality criteria for saltwater aquatic life based on total ammonia (mg/L).

Criteria Maximum Concentrations

pH	Temperature (°C) 0	5	10	15	20	25	30	35
	Salinity = 10 g/kg							
7.0	270	191	131	92	62	44	29	21
7.2	175	121	83	58	40	40	19	13
7.4	110	77	52	35	25	25	12	8.3
7.6	69	48	33	23	16	16	7.7	5.6
7.8	44	31	21	15	10	10	5.0	3.5
8.0	27	19	13	9.4	6.4	6.4	3.1	2.3
8.2	18	12	8.5	5.8	4.2	4.2	2.1	1.5
8.4	11	7.9	5.4	3.7	2.7	2.7	1.4	1.0
8.6	7.3	5.0	3.5	2.5	1.8	1.8	0.98	0.75
8.8	4.6	3.3	2.3	1.7	1.2	1.2	0.71	0.56
9.0	2.9	2.1	1.5	1.1	0.85	0.85	0.52	0.44
	Salinity = 20 g/kg							
7.0	291	200	137	96	64	44	31	21
7.2	183	125	87	60	42	29	20	14
7.4	166	79	54	37	27	18	12	8.7
7.6	73	50	35	23	17	11	7.9	5.6
7.8	46	31	23	15	11	7.5	5.2	3.5
8.0	29	20	14	9.8	6.7	4.8	3.3	2.3
8.2	19	13	8.9	6.2	4.4	3.1	2.1	1.6
8.4	12	8.1	5.6	4.0	2.9	2.0	1.5	1.1
8.6	7.5	5.2	3.7	2.7	1.9	1.4	1.0	0.77
8.8	4.8	3.3	2.5	1.7	1.3	0.94	0.73	0.56
9.0	3.1	2.3	1.6	1.2	0.87	0.69	0.54	0.44
	Salinity = 30 g/kg							
7.0	312	208	148	102	71	48	33	23
7.2	196	135	94	64	44	31	21	15
7.4	125	85	58	40	27	19	13	9.4
7.6	79	54	37	25	21	12	8.5	6.0
7.8	50	33	23	16	11	7.9	5.4	3.7
8.0	31	21	15	10	7.3	5.0	3.5	2.5
8.2	20	14	9.6	6.7	4.6	3.3	2.3	1.7
8.4	12.7	8.7	6.0	4.2	2.9	2.1	1.6	1.1
8.6	8.1	5.6	4.0	2.7	2.0	1.4	1.1	0.81
8.8	5.2	3.5	2.5	1.8	1.3	1.0	0.75	0.58
9.0	3.3	2.3	1.7	1.2	0.94	0.71	0.56	0.46

In the Agency's best scientific judgment, the average amount of time aquatic ecosystems should be provided between excursions is three years. The ability of ecosystems to recover differ greatly.

Site-specific criteria may be established if adequate justification is provided. This site-specific criterion may include not only site-specific criteria concentrations, and mixing zone considerations, but also site-specific durations of averaging periods and site-specific frequencies of allowed exceedences.

Table 6.—Water quality criteria for saltwater aquatic life based on total ammonia (mg/L).

Criteria Continuous Concentrations

pH	Temperature (°C) 0	5	10	15	20	25	30	35
	Salinity = 10 g/kg							
7.0	41	29	20	14	9.4	6.6	4.4	3.1
7.2	26	18	12	8.7	5.9	4.1	2.8	2.0
7.4	17	12	7.8	5.3	3.7	2.6	1.8	1.2
7.6	10	7.2	5.0	3.4	2.4	1.7	1.2	0.84
7.8	6.6	4.7	3.1	2.2	1.5	1.1	0.75	0.53
8.0	4.1	2.9	2.0	1.4	0.97	0.69	0.47	0.34
8.2	2.7	1.8	1.3	0.87	0.62	0.44	0.31	0.23
8.4	1.7	1.2	0.81	0.56	0.41	0.29	0.21	0.16
8.6	1.1	0.75	0.53	0.37	0.27	0.20	0.15	0.11
8.8	0.69	0.50	0.34	0.25	0.18	0.14	0 11	0.08
9.0	0.44	0.31	0.23	0.17	0.13	0.10	0.08	0.07
	Salinity = 20 g/kg							
7.0	44	30	21	14	9.7	6.6	4.7	3.1
7.2	27	19	13	9.0	6.2	4.4	3.0	2.1
7.4	18	12	8.1	5.6	4.1	2.7	1.9	1.3
7.6	11	7.5	5.3	3.4	2.5	1.7	1.2	0.84
7.8	6.9	4.7	3.4	2.3	1.6	1.1	0.78	0.53
8.0	4.4	3.0	2.1	1.5	1.0	0.72	0.50	0.34
8.2	2.8	1.9	1.3	0.94	0.66	0.47	0.31	0.24
8.4	1.8	1.2	0.84	0.59	0.44	0.30	0.22	0.16
8.6	1.1	0.78	0.56	0.41	0.28	0.20	0.15	0.12
8.8	0.72	0.50	0.37	0.26	0.19	0.14	0.11	0.08
9.0	0.47	0.34	0.24	0.18	0.13	0.10	0.08	0.07
	Salinity = 30 g/kg							
7.0	47	31	22	15	11	7.2	5.0	3.4
7.2	29	20	14	9.7	6.6	4.7	3.1	2.2
7.4	19	13	8.7	5.9	4.1	2.9	2.0	1.4
7.6	12	8.1	5.6	3.7	3.1	1.8	1.3	0.90
7.8	7.5	5.0	3.4	2.4	1.7	1.2	0.81	0.56
8.0	4.7	3.1	2.2	1.6	1.1	0.75	0.53	0.37
8.2	3.0	2.1	1.4	1.0	0.69	0.50	0.34	0.25
8.4	1.9	1.3	0.90	0.62	0.44	0.31	0.23	0.17
8.6	1.2	0.83	0.59	0.41	0.30	0.22	0.16	0.12
8.8	0.78	0.53	0.37	0.27	0.20	0.15	0.11	0.09
9.0	0.50	0.34	0.26	0.19	0.14	0.11	0.08	0.07

Use of criteria for developing water quality-based permit limits and for designing waste treatment facilities requires selecting an appropriate wasteload allocation model. Dynamic models are preferred for the application of these criteria. Limited data or other considerations might make their use impractical, causing reliance on a steady-state model.

Implementation

Water quality standards for ammonia developed from these criteria should specify use of environmental monitoring methods that are comparable to the analytical methods employed to generate the toxicity data base. Total ammonia may be measured using an automated idophenol blue method,

such as described by Technicon Industrial Systems (1973) or U.S. EPA (1979) method 350.1. Un-ionized ammonia concentrations should be calculated using the dissociation model of Whitfield (1974) as programmed by Hampson (1977). This program was used to calculate most of the un-ionized values for saltwater organisms. Accurate measurement of sample pH is crucial in the calculation of the un-ionized ammonia fraction. The following equipment and procedures were used by EPA in the ammonia toxicity studies to enhance the precision of pH measurements in salt water. The pH meter reported two decimal places. A Ross electrode with ceramic junction was used because of its rapid response time; an automatic temperature compensation probe provided temperature correction. Note that the responsiveness of a new electrode may be enhanced by holding it in seawater for several days prior to use. Two National Institute of Standards and Technology buffer solutions for calibration preferred for their stability were (1) potassium hydrogen phthalate (pH 4.00) and (2) disodium hydrogen phosphate (pH 7.4). For overnight or weekend storage, the electrode was held in salt water, leaving the fill hole open. For daily use, the outer half-cell was filled with electrolyte to the fill hole and the electrode checked for stability. The electrode pair was calibrated once daily prior to measuring pH of samples; it was never recalibrated during a series of measurements. Following calibration, the electrode was soaked in sea water, of salinity similar to the sample, for at least 15 minutes to achieve chemical equilibrium and a steady-state junction potential. When measuring pH, the sample was initially gently agitated or stirred to assure good mixing at the electrode tip but without entraining air bubbles in the sample. Stirring was stopped to read the meter. The electrode was allowed to equilibrate, so the change in meter reading was less than 0.02 pH unit/minute before recording. Following each measurement, the electrode was rinsed with sea water and placed in fresh sea water for the temporary storage between measurements.

Fresh Water — (50 F.R. 30784, July 29, 1985)
Salt Water — (54 F.R. 19227, May 4, 1989)
See Appendix A for Aquatic Life Methodology.

ANTIMONY

7440-36-0

CRITERIA

Aquatic Life

The available data for antimony indicate that acute and chronic toxicity to freshwater aquatic life occur at concentrations as low as 9,000 and 1,600 μg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested. Toxicity to algae occurs at concentrations as low as 610 μg/L.

No saltwater organisms have been adequately tested with antimony, and no statement can be made concerning acute or chronic toxicity.

Human Health

For the protection of human health from the toxic properties of antimony ingested through water and contaminated aquatic organisms, the ambient water criterion is 146 μg/L.

For the protection of human health from the toxic properties of antimony ingested through contaminated aquatic organisms alone, the ambient water criterion is 45 mg/L.

Published human health critiera was recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 10/92. Recalculated IRIS values for antimony are 146.0 μg/L for ingestion of contaminated water and organisms and 4,500 μg/L for ingestion of contaminated aquatic organisms only.

(45 F.R. 79318, November 28, 1980)
See Appendix C for Human Health Methodology.

ARSENIC

7440-38-2

CRITERIA

Aquatic Life

Arsenic (III) — Freshwater — 1-hour average of 360 µg/L
4-day average of 190 µg/L
Saltwater — 1-hour average of 69 µg/L
4-day average of 36 µg/L

Summary

The chemistry of arsenic in water is complex, and the form present in a solution is dependent on environmental conditions such as Eh, pH, organic content, suspended solids, and sediment. The relative toxicities of the various forms of arsenic apparently vary from species to species. For inorganic arsenic (III), acute values for 16 freshwater animal species ranged from 812 µg/L for a cladoceran to 97,000 µg/L for a midge, but the three acute-chronic ratios only ranged from 4.660 to 4.862. The five acute values for inorganic arsenic (V) covered about the same range, but the single acute-chronic ratio was 28.71. The six acute values for MSMA ranged from 3,243 to 1,403,000 µg/L. The freshwater residue data indicated that arsenic is not bioconcentrated to a high degree, but that lower forms of aquatic life may accumulate higher arsenic residues than fish. The low bioconcentration factor and short half-life of arsenic in fish tissue suggest that residues should not be a problem to predators of aquatic life.

The available data indicate that freshwater plants differ a great deal as to their sensitivity to arsenic (III) and arsenic (V). In comparable tests, the alga, *Selenastrum capricornutum*, was 45 times more sensitive to arsenic (V) than to arsenic (III), although other data present conflicting information on the sensitivity of this alga to arsenic (V). Many plant values for inorganic arsenic (III) were in the same range as the available chronic values for freshwater animals; several plant values for arsenic (V) were lower than the one available chronic value.

The other toxicological data revealed a wide range of toxicity based on tests with a variety of freshwater species and endpoints. Tests with early life stages appeared to be the most sensitive indicator of arsenic toxicity. Values obtained from this type of test with inorganic arsenic (III) were lower than chronic values. For example, an effect concentration of 40 µg/L was obtained in a test on inorganic arsenic (III) with embryos and toad larvae.

Twelve species of saltwater animals have acute values for inorganic arsenic (III) from 232 to 16,030 µg/L, and the single acute-chronic ratio is 1.945. The only values available for inorganic arsenic (V) are for two invertebrate and are between 2,000 and 3,000 µg/L. Arsenic (III) and arsenic (V) are equally toxic to various species of saltwater algae, but the sensitivities of the species range from 19 µg/L to more than 1,000 µg/L. In a test with an oyster, a BCF of 350 was obtained for inorganic arsenic (III).

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 of arsenic (III) does not exceed 190 μg/L more than once every three years on the average and if the one-hour average concentration does not exceed 360 μg/L more than once every three years on the average.

The procedures described in the guidelines 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 arsenic (III) does not exceed 36 μg/L more than once every three years on the average and if the one-hour average concentration does not exceed 69 μg/L more than once every three years on the average. This criterion might be too high wherever *Skeletonema cosrarum or Thalassiosira aestivalis* are *ecologically* important.

Not enough data are available to allow derivation of numerical national water quality criteria for freshwater aquatic life for inorganic arsenic (V) or any organic arsenic compound. Inorganic arsenic (V) is acutely toxic to freshwater aquatic animals at concentrations as low as 850 μg/L, and an acute-chronic ratio of 28 was obtained with the fathead minnow. Arsenic (V) affected freshwater aquatic plants at concentrations as low as 48 μg/L. Monosodium methanearsenace (MSMA) is acutely toxic to aquatic animals at concentrations as low as 1,900 μg/L, but no data are available concerning chronic toxicity to animals or toxicity to plants.

Very few data are available concerning the toxicity of any form of arsenic other than inorganic arsenic (III) to saltwater aquatic life. The available data do show that inorganic arsenic (V) is acutely toxic to saltwater animals at concentrations as low as 2,319 μg/L and affected some saltwater plants at 13 to 56 μg/L. No data are available concerning the chronic toxicity of any form of arsenic other than inorganic arsenic (III) to saltwater aquatic life.

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 arsenic (III) exceeds the criterion. A stressed system, 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 design flow and 7Q5 or 7Q10 for the Criterion Continuous Concentration design flow in steady-state models for unstressed and stressed systems, respectively. These matters are discussed in more detail in EPA's "Technical Support Document for Water Quality-Based Toxics Control."

(50 F.R. 30784, July 29, 1985)

Human Health Criteria

For the maximum protection of human health from the potential carcinogenic effects due to exposure of inorganic arsenic through ingestion of water and aquatic organisms that are contaminated, the ambient water concentration should be zero based on the non-threshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels that may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}.

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60848). Recalculated IRIS values for arsenic are 0.018 μg/L for ingestion of contaminated water and organisms and 0.14 μg/L for ingestion of contaminated aquatic organisms only. IRIS values are based on a 10^{-6} risk level for carcinogens.

(45 F.R. 79318, November 28,1980) (50 F.R. 30784, July 29, 1985)
(57 F.R. 60848, December 22, 1993)
See Appendix A for Aquatic Life Methodology.
See Appendix C for Human Health Methodology.

ASBESTOS

1332-21-4

CRITERIA

Aquatic Life

No freshwater or saltwater organisms have been tested with any asbestiform mineral, therefore no statement can be made concerning its acute or chronic toxicity.

Human Health

Published human health criteria were recalculated to reflect data as of 12/92 (57 F.R. 60911). For the maximum protection of human health from the potential carcinogenic effects of exposure to asbestos through ingestion of water and contaminated aquatic organisms, the ambient water concentration should be zero. The estimated level that would result in increased lifetime cancer risks of 10^{-6} is 7,000,000 fibers/L. Estimates are for consumption of aquatic organisms only, excluding the consumption of water.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60911, December 22, 1993)
See Appendix C for Human Health Methodology.

BACTERIA

CRITERIA

Bathing Waters

(Full Body Contact) Based on a statistically sufficient number of samples (generally not less than five samples equally spaced over a 30-day period), the geometric mean of the indicated bacterial densities should not exceed one of the following (as displayed in Table 1).

Freshwater

Based on a statistically sufficient number of samples (generally not less than five samples equally spaced over a 30-day period), the geometric mean of the indicated bacterial densities should not exceed one or the other of the following:*

E. coli	126 per 100 ml; or
enterococci	33 per 100 ml;

no sample should exceed a one-sided confidence limit (C.L.) calculated using the following as guidance:

designated bathing beach	75% C.L.
moderate use for bathing	82% C.L.
light use for bathing	90% C.L.
infrequent use for bathing	95% C.L.

based on a site-specific log standard deviation; or if site data are insufficient to establish a log standard deviation, then using 0.4 as the log standard deviation for both indicators.

Marine Water

Based on a statistically sufficient number of samples (generally not less than five samples equally spaced over a 30-day period), the geometric mean of the enterococci densities should not exceed 35 per 100 ml; no sample should exceed a one-sided confidence limit using the following as guidance:

designated bathing beach	75% C.L.
moderate use for bathing	82% C.L.
light use for bathing	90% C.L.
infrequent use for bathing	95% C.L.

based on a site-specific log standard deviation; or if site data are insufficient to establish a log standard deviation, then using 0.7 as the log standard deviation.

*Only one indicator should be used. The regulatory agency should select the appropriate indicator for its conditions.

Shellfish Harvesting Waters

The median fecal coliform bacterial concentration should not exceed 14 Most Probable Number (MPN) per 100 mL, with not more than 10 percent of samples exceeding 43 MPN per 100 mL for the taking of shellfish.

The microbiological criterion for shellfish water quality has been established by international agreement to be 70 total coliforms per 100 mL, using a median MPN, with no more than 10 percent of the values exceeding 230 total coliforms per 100 mL. For evaluation of waters for recreational taking of shellfish, EPA recommends fecal coliform bacteria rather than total coliform bacteria.

REFERENCES:
Bathing Water Criteria – EPA 440/5-84-002
Bathing Water Criteria Laboratory Methods – EPA 600/4-85/076
Shellfish Water Criteria – Quality Criteria for Water (1976)
GPO Access #055-001-01049-4

(51 F.R. 8012, March 7, 1986)

Table 1.—Criteria for indicator for biological densities.

	SINGLE SAMPLE MAXIMUM ALLOWABLE DENSITY (4), (5)					
	ACCEPTABLE SWIMMING ASSOCIATED GASTROENTERITIS RATE PER 1000 SWIMMERS	STEADY-STATE GEOMETRIC MEAN INDICATOR DENSITY	DESIGNATED BEACH AREA (UPPER 75% C.L.)	MODERTE FULL BODY CONTACT RECREATION (UPPER 82%C.L.)	LIGHTLY USED FULL BODY CONTACT RECREATION (UPPER 90% C.L.)	INFREQUENTLY USED FULL BODY CONTACT RECREATION (UPPER 95%C.L.)
Freshwater*						
enterococci	8	33(1)	61	78	107	151
E. *coli*	8	126(2)	235	298	409	575
Marine Water						
enterococci	19	35(3)	104	158	276	501

* Only one indicator should be used. The regulatory agency should select the appropriate indicator for its conditions.

NOTES:

(1) Calculated to nearest whole number using equation:

$$\text{(mean enterococci density)} = \text{antilog}_{10} \frac{\textit{illness rate/1000 people} + 6.28}{9.40}$$

(2) Calculated to nearest whole number using equation:

$$\text{(mean E. } \textit{coli} \text{ density)} = \text{antilog}_{10} \frac{\textit{illness rate/1000 people} + 11.74}{9.40}$$

(3) Calculated to nearest whole number using equation:

$$\text{(mean enterococci density)} = \text{antilog}_{10} \frac{\textit{illness rate/1000 people} - 0.20}{12.17}$$

(4) Single sample limit = antilog_{10} [($\log_{10}$ indicator geometric mean density/100ml) + (factor determined from areas under the normal probability curve for the assumed level of probability) x ($\log_{10}$ stand. deviation)]

The appropriate factors for the indicated one-sided confidence levels:

75% C.L. — .675
82% C.L. — .935
90% C.L. — 1.28
95% C.L. — 1.65

(5) Based on the observed log standard deviations during the EPA studies: 0.4 for freshwater E. *coli* and enterococci; and 0.7 for marine water enterococci. Each jurisdiction should establish its own standard deviation for its conditions, which would than vary the single sample limit.

BARIUM

7440-39-3

CRITERIA

Aquatic Life

1 mg/L for domestic water supply (health).

Introduction

Barium is a yellowish white metal of the alkaline earth group. It occurs in nature chiefly as barite ($BaSO_4$) and witherite ($BaCO_3$), both highly insoluble salts. The metal is stable in dry air but readily oxidized by humid air or water.

Many of the salts of barium are soluble in both water and acid; soluble barium salts are reported to be poisonous. However, barium ions are generally thought to precipitate rapidly or be removed from a solution by absorption and sedimentation.

While barium is a malleable, ductile metal, its major commercial value is in its compounds. Barium compounds are used in a variety of industrial applications, including the metallurgic, paint, glass, and electronics industries, as well as for medicinal purposes.

Human Health

For the protection of human health from the toxic properties of barium ingested through water and contaminated aquatic organisms, the ambient water criterion is 1 mg/L.

(Quality Criteria for Water, July 1976) PB-263943
See Appendix D for Methodology.

BENZENE

71-43-2

CRITERIA

Aquatic Life

The available data for benzene indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 5,300 μg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of benzene to sensitive freshwater aquatic life.

The available data for benzene indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 5,100 μg/L and would occur at lower concentrations among species that are more sensitive than those tested. No definitive data are available concerning the chronic toxicity of benzene to sensitive saltwater aquatic life, but adverse effects occur at concentrations as low as 700 μg/L with a fish species exposed for 168 days.

Human Health

For the maximum protection of human health from the potential carcinogenic effects of exposure to benzene through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentrations should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time.

Published human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60911). Recalculated IRIS values for benzene are 1.2 μg/L for ingestion of contaminated water and organisms and 71 μg/L for ingestion of contaminated aquatic organisms only. IRIS values are based on a 10^{-6} risk level for carcinogens.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60911, December 22, 1992)
See Appendix C for Human Health Methodology.

BENZIDINE

92-87-5

CRITERIA

Aquatic Life

The available data for benzidine indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 2,500 μg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of benzidine to sensitive freshwater aquatic life.

Since saltwater organisms have not been tested with benzidine, no statement can be made concerning its acute and chronic toxicity.

Human Health

For the maximum protection of human health from the potential carcinogenic effects of exposure to benzidine through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentrations should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels that may result in an incremental increase of cancer risk over a lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}.

Published human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60913). Recalculated IRIS values for benzidine are 0.00012 μg/L for ingestion of contaminated water and organisms and 0.00054 μg/L for ingestion of contaminated aquatic organisms only. IRIS values are based on a 10^{-6} risk level for carcinogens.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60913, December 22, 1993)
See Appendix C for Human Health Methodology.

BERYLLIUM

7440-41-7

CRITERIA

Aquatic Life

The available data for beryllium indicate that acute and chronic toxicity to freshwater aquatic life occur at concentrations as low as 130 and 5.3 μg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested. Hardness has a substantial effect on acute toxicity.

The limited saltwater database available for beryllium does not permit any statement concerning acute or chronic toxicity.

Human Health

Human health criteria have been withdrawn for this compound (see 57 F.R. 60885, December 22, 1992). Although the human health criteria are withdrawn, EPA published a document for this compound that may contain useful human health information. This document was originally noticed in 45 F.R. 79326, November 28, 1980.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60911, December 22, 1992)
See Appendix C for Human Health Methodology.

BENZENE HEXACHLORIDE (BHC)

680-73-1

(See also: Hexachlorocyclohexane)

CRITERIA

Aquatic Life

The available data for a mixture of isomers of BHC indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 100 μg/L and would occur at lower concentrations among species that are more sensitive than those tested.

No data are available concerning the chronic toxicity of a mixture of isomers of BHC to sensitive freshwater aquatic life.

The available data for a mixture of isomers of BHC indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 0.34 μg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of a mixture of isomers of BHC to sensitive saltwater aquatic life.

(45 F.R. 79318, November 28, 1980)

See Appendix B for Aquatic Life Methodology.

BORON

CRITERION

750 μg/L for long-term irrigation on sensitive crops. Data are insufficient to determine acute or chronic toxicity of boron to freshwater or saltwater aquatic life.

Introduction

Boron is usually found as a sodium or calcium borate salt in nature, rather than in an elemental form. Boron salts are used in fire retardants, production of glass, leather tanning and finishing industries, cosmetics, photographic materials, metallurgy, and for high energy rocket fuels. Elemental boron also can be used in nuclear reactors for neutron absorption. Borates are used as "burnable" poisons.

Rationale

Boron is an essential element for the growth of plants, but no evidence indicates that it is required by animals. The maximum concentration found in 1,546 samples of river and lake waters from various parts of the United States was 5.0 mg/L; the mean value was 0.1 mg/L. Groundwaters could contain substantially higher concentrations in certain places. The concentration in sea water is reported as 4.5 mg/L in the form of borate. Naturally occurring concentrations of boron should have no effect on aquatic life.

The minimum lethal dose for minnows exposed to boric acid at 20°C for 6 hours was reported to be 18,000 to 19,000 mg/L in distilled water and 19,000 to 19,500 mg/L in hard water. In the dairy cow, 16 to 20 g/day of boric acid for 40 days produced no ill effects.

Sensitive crops have shown toxic effects at 1,000 ug/L or less of boron. When the boron concentration in irrigation waters was greater than 0.75 μg/L, some sensitive plants such as citrus began to show injury. Water containing 2 μg/L boron (with neutral and alkaline soils of high absorption capacities) might be used for some time without injury to sensitive plants. The criterion of 750 μg/L is thought to protect sensitive crops during long-term irrigation.

(Quality Criteria for Water, July 1976) PB-263943

See Appendix D for Methodology.

CADMIUM

7440-43-9

CRITERIA

Aquatic Life

Saltwater — 1-hour average of 43.0 µg/L
4-day average of 9.3 µg/L
Freshwater criteria are hardness dependant. See text.

Criteria

Freshwater acute values for cadmium are available for species in 44 genera and range from 1.0 µg/L for rainbow trout to 28,000 µg/L for a mayfly. The antagonistic effect of hardness on acute toxicity has been demonstrated with five species. Chronic tests have been conducted on cadmium with 12 freshwater fish species and four invertebrate species with chronic values ranging from 0.15 µg/L for *Daphnia magna* to 156 µg/L for the Atlantic salmon. Acute-chronic ratios are available for eight species and range from 0.9021 for the chinook salmon to 433.8 for the flagfish. Freshwater aquatic plants are affected by cadmium at concentrations ranging from 2 to 7,400 µg/L. These values are in the same range as the acute toxicity values for fish and invertebrate species and are considerably above the chronic values. Bioconcentration factors (BCFs) for cadmium in freshwater range from 164 to 4,190 for invertebrates and from 3 to 2,213 for fishes.

Saltwater acute values for cadmium and five species of fishes range from 577 µg/L for larval Atlantic silverside to 114,000 µg/L for juvenile mummichog. Acute values for 30 species of invertebrates range from 15.5 µg/L for a mysid to 135,000 µg/L for an oligochaete worm. The acute toxicity of cadmium generally increases as salinity decreases.

The effect of temperature seems to be species-specific. Two life-cycle tests with *Mysidopsis bahia* under different test conditions resulted in similar chronic values of 8.2 and 7.1 µg/L, but the acute-chronic ratios were 1.9 and 15, respectively. The acute values appear to reflect effects of salinity and temperature, whereas the few available chronic values apparently do not. A life-cycle test with *Mysidopsis bigelowi* also resulted in a chronic value of 7.1 µg/L and an acute-chronic ratio of 15. Studies with microalgae and macroalgae revealed effects at 22.8 to 860 µg/L.

BCFs determined with a variety of saltwater invertebrates ranged from 5 to 3,160. BCFs for bivalve molluscs were above 1,000 in long exposures, with no indication that steady state had been reached. Cadmium mortality is cumulative for exposure periods beyond four days. Chronic cadmium exposure resulted in significant effects on the growth of bay scallops at 78 µg/L and on reproduction of a copepod at 44 µg/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, freshwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration (in µg/L) of cadmium does not exceed the numerical value given by

$$e^{(0.7852[\ln(\text{hardness})]-3.490)}$$

more than once every three years on the average and if the one-hour average concentration (in μg/L) does not exceed the numerical value given by

$$e^{(1.128\ [\ln\ (\text{hardness})]-3.828)}$$

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 cadmium are 0.66, 1.1, and 2.0 μg/L, respectively, and the one-hour average concentrations are 1.8, 3.9, and 8.6 μg/L. If brook trout, brown trout, and striped bass are as sensitive as some data indicate, they might not be protected by this criterion.

The procedures described in the guidelines 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 cadmium does not exceed 9.3 μg/L more than once every three years on the average, and if the one-hour average concentration does not exceed 43 μg/L more than once every three years on the average.

The little information that is available concerning the sensitivity of the American lobster to cadmium indicates that this important species might not be protected by this criterion. In addition, data suggest that the acute toxicity of cadmium is salinity dependent; therefore, the one-hour average concentration might be underprotective at low salinities and overprotective at high salinities.

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 cadmium exceeds the criterion. A stressed system, 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 design flow and 7Q5 or 7Q10 for the Criterion Continuous Concentration 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."

Human Health Criteria

Human health criteria have been withdrawn for this compound (see 57 F.R. 60885, December 22, 1992). Although the human health criteria are withdrawn, EPA published a document for this compound that may contain useful human health information. This document was originally noticed in 45 F.R. 79326, November 28, 1980.

(45 F.R. 79318, November 28,1980) (50 F.R. 30784, July 29, 1985)
(57 F.R. 60885, December 22, 1992)
See Appendix A for Aquatic Life Methodology.

CARBON TETRACHLORIDE

56-23-5

CRITERIA

Aquatic Life

The available data for carbon tetrachloride indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 35,200 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of carbon tetrachloride to sensitive freshwater aquatic life.

The available data for carbon tetrachloride indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 50,000 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of carbon tetrachloride to sensitive saltwater aquatic life.

Human Health

For the maximum protection of human health from the potential carcinogenic effects of exposure to carbon tetrachloride through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentrations should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels that may result in incremental increase of cancer risk over a lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}.

Published human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60911, December 22, 1992). Recalculated IRIS values for carbon tetrachloride are 0.25 µg/L for ingestion of contaminated water and organisms and 4.4 µg/L for ingestion of contaminated aquatic life organisms. IRIS values are based on a 10^{-6} risk level for carcinogens.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60911, December 22, 1992)
See Appendix C for Human Health Methodology.

CHLORDANE

57-74-9

CRITERIA

Aquatic Life

For chlordane, the criterion to protect freshwater aquatic life as derived using the guidelines is 0.0043 μg/L as a 24-hour average. The concentration should not exceed 2.4 μg/L at any time.

For chlordane, the criterion to protect saltwater aquatic life as derived using the guidelines is 0.0040 μg/L as a 24-hour average. The concentration should not exceed 0.09 μg/L at any time.

Human Health

For the maximum protection of human health from the potential carcinogenic effects of exposure to chlordane through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels that may result in incremental increase of cancer risk over a lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}.

Published human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60914, December 22, 1992). Recalculated IRIS values for chlordane are 0.00057 μg/L for ingestion of contaminated water and organisms and 0.00059 μg/L for ingestion of contaminated aquatic organisms only. IRIS values are based on a 10^{-6} risk level for carcinogens.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60914, December 22, 1992)

See Appendix B for Aquatic Life Methodology.
See Appendix C for Human Health Methodology.

CHLORIDE

16887-00-6

CRITERIA

Aquatic Life

Not to exceed a one-hour average of 860 mg/L or a four-day average of 230 mg/L for freshwater aquatic life.

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 of dissolved chloride, when associated with sodium, does not exceed 230 mg/L more than once every three years on the average and if the one-hour average concentration does not exceed 860 mg/L more than once every three years on the average.

These criteria probably will not be adequately protective when the chloride is associated with potassium, calcium, or magnesium, rather than sodium. In addition, because freshwater animals have a narrow range of acute susceptibilities to chloride, excursions above these criteria might affect a substantial number of species.

(53 F.R. 19028, May 26, 1988)

See Appendix A for Aquatic Life Methodology.

CHLORINATED BENZENES

CRITERIA

Aquatic Life

The available data for chlorinated benzenes indicate that acute toxicity to freshwater aquatic life would occur at concentrations as low as 250 µg/L and at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of the more toxic of the chlorinated benzenes to sensitive freshwater aquatic life, but toxicity occurs at concentrations as low as 50 µg/L for a fish species exposed for 7.5 days.

The available data for chlorinated benzenes indicate that acute and chronic toxicity to saltwater aquatic life occur at concentrations as low as 160 and 129 µg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested.

Human Health

Monochlorobenzene (Chlorobenzene) 108-90-7

Published human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60911, December 22, 1992). Recalculated IRIS values for chlorobenzene are 680.0 µg/L for contaminated water and organisms and 21,000 µg/L for ingestion of contaminated aquatic organisms only.

Trichlorobenzenes

Because of insufficiency in the available information for the trichlorobenzenes, a criterion cannot be derived at this time using the present guidelines.

1,2,4,5-Tetrachlorobenzene 95-94-3

For the protection of human health from the toxic properties of 1,2,4,5-tetrachlorobenzene ingested through water and contaminated aquatic organisms, the ambient water criterion is determined to be 38 µg/L.

For the protection of human health from the toxic properties of 1,2,4,5-tetrachlorobenzene ingested through contaminated aquatic organisms alone, the ambient water criterion is determined to be 48 µg/L.

Pentachlorobenzene 608-93-5

For the protection of human health from the toxic properties of pentachlorobenzene ingested through water and contaminated aquatic organisms, the ambient water criterion is determined to be 74 µg/L.

For the protection of human health from the toxic properties of pentachlorobenzene ingested through contaminated aquatic organisms alone, the ambient water criterion is determined to be 85 µg/L.

For the maximum protection of human health from the potential carcinogenic effects due to exposure of hexachlorobenzene through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels that may result in incremental increase of cancer risk over a lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}.

Published human health criteria were recalculated using IRIS to reflect available data as of 12/92 (57 F.R. 609121, December 22, 1992). The recalculated IRIS value for chlorinated benzenes is .00075 μg/L for ingestion of contaminated water and organisms and 0.00077 μg/L for ingestion of contaminated organisms only. IRIS values are based on a 10^{-6} risk level for carcinogens.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60848, December 22, 1992)
See Appendix C for Human Health Methodology

CHLORINATED ETHANES

CRITERIA

Aquatic Life

The available freshwater data for chlorinated ethanes indicate that toxicity increases greatly with increasing chlorination and that acute toxicity occurs at concentrations as low as 118,000 µg/L for 1,2-dichloroethane; 18,000 µg/L for two trichloroethanes; 9,320 µg/L for two tetrachloroethanes; 7,240 µg/L for pentachloroethane; and 980 µg/L for hexachloroethane.

Chronic toxicity occurs at concentrations as low as 20,000 µg/L for 1,2-dichloroethane; 9,400 µg/L for 1,1,2-trichchloroethane; 2,400 µg/L for 1,1,2,2-tetrachloroethane; 1,100 µg/L for pentachloroethane; and 540 µg/L for hexachloroethane. Acute and chronic toxicity would occur at lower concentrations among species that are more sensitive than those tested.

The available saltwater data for chlorinated ethanes indicate that toxicity increases greatly with increasing chlorination and that acute toxicity to fish and invertebrate species occurs at concentrations as low as 113,000 µg/L for 1,2-dichloroethane; 31,200 µg/L for 1,1,1-trichloroethane; 9,020 µg/L for 1,1,2,2-tetrachloroethane; 390 µg/L for pentachloroethane; and 940 µg/L for hexachloroethane.

Chronic toxicity occurs at concentrations as low as 281 µg/L for pentachloroethane. Acute and chronic toxicity would occur at lower concentrations among species that are more sensitive than those tested.

Human Health

1,2-Dichloroethane 107-06-2

For the maximum protection of human health from the potential carcinogenic effects of exposure to 1,2-dichloroethane through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels that may result in incremental increase of cancer risk over a lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}.

Health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60848). Recalculated IRIS values for 1,2-dichloroethane are 0.38 µg/L for ingestion of contaminated water and organisms and 99 µg/L for ingestion of contaminated aquatic organisms only. IRIS values are based on 10^{-6} risk level for carcinogens.

1,1,2-Trichloroethane 79-00-5

For the maximum protection of human health from the potential carcinogenic effects of exposure to 1,1,2-trichloroethane through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the

present time. Therefore, the levels that may result in incremental increase of cancer risk over a lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}.

Human health criteria were recalculated using IRIS to reflect available data as of 12/92 (57 F.R. 60948). Recalculated IRIS values for 1,1,2-trichloroethane are 0.60 μg/L for ingestion of contaminated water and organisms and 42 μg/L for ingestion of contaminated aquatic organisms only. IRIS values are based on a 10^{-6} risk level for carcinogens.

1,1,2,2-Tetrachloroethane 79-34-5

For the maximum protection of human health from the potential carcinogenic effects of exposure to 1,1,2,2-tetrachloroethane through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels that may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}.

Human health criteria were recalculated using IRIS to reflect available data as of 12/92 (57 F.R. 60848). Recalculated IRIS values for 1,1,2,2- tetrachloroethane are 0.17 μg/L for ingestion of contaminated water and organisms and 11.0 μg/L for ingestion of contaminated aquatic organisms only. IRIS values are based on a 10^{-6} risk level for carcinogens.

Hexachloroethane 67-72-1

For the maximum protection of human health from the potential carcinogenic effects of exposure to hexachloroethane through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels that may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}.

Human health criteria were recalculated using IRIS to reflect available data as of 12/92 (57 F.R. 60848). Recalculated IRIS values for hexachloroethane are 1.9 μg/L for ingestion of contaminated water and organisms and 8.9 μg/L for ingestion of contaminated aquatic organisms only. IRIS values are based on 10^{-6} risk level for carcinogens.

1,1,1-Trichloroethane 71-55-6

Human health criteria have been withdrawn for one chlorinated ethane, 1,1, 1-trichloroethane (see 57 F.R. 60885, December 22, 1992). Although the human health criteria are withdrawn, EPA published a document for this compound that may contain useful human health information. This document was originally noticed in 45 F.R. 79328, November 28, 1980.

Because of insufficient available data for monochloroethane, 1,1-dichloroethane, 1,1,1,2-tetrachloroethane, and pentachloroethane, satisfactory criteria cannot be derived at this time using the present guidelines.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60848, December 22, 1992)
See Appendix C for Human Health Methodology.

CHLORINATED NAPHTHALENES

CRITERIA

Aquatic Life

The available data for chlorinated naphthalenes indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 1,600 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of chlorinated naphthalenes to sensitive freshwater aquatic life.

The available data for chlorinated naphthalenes indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 7.5 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of chlorinated naphthalenes to sensitive saltwater aquatic life.

Human Health

Human health criteria was calculated for 2-chloronaphthelene using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60890). The calculated IRIS values for 2-chloronaphthelene is 1,700 µg/L for ingestion of contaminated water and organisms and 4,300 mg/L for organisms only.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60890, December 22, 1992)
See Appendix B for Aquatic Life Methodology.

CHLORINATED PHENOLS

CRITERIA

Aquatic Life

The available freshwater data for chlorinated phenols indicate that toxicity generally increases with increased chlorination and that acute toxicity occurs at concentrations as low as 30 µg/L for 4-chloro-3-methylphenol to greater than 500,000 µg/L for other compounds. Chronic toxicity occurs at concentrations as low as 970 µg/L for 2,4,6-trichlorophenol. Acute and chronic toxicity would occur at lower concentrations among species that are more sensitive than those tested.

The available saltwater data for chlorinated phenols indicate that toxicity generally increases with increasing chlorination and that acute toxicity occurs at concentrations as low as 440 µg/L for 2,3,5,6-tetrachlorophenol and 29,700 µg/L for 4-chlorophenol. Acute toxicity would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of chlorinated phenols to sensitive saltwater aquatic life.

Human Health

3-Chlorophenol

Sufficient data are not available for 3-chlorophenol to derive a level that would protect against the potential toxicity of this compound. According to available organoleptic data, the estimated level is 0.1 µg/L to control undesirable taste and odor qualities of ambient water. Organoleptic data do have limitations as a basis for establishing a water quality criterion but do not have a demonstrated relationship to potentially adverse effects on human health.

4-Chlorophenol 106-48-9

Sufficient data are not available for 4-chlorophenol to derive a level that would protect against the potential toxicity of this compound. According to available organoleptic data, to control undesirable taste and odor qualities of ambient water the estimated level is 0.1 µg/L. Organoleptic data do have limitations as a basis for establishing a water quality criterion but do not have a demonstrated relationship to potentially adverse effects on human health.

2,3-Dichlorophenol

Sufficient data are not available for 2,3-dichlorophenol to derive a level that would protect against the potential toxicity of this compound. According to available organoleptic data, the estimated level is 0.04 µg/L to control undesirable taste and odor qualities of ambient water. Organoleptic data do have limitations as a basis for establishing a water quality criterion but do not have a demonstrated relationship to potentially adverse effects on human health.

2,5-Dichlorophenol

Sufficient data are not available for 2,5-dichlorophenol to derive a level that would protect against the potential toxicity of this compound. According to available organoleptic data, the estimated level is 0.5 μg/L to control undesirable taste and odor qualities of ambient water. Organoleptic data do have limitations as a basis for establishing a water quality criterion but do not have a demonstrated relationship to potentially adverse effects on human health.

2,6-Dichlorophenol

Sufficient data are not available for 2,6-dichlorophenol to derive a level that would protect against the potential toxicity of this compound. According to available organoleptic data, to control undesirable taste and odor qualities of ambient water, the estimated level is 0.2 μg/L. Organoleptic data do have limitations as a basis for establishing a water quality criterion but do not have a demonstrated relationship to potentially adverse effects on human health.

3,4-Dichlorophenol

Sufficient data are not available for 3,4-dichlorophenol to derive a level that would protect against the potential toxicity of this compound. According to available organoleptic data, to control undesirable taste and odor qualities of ambient water, the estimated level is 0.3 μg/L. Organoleptic data do have limitations as a basis for establishing a water quality criterion but do not have a demonstrated relationship to potentially adverse effects on human health.

2,4,5-Trichlorophenol 95-95-4

For comparison purposes, two approaches were used to derive criterion levels for 2,4,5-trichlorophenol. Based on available toxicity data, to protect public health, the derived level is 2.6 mg/L. Using available organoleptic data, to control undesirable taste and odor quality of ambient water, the estimated level is 1.0 μg/L. Organoleptic data do have limitations as a basis for establishing a water quality criterion but do not have a demonstrated relationship to potentially adverse effects on human health.

2,4,6-Trichlorophenol 88-06-2

For the maximum protection of human health from the potential carcinogenic effects of exposure to 2,4,6-trichlorophenol through the ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels that may result in incremental increase of cancer risk over a lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}.

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60848). Recalculated IRIS values for 2,4,6-Trichlorophenol are 2.1 μg/L for ingestion of contaminated water and organisms and 6.5 μg/L for ingestion of contaminated organisms only. IRIS values are based on a 10^{-6} risk level for carcinogens.

2,3,4,6-Tetrachlorophenol

Sufficient data are not available for 2,3,4,6-tetrachlorophenol to derive a level that would protect against the potential toxicity of this compound. According to available organoleptic data, to control undesirable taste and odor qualities of ambient water the estimated level is 1.0 μg/L. Organoleptic data have limitations as a basis for establishing a water quality criterion and a demonstrated relationship to potentially adverse effects on human health.

2-Methyl-4-Chlorophenol

Sufficient data are not available for 2-methyl-4-chlorophenol to derive a criterion level that would protect against any potential toxicity of this compound. According to available organoleptic data, to control undesirable taste and odor qualities of ambient water, the estimated level is 1,800 μg/L. Organoleptic data do have limitations as a basis for establishing a water quality criterion but do not have a demonstrated relationship to potentially adverse effects on human health.

3-Methyl-4-Chlorophenol 59-50-7

Sufficient data are not available for 3-methyl-4-chlorophenol to derive a criterion level that would protect against any potential toxicity of this compound. According to available organoleptic data, to control undesirable taste and odor qualities of ambient water, the estimated level is 3,000 μg/L. Organoleptic data do have limitations as a basis for establishing a water quality criterion but do not have a demonstrated relationship to potentially adverse effects on human health.

3-Methyl-6-Chlorophenol

Sufficient data are not available for 3-methyl-6-chlorophenol to derive a criterion level that would protect against any potential toxicity of this compound. According to available organoleptic data, the estimated level is 20 μg/L to control undesirable taste and odor qualities of ambient water. Organoleptic data do have limitations as a basis for establishing a water quality criterion but do not have a demonstrated relationship to potentially adverse effects on human health.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60848, December 22, 1992)
See Appendix C for Human Health Methodology.

CHLORINE

7782-50-5

CRITERIA

Aquatic Life

Freshwater — 4-day average of 11 μg/L
1-hour average of 19 μg/L
Saltwater — 4-day average of 7.5 μg/L
1-hour average of 13 μg/L

Summary

Thirty-three freshwater species in 28 genera have been exposed to total residual chlorine (TRC); the acute values range from 28 μg/L for *Daphnia magna* to 710 μg/L for the threespine stickleback. Fish and invertebrate species had similar ranges of sensitivity. Freshwater chronic tests have been conducted with two invertebrate and one fish species. The chronic values for these three species ranged from less than 3.4 to 26 μg/L, with acute-chronic ratios from 3.7 to greater than 78.

The acute sensitivities of 24 species of saltwater animals in 21 genera have been determined for CPO, and the LC50 range is from 26 μg/L for the eastern oyster to 1,418 μg/L for a mixture of two shore crab species. This range is very similar to that observed with freshwater species: fishes and invertebrates had similar sensitivities. Only one chronic test has been conducted with a saltwater species, *Menidia peninsulae*; the acute chronic ratio was 1.162.

The available data indicate that aquatic plants are more resistant to chlorine than fish and invertebrate species.

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 of total residual chlorine does not exceed 11 μg/L more than once every three years on the average, and if the one-hour average concentration does not exceed 19 μg/L more than once every three years on the average.

The procedures described in the guidelines indicate that saltwater aquatic organisms and their uses should not be affected unacceptably (except possibly where a locally important species is very sensitive) if the four-day average concentration of chlorine-produced oxidants does not exceed 7.5 μg/L more than once every three years on the average, and if the one-hour average concentration does not exceed 13 μg/L more than once every three years on the average.

The recommended exceedence frequency of three years is the Agency's best scientific judgment of the average amount of time for an unstressed system to recover from a pollution event in which exposure to chlorine exceeds the criterion. A stressed system — for example, one in which several outfalls occur in a limited area — would be expected to require more time

for recovery. An ecosystem's resilience and 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; however, if limited data or other factors make their use impractical, one should rely on a steady-state model. The Agency recommends the interim use of 1Q5 or 1Q10 for Criterion Maximum Concentration design flow and 7Q5 or 7Q10 for the Criterion Continuous Concentration design flow in steady-state models for unstressed and stressed systems, respectively. These matters are discussed in more detail in EPA's "Technical Support Document for Water Quality-Based Toxics Control."

(50 F.R. 30784, July 29, 1985)

See Appendix A for Aquatic Life Methodology.

CHLOROALKYL ETHERS

CRITERIA

Aquatic Life

The available data for chloroalkyl ethers indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 238,000 μg/L and would occur at lower concentrations among species that are more sensitive than those tested. No definitive data are available concerning the chronic toxicity of chloroalkyl ethers to sensitive freshwater aquatic life.

No saltwater organism has been tested with any chloroalkyl ether, and therefore, no statement can be made concerning acute or chronic toxicity.

Human Health

Bis(2-Chloroisopropyl) 108-60-1

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60848). Recalculated IRIS values for bis(2-chloroisopropyl) ether are 1,400 μg/L for ingestion of contaminated water and organisms and 170,000 μg/L for ingestion of contaminated aquatic organisms only.

Bis(Chloromethyl)

For the maximum protection of human health from the potential carcinogenic effects of exposure to bis(chloromethyl) ether through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentrations should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels that may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}. The corresponding recommended criteria are 37.6×10^{-6} μg/L, and 0.376×10^{-6} μg/L, respectively. If these estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 18.4×10^{-3} μg/L, 1.84×10^{-3} μg/L, respectively.

Bis(2-Chloroethyl) 111-44-4

For the maximum protection of human health from the potential carcinogenic effects of exposure to bis(2-chloroethyl) ether through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentrations should be zero based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels that may result in incremental increase of cancer risk over a lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}.The corresponding recommended criteria are 0.30 μg/L, 0.030 μg/L, and 0.003 μg/L, respectively. If these estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 13.6 μg/L, 1.36 μg/L, and 0.136 μg/L, respectively.

(45 F.R. 79318, November 28, 1980)

See Appendix C for Human Health Methodology.

CHLOROFORM

67-66-3

CRITERIA

Aquatic Life

The available data for chloroform indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 28,900 μg/L and would occur at lower concentrations among species that are more sensitive than the three tested species. As indicated by 27-day LC50 values, chronic toxicity occurs at concentrations as low as 1,240 μg/L and could occur at lower concentrations among species or other life stages that are more sensitive than the earliest life-cycle stages of the rainbow trout. The data base for saltwater species is limited to one test, and therefore, no statement can be made concerning acute or chronic toxicity.

Human Health

For the maximum protection of human health from the potential carcinogenic effects of exposure to chloroform through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentrations should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels that may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}.

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60848). Recalculated IRIS values for chloroform are 5.7 μg/L for ingestion of contaminated water and organisms and 470 μg/L for ingestion of contaminated aquatic organisms only. IRIS values are based on a 10^{-6} risk level for carcinogens.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60848, December 22, 1992)
See Appendix C for Human Health Methodology.

2-CHLOROPHENOL

95-57-8

CRITERIA

Aquatic Life

The available data for 2-chlorophenol indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 4,380 μg/L and would occur at lower concentrations among species that are more sensitive than those tested. No definitive data are available concerning the chronic toxicity of 2-chlorophenol to sensitive freshwater aquatic life, but flavor impairment occurs in one species of fish at concentrations as low as 2,000 μg/L.

No saltwater organisms have been tested with 2-chlorophenol, and therefore, no statement can be made concerning acute or chronic toxicity.

Human Health

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60890). Recalculated IRIS values for 2-chlorophenol are 120 μg/L for ingestion of contaminated water and organisms and 400 μg/L for ingestion organisms only. According to available organoleptic data, the estimated level is 0.1 μg/L to control undesirable taste and odor qualities of ambient water. Organoleptic data do have limitations as a basis for establishing a water quality criterion but do not have a demonstrated relationship to potentially adverse effects on human health.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60890, December 22, 1992)
See Appendix C for Human Health Methodology.

CHLOROPHENOXY HERBICIDES

2,4-D 94-75-7; *2,4,5-TP* 93-72-1

CRITERIA

2,4-D 100 µg/L for domestic water supply (health).
2,4,5-TP 10 µg/L for domestic water supply (health).

Rationale

Two widely used herbicides are 2,4-D (2,4-dichlorophenoxy) acetic acid and 2,4,5-TP (silvex) 2-(2,4,5-trichlorophenoxy) propionic acid. These compounds may exhibit different herbicidal properties, but all are hydolyzed rapidly to the corresponding acid in the body.

The subacute oral toxicity of chlorophenoxy herbicides has been investigated in a number of animals. The dog was found to be sensitive and often displayed mild injury in response to doses of 10 mg/kg/day for 90 days and serious effects from doses of 20 mg/kg/day for 90 days. The no-effect level of 2,4-D is 0.5 mg/kg/day in the rat and 8.0 mg/kg/day in the dog.

Table 1.—Derivation of approval limits (AL) for chlorophenoxy herbicides.

	LOWEST LONG TERM LEVELS WITH MINIMAL OR NO EFFECTS		CALCULATED MAXIMUM SAFE LEVELS FROM ALL SOURCES OF EXPOSURE			WATER	
Compound	Species	mg/kg/day[a]	Safety factor(X)	mg/kg/day	mg/man/day[b]	% of Safe level	AL mg/l[c]
2,4-D	Rat	0.5	1/500	0.1	7.0	20	0.1
	Dog	8.0	1/500	0.016	1.12[d]		
2,4,5-TP...	Rat	2.6	1/500	0.005	0.35	20	0.01
	Dog	0.9	1/500	0.002	0.14[d]		

[a]Assume weight of rat = 0.3 kg and of dog = 10 kg, assume average daily food consumption of rat = 0.05 kg and of dog = 0.2 kg
[b]Assume average weight of human adult = 70 kg
[c]Assume average daily intake of water for man = 2 liters
[d]Chosen as basis on which to derive AL

Table 1 illustrates the derivation of the criteria for the two chlorophenoxy herbicides. The long-term, no-effect levels (mg/kg/day) are listed for the rat and the dog. These values are adjusted by a factor of 1/500 for 2,4-D and 2,4,5-TP. The safe levels are then readjusted to reflect total allowable intake per person. Since little 2,4-D or 2,4,5-TP is expected to occur in foods, 20 percent of the safe exposure level can reasonably be allocated to water without jeopardizing the health of the consumer.

(Quality Criteria for Water, July 1976) PB-263943
See Appendix D for Methodology.

CHLORPYRIFOS

2921-88-2

CRITERIA

Aquatic Life

Freshwater — 4-day average of 0.041 μg/L
1-hour average of 0.083 μg/L
Saltwater — 4-day average of 0.0056 μg/L
1-hour average of 0.011 μg/L

Summary

The acute values for 18 freshwater species in 15 genera range from 0.11 μg/L for an amphipod to greater than 806 μg/L for 2 fishes and a snail. The bluegill is the most sensitive fish species with an acute value of 10 μg/L, but 7 intervetebrate genera are more sensitive. Smaller organisms seem to be more acutely sensitive than larger ones.

Chronic toxicity data are available for 1 freshwater species, the fathead minnow. Unacceptable effects occurred in second generation larvae at 0.12 μg/L, which was the lowest concentration tested. The resulting acute-chronic ratio was greater than 1,417.

Little information is available on the toxicity of chlorpyrifos to freshwater plants, although algal blooms frequently follow field applications. The only available bioconcentration test on chlorpyrifos with a freshwater species (the fathead minnow) resulted in a bioconcentration factor of 1,673.

The acute toxicity of chlorpyrifos has been determined for 15 species of saltwater animals in 12 genera with the acute values ranging from 0.01 μg/L for the Korean shrimp, *Palaemon macrodactylus*, to 1,911 μg/L for larvae of the eastern oyster, *Crassostrea virginica*. Arthropods are particularly sensitive to chlorpyrifos. Among the 10 species of fish tested, the 96-hour LC50s range from 0.58 μg/L for striped bass to 520 μg/L for gulf toadfish. Fish larvae are more sensitive than other life stages. Growth of the mysid, *Mysidopsis bahia*, was reduced at 0.004 μg/L in a life-cycle test. In early life-stage tests, the California grunion, *Leuresthes tenuis*, was the most sensitive of the six fishes, with growth being reduced at 0.30 μg/L. Of the seven acute-chronic ratios that have been determined with saltwater species, the five lowest range from 2.388 to 12.50, whereas the highest is 228.5.

Concentrations of chlorpyrifos affecting six species of saltwater phytoplankton range from 138 to 10,000 μg/L. Bioconcentration factors (BCFs) ranged from 100 to 5,100 when the gulf toadfish was exposed to concentrations increasing from 1.4 to 150 μg/L. Steady-state BCFs averaged from 100 to 757 for five fishes exposed in early life-stage tests.

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 of chlorpyrifos does not exceed 0.041 μg/L more than once every three years on the average, and if the one-hour average concentration does not exceed 0.083 μg/L more than once every three years on the average.

The procedures described in the guidelines also 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 chlorpyrifos does not exceed 0.0056 μg/L more than once every three years on the average, and if the one-hour average concentration does not exceed 0.011 μg/L more than once every three years on the average.

In the Agency's best scientific judgment, three years is the average amount of time aquatic ecosystems should be provided between excursions. The resiliences of ecosystems and their abilities to recover differ greatly, however, and site-specific allowed excursion frequencies can be established if adequate justification is provided.

When designing waste treatment facilities, criteria for developing water quality-based permit limits must be applied to an appropriate wasteload allocation model. Dynamic models are preferred for the application of these criteria. Limited data or other considerations might make their use impractical, in which case one must rely on a steady-state model.

(51 F.R. 43665, December 3, 1986)
See Appendix A for Aquatic Life Methodology.

CHROMIUM

CRITERIA

Aquatic Life

Chromium (VI) Freshwater — 4-day average of 11 μg/L
1-hour average of16 μg/L
Saltwater — 4-day average of 50 μg/L
1-hour average of 1,100 μg/L

Summary

Chromium (VI) 7440-47-3

Acute toxicity values for chromium (VI) are available for freshwater animal species in 27 genera and range from 23.07 μg/L for a cladoceran to 1,870,000 μg/L for a stonefly. These species include a wide variety of animals that perform a spectrum of ecological functions. All five tested species of daphnids are especially sensitive. The few data that are available indicate that the acute toxicity of chromium (VI) decreases as hardness and pH increase.

The chronic value for both rainbow trout and brook trout is 264.6 mg/L, which is much lower than the chronic value of 1,987 μg/L for the fathead minnow. The acute-chronic ratios for these three fishes range from 18.55 to 260.8. In all three chronic tests, a temporary reduction in growth occurred at low concentrations. Six chronic tests with five species of daphnids gave chronic values that range from 2.5 to 40 μg/L; the acute-chronic ratios range from 1.130 to 9.680. Except for the fathead minnow, all the chronic tests were conducted in soft water. Green algae are quite sensitive to chromium (VI). The bioconcentration factor obtained with rainbow trout is less than 3. Growth of chinook salmon was reduced at a measured concentration of 16 μg/L.

The acute toxicity of chromium (VI) to 23 saltwater vertebrate and invertebrate species ranges from 2,000 μg/L for a polychaete worm and a mysid to 105,000 μg/L for the mud snail. The chronic values for a polychaete range from 13 to 36.74 μg/L, whereas that for a mysid is 132 μg/L. The acute-chronic ratios range from 15.38 to 238.5. Toxicity to macroalgae was reported at 1,000 and 5,000 μg/L. Bioconcentration factors for chromium (VI) range from 125 to 236 for bivalve molluscs and polychaetes.

Chromium (III) 1308-14-1

Acute values for chromium (III) are available for 20 freshwater animal species in 18 genera ranging from 2,221 μg/L for a mayfly to 71,060 μg/L for a caddisfly. Hardness has a significant influence on toxicity, with chromium (III) being more toxic in soft water.

A life-cycle test with *Daphnia magna* in soft water gave a chronic value of 66 μg/L. In a comparable test in hard water, the lowest test concentration of 44 μg/L inhibited reproduction of *D. magna*, but this effect may have resulted from ingested precipitated chromium. In a life-cycle test with the fathead minnow in hard water, the chronic value was 1,025 μg/L.

Toxicity data are available for only two freshwater plant species: a concentration of 9,900 µg/L inhibited growth of roots of Eurasian watermilfoil, and a freshwater green alga was affected by a concentration of 397 µg/L in soft water. No bioconcentration factor has been measured for chromium (III) with freshwater organisms.

Only two acute values are available for chromium (III) in saltwater 10,300 µg/L for the eastern oyster and 31,500 µg/L for the mummichog. In a chronic test, effects were not observed on a polychaete worm at 50,400 µg/L at pH=7.9, but acute lethality occurred at pH=4.5. Bioconcentration factors for saltwater organisms and chromium (III) range from 86 to 153, similar to the bioconcentration factors for chromium (VI) and saltwater species.

NATIONAL CRITERIA

Aquatic Life

Chromium (VI)

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 of chromium (VI) does not exceed 11 µg/L more than once every three years on the average, and if the one-hour average concentration does not exceed 16 µg/L more than once every three years on the average.

The procedures described in the guidelines 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 chromium (VI) does not exceed 50 µg/L more than once every three years on the average, and if the one-hour average concentration does not exceed 1,100 µg/L more than once every three years on the average. Data suggest that the acute toxicity of chromium (VI) is salinity dependent; therefore, the one-hour average concentration might be underprotective at low salinities.

Chromium (III)

The procedures described in the guidelines 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 µg/L) of chromium (III) does not exceed the numerical value given by

$$e^{(0.8190[\ln(\text{hardness})]+1.561)}$$

more than once every three years on the average, and if the one-hour average concentration (in µg/L) does not exceed the numerical value given by

$$e^{(0.8190[\ln(\text{hardness})]+3.688)}$$

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 chromium (III) are 120, 210, and 370 µg/L, respectively, and the one-hour average concentrations are 980, 1,700, and 3,100 µg/L.

No saltwater criterion can be derived for chromium (III), but 10,300 μg/L is the EC50 for eastern oyster embryos, whereas 50,400 μg/L did not affect a polychaete worm in a life-cycle test.

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

In designing waste treatment facilities, criteria must be applied to an appropriate wasteload allocation model; dynamic models are preferred. Limited data or other factors may make use of these models impractical, in which case one should rely on a steady-state model. The Agency recommends the interim use of 1Q5 for 1Q10 for Criterion Maximum Concentration design flow and 7Q5 or 7Q10 for the Criterion Continuous Concentration design flow in steady-state models for unstressed and stressed systems, respectively. These matters are discussed in more detail in EPA's "Technical Support Document for Water Quality-Based Toxics Control."

Human Health

Human health criteria have been withdrawn for this compound (see 57 F.R. 60885, December 22, 1992). Although the human health criteria are withdrawn, EPA published a document for this compound that may contain useful human health information. This document was originally noticed in 45 F.R. 79331, November 28, 1980.

(45 F.R. 79318 November 28,1980) (50 F.R. 30784, July 29, 1985)
(57 F.R. 60848, December 22, 1992)
See Appendix A for Aquatic Life Methodology.

COLOR

CRITERIA

- For aesthetic purposes, waters shall be virtually free from substances producing objectionable color;
- The source of supply should not exceed 75 color units on the platinum-cobalt scale for domestic water supplies; and
- Increased color (in combination with turbidity) should not reduce the depth of the compensation point for photosynthetic activity by more than 10 percent from the seasonally established norm for aquatic life.

Introduction

Degradation processes in the natural environment are the principal contributors to color in water. Although colloidal forms of iron and manganese occasionally color water, the most common causes of color in water are complex organic compounds originating from the decomposition of naturally occurring organic matter. Sources of organic matter include materials in the soil that were generated by humans such as tannins, human acid, humates, and decayed material from plankton and other aquatic plants. Industrial discharges may contribute similar compounds from, for example, the pulp and paper and tanning industries. Other industrial discharges, such as those from certain textile and chemical processes, may contain brightly colored substances (see Table 1).

Table 1.—Maximum color of surface waters that have been used as industrial water supplies.

INDUSTRY OR INDUSTRIAL USE	COLOR UNITS
Boiler makeup water	1,200
Cooling water	1,200
Pulp and paper water	360
Chemical and allied products water	500
Petroleum products water	25

Surface waters may appear colored because of suspended matter, which comprises turbidity. Such color is referred to as "apparent color" and is differentiated from true color caused by colloidal human materials. Natural color is reported in color "units" that generally are determined by the platinum-cobalt method.

No general agreement exists as to the chemical composition of natural color, and in fact, the composition may vary chemically from place to place. Examined color-causing colloids have been characterized as aromatic, polyhydroxy, methoxy carboxylic acids. Color-causing constituents were characterized as being dialyzable and composed of aliphatic, polyhydroxyl carboxylic acids with molecular weights varying from less than 200 to approximately 400. The colloidal fraction of color exists in the 3.5 to 10 mu diameter range. Other characteristics of color observed in laboratory studies of natural waters were summarized as follows: color is caused by

light scattering and fluorescence rather than absorption of light energy, and pH affects both particle size of the color-causing colloids and the intensity of color itself.

(Quality Criteria for Water, July 1976) PB-263943
See Appendix D for Methodology.

*COPPER

7440-50-8

CRITERIA

Aquatic Life

Not to exceed 2.9 μg/L in salt water.
Freshwater criteria are hardness dependent. See text.

Summary

Acute toxicity data are available for species in 41 genera of freshwater animals. At a hardness of 50 mg/L, the genera range in sensitivity from 16.74 μg/L for *Ptychocheilus* to 10,240 μg/L for *Acroneuria*. Data for eight species indicate that acute toxicity decreases as hardness increases. Additional data for several species indicate that toxicity also decreases with increases in alkalinity and total organic carbon.

Chronic values available for 15 freshwater species range from 3.873 μg/L for brook trout to 60.36 μg/L for northern pike. Fish and invertebrate species seem to be almost equally sensitive to the chronic toxicity of copper.

Toxicity tests on copper conducted with a wide range of freshwater plants indicate sensitivities similar to those of animals. Complexing effects of the test media and a lack of good analytical data make it difficult to interpret and apply these results. Protection of animal species, however, appears to offer adequate plant protection. Bioconcentrations of copper are light in edible portion of freshwater aquatic species.

Saltwater animals' acute sensitivities to copper range from 5.8 μg/L for the blue mussel to 600 μg/L for the green crab. A chronic life-cycle test has been conducted with a mysid; adverse effects were observed at 77 μg/L but not at 38 μg/L, which resulted in an acute-chronic ratio of 3.346. Several saltwater algal species have been tested, and effects were observed between 5 and 100 μg/L. Oysters can bioaccumulate copper up to 28,200 times and become bluish green, apparently without significant mortality. In long-term exposures, the bay scallop was killed at 5 μg/L.

National Criteria

The procedures described in the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and 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 μg/L) of copper does not exceed the numerical value given by

$$e^{(0.8545[\ln(\text{hardness})]-1.465)}$$

more than once every three years on the average, and if the one-hour average concentration (in μg/L) does not exceed the numerical value given by

$$e^{(0.9422[\ln(\text{hardness})]-1.464)}$$

*Indicates suspension, cancelation, or restriction by U.S. EPA Office of Pesticides and Toxic Substances.

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 copper are 6.5, 12, and 21 μg/L, respectively, and the one-hour average concentrations are 9.2, 18, and 34 μg/L.

The procedures described in the guidelines 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 one-hour average concentration of copper does not exceed 2.9 μg/L more than once every three years on the average.

The recommended exceedence frequency of three years is the Agency's best scientific judgment of the average time needed for an unstressed system to recover from a pollution event in which exposure to copper exceeds the criterion. For example, a stressed system (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 can be established if adequate justification is provided.

In developing waste treatment facilities, criteria 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 design flow and 7Q5 or 7Q10 for the Criterion Continuous Concentration design flow in steady-state models for unstressed and stressed systems respectively. These matters are discussed in more detail in EPA's "Technical Support Document for Water Quality-Based Toxics Control."

Human Health

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60890). The recalculated IRIS values for copper is 1,300 μg/L for ingestion of contaminated water and organisms. Using available organoleptic data, the estimated level is 1 mg/L for controlling undesirable taste and odor quality of ambient water. Organoleptic data as a basis for establishing a water quality criteria have limitations and no demonstrated relationship to potentially adverse effects on human health.

(45 F.R. 79318 November 28,1980) (50 F.R. 30784, July 29, 1985)
(57 F.R. 60890, December 22, 1992)
See Appendix A for Aquatic Life Methodology.
See Appendix C for Human Health Methodology.

CYANIDE

57-12-5

CRITERIA

Aquatic Life

Freshwater — 4-day average of 5.2 μg/L
1-hour average of 22 μg/L
Saltwater — 1-hour average of 1.0 μg/L

Summary

Data on the acute toxicity of free cyanide (the sum of cyanide present as HCN and CN-, expressed as CN) are available for a wide variety of freshwater species that are involved in diverse community functions. In tests, the acute sensitivities ranged from 44.73 μg/L to 2,490 μg/L, but all of the species with acute sensitivities above 400 μg/L were invertebrates. A long-term survival and a partial and life-cycle test with fish gave chronic values of 13.57, 7.849, and 16.39 μg/L, respectively. Chronic values for two freshwater invertebrate species were 18.33 and 34.06 μg/L. Freshwater plants were affected at cyanide concentrations ranging from 30 μg/L to 26,000 μg/L.

Free cyanide's acute toxicity to saltwater species ranged from 4.893 μg/L to 10,000 μg/L; invertebrates were both the most and least sensitive species. In an early life-stage test with the sheepshead minnow, long-term survival gave a chronic value of 36.12 μg/L. Long-term survival in a mysid life-cycle test resulted in a chronic value of 69.71 μg/L. Tests with the red macroalga, *Champia parvula*, showed cyanide toxicity at 11 to 25 μg/L, but other species were affected at concentrations up to 3,000 μg/L.

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 of cyanide does not exceed 5.2 μg/L more than once every three years on the average, and if the one-hour average concentration does not exceed 22 μg/L more than once every three years on the average.

The procedures described in the guidelines 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 one-hour average concentration of cyanide does not exceed 1.0 μg/L more than once every three years on the average.

EPA believes that a measurement such as free cyanide would provide a more scientifically correct basis upon which to establish criteria for cyanide. The criteria were developed on this basis. However, at this time EPA has approved no methods for such a measurement to implement the criteria through Agency and State regulatory programs.

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

cyanide exceeds the criterion. A stressed system, 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.

In designing waste treatment facilities, criteria must be applied to an appropriate wasteload allocation model; dynamic models are preferred. Limited data or other factors may make use of these models 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 design flow and 7Q5 or 7Q10 for the Criterion Continuous Concentration design flow in steady-state models for unstressed and stressed systems, respectively. These matters are discussed in more detail in EPA's "Technical Support Document for Water Quality-Based Toxics Control."

Human Health

Published human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60911). Recalculated IRIS values for cyanide are 700 μg/L for ingestion of contaminated water and organisms and 220,000 μg/L for ingestion of contaminated aquatic organisms only.

(45 F.R. 79318 November 28, 1980) (50 F.R. 30784, July 29, 1985)
(57 F.R. 60911, December 22, 1992)

See Appendix A for Aquatic Life Methodology.
See Appendix C for Human Health Methodology.

DDT AND METABOLITES

72-54-8

CRITERIA

Aquatic Life

24-hour average for freshwater and saltwater is 0.001 μg/L .
Not to exceed at anytime 1.1 μg/L in fresh water or 0.13 μg/L in salt water.

DDT

The criterion for DDT and its metabolites to protect freshwater aquatic life as derived using the guidelines is 0.0010 μg/L as a 24-hour average. The concentration should not exceed 1.1 μg/L at any time.

The criterion for DDT and its metabolites to protect saltwater aquatic life as derived using the guidelines is 0.0010 μg/L as a 24-hour average. The concentration should not exceed 0.13 μg/L at any time.

TDE

The available data for TDE indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 0.6 μg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning TDE's chronic toxicity to sensitive freshwater aquatic life.

The available data for TDE indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 3.6 μg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning TDE's chronic toxicity to sensitive saltwater aquatic life.

DDE

The available data for DDE indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 1,050 μg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning DDE's chronic toxicity to sensitive freshwater aquatic life.

The available data for DDE indicate that acute toxicity to saltwater aquatic life occurs in concentrations as low as 14 μg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning DDE's chronic toxicity to sensitive saltwater aquatic life.

Human Health

The ambient water concentration should be zero, based on the non-threshold assumption for this chemical, for the maximum protection of human health from the potential carcinogenic effects of exposure to DDT through ingestion of contaminated water and contaminated aquatic organisms. However, zero level may not be attainable at the present time. Therefore, the levels that may result in incremental increase of cancer risk over a lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}.

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60848). The recalculated IRIS value for DDT is 0.00059 μg/L for both ingestion of water and organisms, and ingestion of contaminated aquatic organisms only. IRIS values are based on a 10^{-6} risk level for carcinogens.

The recalculated IRIS value for TDE is 0.00083 μg/L for both ingestion of contaminated water and organisms and for ingestion of contaminated aquatic organisms only.

The recalculated IRIS value for DDE is 0.00059 μg/L for both ingestion of contaminated water and organisms and for ingestion of contaminated aquatic organisms only.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60848, December 22, 1992)

See Appendix B for Aquatic Life Methodology.

See Appendix C for Human Health Methodology.

DEMETON

8065-48-3

CRITERIA

Aquatic Life

0.1 μg/L for both freshwater and saltwater aquatic life.

Rationale

Static LC50 bioassays yielded toxicity values for the phosphorus pesticide demeton and for carp, goldfish, fathead minnow, channel catfish, guppy, rainbow trout, and bluegill ranging from 70 μg/L to 15,000 μg/L. These tests demonstrate a sharp division in species sensitivity, with the bluegill, *Lepomis macrochirus*; rainbow trout, *Oncorhynchus mykiss*; and guppy, *Poecilia reticulata* being susceptible to lower concentrations, while the remaining species were comparatively resistant.

Bluegills with a 24-hour LC50 of 70 μg/L were the most sensitive fish. Acute toxicity values reported for invertebrates range from 10 to 100,000 μg/L.*

Static LC50 data for invertebrate *Gammarus fasciatus* yield 24- and 96-hour LC50 values of 500 μg/L and 27 μg/L, respectively. Studies indicate residual effects of AChE inhibition from exposure to demeton. The few data on toxicity of demeton to marine organisms includes a 48-hour EC50 of 63 μg/L for the pink shrimp, *Penaeus duorarum*, and a 24-hour LC50 of 550 μg/L for the spot, *Leiostomus xanthurus*.

The criterion must be based partly on the fact that all organophosphates inhibit the production of the AChE enzyme. Demeton is unique, however, in that the persistence of its AChE-inhibiting ability is greater than that of 10 other common organophosphates. Because such inhibition may be additive with repeated exposures and may be compounded by any of the organophosphates, it is recommended that a criterion for demeton be based primarily on its enzyme-inhibiting potential. A criterion of 0.1 μg/L demeton for freshwater and marine aquatic life is recommended, since that concentration will not be expected to significantly inhibit AChE over a long period. In addition, the criterion recommendation is in close agreement with the criteria for the other organophosphates.

(Quality Criteria for Water, July 1976) PB-263943

See Appendix D for Methodology.

*Crustaceans and insect larvae were considerably more sensitive overall to demeton than molluscs and tubifex worms.

DICHLOROBENZENES

25321-22-6

CRITERIA

Aquatic Life

The available data for dichlorobenzenes indicate that acute and chronic toxicity to freshwater aquatic life occur at concentrations as low as 1,120 and 763 µg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested.

The available data for dichlorobenzenes indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 1,970 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of dichlorobenzenes to sensitive saltwater aquatic life.

Human Health

1,2-Dichlorobenzene 95-50-1

For the maximum protection of human health from the potential effects of exposure to 1,2-dichlorobenzene, the human health criteria were recalculated using the Integrated Risk Information System (IRIS) to reflect available information as of 12/92 (57 F.R. 60848). The recommended criteria are 2,700 µg/L for ingestion of contaminated water and organisms and 17,000 µg/L for ingestion of contaminated organisms only.

1,3-Dichlorobenzene 541-73-1

For the maximum protection of human health from the potential effects of exposure to 1,3-dichlorobenzene, the recommended criteria are 400 µg/L for ingestion of contaminated water and organisms and 2,600 µg/L for ingestion of contaminated organisms only.

1,4-Dichlorobenzene 106-46-7

For the maximum protection of human health from the potential effects of exposure to 1,4-dichlorobenzene, the recommended criteria are 400 µg/L for ingestion of contaminated water and organisms and 2,600 µg/L for ingestion of contaminated organisms only.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60848, December 22, 1992)
See Appendix C for Human Health Methodology.

DICHLOROBENZIDINE

91-94-1

CRITERIA

Aquatic Life

The data base available for dichlorobenzidines and freshwater organisms is limited to one test on bioconcentration of 3,3-dichlorobenzidine; therefore, no statement can be made concerning acute or chronic toxicity.

No saltwater organisms have been tested with any dichlorobenzidine; therefore, no statement can be made concerning acute or chronic toxicity.

Human Health

For the maximum protection of human health from the potential carcinogenic effects of exposure to dichlorobenzidine through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentrations should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels that may result in incremental increase of cancer risk over a lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}.

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60848). Recalculated IRIS values for 3,3-dichlorobenzidine are 0.04 µg/L for ingestion of contaminated water and organisms and 0.077 µg/L for ingestion of contaminated aquatic organisms only. IRIS values are based on a 10^{-6} risk level for carcinogens.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60848, December 22, 1992)
See Appendix C for Human Health Methodology.

DICHLOROETHYLENES

25323-30-3

CRITERIA

Aquatic Life

The available data for dichloroethylenes indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 11,600 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No definitive data are available concerning the chronic toxicity of dichloroethylenes to sensitive freshwater aquatic life.

The available data for dichloroethylenes indicate that acute and chronic toxicity to saltwater aquatic life occurs at concentrations as low as 224,000 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of dichloroethylenes to sensitive saltwater aquatic life.

Human Health

1,1-Dichloroethylene 75-35-4

For the maximum protection of human health from the potential carcinogenic effects of exposure to 1, 1-dichloroethylene through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentrations should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels that may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}.

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60848). Recalculated IRIS values for 1,1-dichloroethylene are 0.057 µg/L for ingestion of contaminated water and organisms and 3.2 µg/L for ingestion of contaminated aquatic organisms only. IRIS values are based on a 10^{-6} risk level for carcinogens.

1,2-Dichloroethylene 156-60-5

Human health criteria were recalculated using IRIS to reflect available data as of 12/92 (57 F.R. 60890). Recalculated IRIS values for 1,2-trans-dichloroethylene are 700 µg/L for ingestion of contaminated water and organisms. IRIS values are based on a 10^{-6} risk level for carcinogens.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60890, December 22, 1992)
See Appendix C for Human Health Methodology.

2,4-DICHLOROPHENOL

120-83-2

CRITERIA

Aquatic Life

The available data for 2,4-dichlorophenol indicate that acute and chronic toxicity to freshwater aquatic life occurs at concentrations as low as 2,020 and 365 μg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested. Mortality to early life stages of one species of fish occurs at concentrations as low as 70 μg/L.

Only one test has been conducted with saltwater organisms and 2,4-dichlorophenol, and therefore, no statement can be made concerning acute or chronic toxicity.

Human Health

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60848). Recalculated IRIS values for 2,4-dichlorophenol are 93.0 μg/L for ingestion of contaminated water and organisms and 790 μg/L for ingestion of contaminated aquatic organisms only.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60848, December 22, 1992)
See Appendix C for Human Health Methodology.

DICHLOROPROPANE

26638-19-7

DICHLOROPROPENE

26952-23-8

CRITERIA

Aquatic Life

The available data for dichloropropane indicate that acute and chronic toxicity to freshwater aquatic life occurs at concentrations as low as 23,000 and 5,700 µg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested. Acute and chronic toxicity to saltwater aquatic life occur at concentrations as low as 10,300 and 3,040 µg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested.

The available data for dichloropropene indicate that acute and chronic toxicity to freshwater aquatic life occurs at concentrations as low as 6,060 and 244 µg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested. Acute toxicity to saltwater aquatic life occurs at concentrations as low as 790 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of dichloropropene to sensitive saltwater aquatic life.

Human Health

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60890). Recalculated IRIS values for 1,2-dichloropropane is 0.52 µg/L per ingestion of water and organisms and 39.0 µg/L per ingestion of organisms only.

For the protection of human health from the toxic properties of dichloropropenes ingested through water and contaminated aquatic organisms, the ambient water criterion is 87 µg/L.

For the protection of human health from the toxic properties of dichloropropenes ingested through contaminated aquatic organisms alone, the ambient water criterion is 14.1 mg/L.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60890, December 22, 1992)
See Appendix C for Human Health Methodology.

DIELDRIN

60-57-1

CRITERIA

Aquatic Life

Freshwater — 24-hour average of 0.0019 µg/L
Never to exceed 2.5 µg/L
Saltwater — 24-hour average of 0.0019 µg/L
Never to exceed 0.71 µg/L

To protect freshwater aquatic life, the criterion for dieldrin is 0.0019 µg/L as a 24-hour average. The concentration should not exceed 2.5 µg/L at any time.

To protect saltwater aquatic life, the criterion as derived using the guidelines is 0.0019 µg/L as a 24-hour average. The concentration should not exceed 0.71 µg/L at any time.

Human Health

For the maximum protection of human health from the potential carcinogenic effects of exposure to dieldrin through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels that may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}.

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60848). Recalculated IRIS values for dieldrin are 0.00014 µg/L for ingestion of contaminated water and organisms and also for ingestion of contaminated aquatic organisms only. IRIS values are based on a 10^{-6} risk level for carcinogens.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60848, December 22, 1992)
See Appendix B for Aquatic Life Methodology.
See Appendix C for Human Health Methodology.

2,4-DIMETHYLPHENOL

105-67-9

CRITERIA

Aquatic Life

The available data for 2,4-dimethylphenol indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 2,120 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of dimethylphenol to sensitive freshwater aquatic life.

No saltwater organisms have been tested with 2,4-dimethylphenol, and therefore, no statement can be made concerning acute or chronic toxicity.

Human Health

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect data available as of 12/92 (57 F.R. 60890). Recalculated IRIS values for 2,4-dimethylphenol are 540 µg/L for ingestion of contaminated water and organisms and 2,300 µg/L for ingestion of contaminated organisms only.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60890, December 22, 1992)
See Appendix C for Human Health Methodology.

DINITROTOLUENE

25321-14-6

CRITERIA

Aquatic Life

The available data for 2,4-dinitrotoluene indicate that acute and chronic toxicity to freshwater aquatic life occurs at concentrations as low as 330 and 230 μg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested.

The available data for 2,4-dinitrotoluene indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 590 μg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of dinitrotoluenes to sensitive saltwater aquatic life but a decrease in algal cell numbers occurs at concentrations as low as 370 μg/L.

Human Health

2,4-Dinitrotoluene 121-14-2

For the maximum protection of human health from the potential carcinogenic effects of exposure to 2,4-dinitrotoluene through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels that may result in incremental increase of cancer risk over a lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}. The corresponding recommended criteria are 1.1 μg/L, 0.11 μg/L, and 0.011 μg/L, respectively. If these estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 91 μg/L, 9.1 μg/L, and 0.91 μg/L, respectively.

(45 F.R. 79318, November 28, 1980)
See Appendix C for Human Health Methodology.

DIPHENYLHYDRAZINE

122-66-7

CRITERIA

Aquatic Life

1, 2-Diphenylhydrazine

The available data for 1,2-diphenylhydrazine indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 270 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of 1,2-diphenylhydrazine to sensitive freshwater aquatic life.

No saltwater organisms have been tested with 1,2-diphenylhydrazine, and therefore, no statement can be made concerning its acute or chronic toxicity.

Human Health

For the maximum protection of human health from the potential carcinogenic effects of exposure to diphenylhydrazine through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentrations should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels that may result in incremental increase of cancer risk over a lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}. The corresponding recommended criteria are 422 ng/L, 42 ng/L, and 4 ng/L, respectively. If these estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 5.6 µg/L, 0.56 µg/L, and 0.056 µg/L, respectively.

Published human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 E.R. 60913). Recalculated IRIS values for 1,2-diphenylhydrazine are 0.040 µg/L for ingestion of contaminated water and organisms and 0.54 µg/L for ingestion of contaminated aquatic organisms only. IRIS values are based on a 10^{-6} risk level for carcinogens.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60913, December 22, 1992)
See Appendix C for Human Health Methodology.

DI-2-ETHYLHEXYL PHTHALATE

117-81-7

CRITERIA

Aquatic Life

The procedures described in the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" do not allow for the derivation of national criteria for di-2-ethylhexyl phthalate (DEHP), based on the available test information.

Limited data indicate that acute toxicity occurs to freshwater aquatic life at a concentration as low as 2,100 µg/L, which is above the reported solubility limit for DEHP. Chronic toxicity occurs to one freshwater species at a concentration as low as 160 µg/L.

Toxicity data for DEHP and saltwater life is limited. However, if their chronic sensitivity to DEHP is similar to that of freshwater aquatic life, adverse effects on individual species might be expected at ≤160 µg/L. An ecosystem process, ammonia flux, has been shown to be reduced at 15.5 µg/L in summer months.

Human Health

Refer to phthalate esters.

DISSOLVED OXYGEN

7782-44-7

National Criteria

The national criteria for ambient dissolved oxygen concentrations for the protection of freshwater aquatic life are presented in Table 1. The criteria are derived from the production impairment estimates that are based primarily upon growth data and information on temperature, disease, and pollutant stresses. The average dissolved oxygen concentrations selected are values 0.5 mg/L above the slight production impairment values and represent values between no production impairment and slight production impairment. Each criterion may thus be viewed as an estimate of the threshold concentration below which detrimental effects are expected.

Table 1 —Water quality criteria for ambient dissolved oxygen concentration

	COLDWATER CRITERIA		WARMWATER CRITERIA	
	EARLY LIFE STAGES[1,2]	OTHER LIFE STAGES	EARLY LIFE STAGES[2]	OTHER LIFE STAGES
30 Day Mean	NA[3]	6.5	NA	5.5
7 Day Mean	9.5 (6.5)	NA	6.0	NA
7 Day Mean Minimum	NA	5.0	NA	4.0
1 Day Minimum[4,5]	8.0 (5.0)	4.0	5.0	3.0

[1]These are water column concentrations recommended to achieve the required intergravel dissolved oxygen concentrations shown in parentheses. The 3 mg/L differential is discussed in the criteria document. For species that have early life stages exposed directly to the water column, the figures in parentheses apply.
[2]Includes all embryonic and larval stages and all juvenile forms to 30 days following hatching.
[3]NA (not applicable)
[4]For highly manipulatable discharges, further restrictions apply.
[5]All minimum values should be considered as instantaneous concentrations to be achieved at all times.

Criteria for coldwater fish are intended to apply to waters containing a population of one or more species in the family Salmonidae or to waters containing other coldwater or coolwater fish deemed by the user to be closer to salmonids in sensitivity than to most warmwater species. Although the acute lethal limit for salmonids is at or below 3 mg/L, the coldwater minimum has been established at 4 mg/L because a significant proportion of the insect species common to salmonid habitats are less tolerant of acute exposures to low dissolved oxygen than are salmonids. Some coolwater species may require more protection than that afforded by the other life-stage criteria for warmwater fish, and protecting sensitive coolwater species with the coldwater criteria may be desirable.

Many States have more stringent dissolved oxygen standards for cooler waters that contain either salmonids, nonsalmonid coolwater fish, or the sensitive Centrarchidae, the smallmouth bass. The warmwater criteria are necessary to protect early life stages of warmwater fish as sensitive as channel catfish and to protect other life stages of fish as sensitive as largemouth bass. Criteria for early life stages are intended to apply only where and when these stages occur. These criteria represent dissolved oxygen concentrations that EPA believes provide a reasonable and adequate degree of protection for freshwater aquatic life.

The criteria do not represent assured no-effect levels. However, because the criteria represent worst case conditions (i.e, for wasteload

allocation and waste treatment plant design), conditions will be better than the criteria nearly all of the time at most sites. In situations where criteria conditions are just maintained for considerable periods, the proposed criteria represent some risk of production impairment. This impairment would depend on innumerable other factors. If slight production impairment or a small but undefinable risk of moderate impairment is unacceptable, then one should use the "no production impairment" values given in the document as means and the "slight production impairment" values as minimum. Table 2 presents these concentrations.

The criteria represent dissolved oxygen concentrations believed to protect the more sensitive populations of organisms against potentially damaging production impairment. The dissolved oxygen concentrations in the criteria are intended to be protective at typically high, seasonal environmental temperatures for the appropriate taxonomic and life-stage classifications, temperatures that are often higher than those used in the research from which the criteria were generated, especially for other than early life stages.

Where natural conditions alone create dissolved oxygen concentrations less than 110 percent of the applicable criteria means or minima or both, the minimum acceptable concentration is 90 percent of the natural concentration. These values are similar to those presented graphically and to those calculated from 1972 Water Quality Criteria. Absolutely no anthro-

Table 2.—Dissolved oxygen concentrations (MG/L) versus quantitative level of effect.

1. *Salmonid Waters*	
a. Embryo and Larval Stages	
No Production Impairment	= 11* (8)
Slight Production Impairment	= 9* (6)
Moderate Production Impairment	= 8* (5)
Severe Production Impairment	= 7* (4)
Limit to Avoid Acute Mortality	= 6* (3)

*Note: These are water column concentrations recommended to achieve the required intergravel dissolved oxygen concentrations shown in parentheses. The 3 mg/L difference is discussed in the criteria document.

b. Other Life Stages	
No Production Impairment	= 8
Light Production Impairment	= 6
Moderate Production Impairment	= 5
Severe Production Impairment	= 4
Limit to Avoid Acute Mortality	= 3
2. *Nonsalmonid Waters*	
a. Early Life Stages	
No Production Impairment	= 6.5
Slight Production Impairment	= 5.5
Moderate Production Impairment	= 5
Severe Production Impairment	= 4.5
Limit to Avoid Acute Mortality	= 4
b. Other Life Stages	
No Production Impairment	= 6
Slight Production Impairment	= 5
Moderate Production Impairment	= 4
Severe Production Impairment	= 3.5
Limit to Avoid Acute Mortality	= 3
3. Invertebrates	
No Production Impairment	= 8
Some Production Impairment	= 5
Acute Mortality Limit	= 4

pogenic dissolved oxygen depression in the potentially lethal area below the one-day minima should be allowed unless special care is taken to ascertain the tolerance of resident species to low dissolved oxygen.

If daily cycles of dissolved oxygen are essentially sinusoidal, a reasonable daily average is calculated from the day's high and low dissolved oxygen values. A time-weighted average may be required if the dissolved oxygen cycles are decidedly nonsinusoidal. Determining the magnitude of daily dissolved oxygen cycles requires at least two appropriately timed measurements daily, and characterizing the shape of the cycle requires several more appropriately spaced measurements.

Once a series of daily mean dissolved oxygen concentrations are calculated, an average of these daily means can be calculated (Table 3). For embryonic, larval, and early life stages, the averaging period should not exceed seven days. This short time is needed to adequately protect these often short-duration, most sensitive life stages. Other life stages can probably be adequately protected by 30-day averages. Regardless of the averaging period, the average should be considered a moving average rather than a calendar-week or calendar-month average.

Table 3.—Sample calculations for determining daily means and seven-day mean dissolved oxygen concentrations (30-day averages are calculated in a similar fashion using 30 days data).

DAY	DISSOLVED OXYGEN (MG/L)		DAILY MEAN
	DAILY MAX	DAILY MIN	
1	9.0	7.0	8.0
2	10.0	7.0	8.5
3	11.0	8.0	9.5[b]
4	12.0[a]	8.0	9.5
5	10.0	8.0	9.0
6	11.0	9.0	10.0
7	12.0[a]	10.0	10.5[c]
		57.0	65.0
1-day Minimum		7.0	
7-day Mean Minimum		8.1	
7-day Mean			9.3

[a]Above air saturation concentration (assumed to be 11.0 mg/L for this example)
[b](11.0 + 8.0)2
[c](11.0 + 10.0)2

The criteria have been established on the basis that the maximum dissolved oxygen value actually used in calculating any daily mean should not exceed the air saturation value. This consideration is based primarily on analysis of studies of cycling dissolved oxygen and the growth of largemouth bass, which indicated that high dissolved oxygen levels (6 mg/L) had no beneficial effect on growth.

During periodic cycles of dissolved oxygen concentrations, minima lower than acceptable constant exposure levels are tolerable so long as:

1. The average concentration attained meets or exceeds the criterion;
2. The average dissolved oxygen concentration is calculated as recommended in Table 3; and
3. The minima are not unduly stressful and clearly are not lethal.

A daily minimum has been included to make certain that no acute mortality of sensitive species occurs as a result of lack of oxygen. Because repeated exposure to dissolved oxygen concentrations at or near the acute lethal threshold will be stressful and because stress can indirectly produce

mortality or other adverse effects (e.g., through disease), the criteria are designed to prevent significant episodes of continuous or regularly recurring exposures to dissolved oxygen concentrations at or near the lethal threshold. This protection has been achieved by setting the daily minimum for early life stages at the subacute lethality threshold, by the use of a seven-day averaging period for early life stages, by stipulating a seven-day mean minimum value for other life stages, and by recommending additional limits for manipulatable discharges.

The previous EPA criterion for dissolved oxygen published in "Quality Criteria for Water" (1976) was a minimum of 5 mg/L (usually applied as a 7Q10), which is similar to the current criterion minimum except for other life stages of warmwater fish that now allows a seven-day mean minimum of 4 mg/L. The new criteria are similar to those contained in the 1968 "Green Book" of the Federal Water Pollution Control Federation.

The Criteria and Monitoring and Design Conditions

The acceptable mean concentrations should be attained most of the time, but some deviation below these values would probably not cause significant harm. Deviations below the mean will probably be serially correlated and hence apt to occur on consecutive days. The significance of deviations below the mean will depend on whether they occur continuously or in daily cycles, the former being more adverse than the latter. Current knowledge regarding such deviations is limited primarily to laboratory growth experiments and by extrapolation to other activity-related phenomena.

Under conditions where large daily cycles of dissolved oxygen occur, it is possible to meet the criteria mean values and consistently violate the mean minimum criteria. Under these conditions, the mean minimum criteria will clearly be the limiting regulation unless alternatives, such as nutrient control, can dampen the daily cycles.

The significance of conditions that fail to meet the recommended dissolved oxygen criteria depend largely upon five factors: (1) the duration of the event; (2) the magnitude of the dissolved oxygen depression; (3) the frequency of recurrence; (4) the proportional area of the site failing to meet the criteria; and (5) the biological significance of the site where the event occurs.

Evaluation of an event's significance must be largely case- and site-specific. Common sense would dictate that the magnitude of the depression would be the single most important factor in general, especially if the acute value is violated. A logical extension of these considerations is that the event must be considered in the context of the resolution level of the monitoring or modeling effort. Evaluating the extent, duration, and magnitude of an event must be a function of the spatial and temporal frequency of the data. Thus, a single deviation below the criterion takes on considerably less significance where continuous monitoring occurs than where sampling is comprised of once-a-week grab samples. This is so because, based on continuous monitoring, the event is provably small; but with the much less frequent sampling, the event is not provably small and can be considerably worse than indicated by the sample.

The frequency of recurrence is of considerable interest to those modeling dissolved oxygen concentrations because the return period, or period between recurrences, is a primary modeling consideration contingent upon probabilities of receiving water volumes, waste loads, temperatures, and so forth. It should be apparent that the return period cannot be isolated from the other four factors discussed above. Ultimately, the question of return period may be decided on a site-specific basis, taking into ac-

count the other factors (duration, magnitude, areal extent, and biological significance) mentioned above. Future studies of temporal patterns of dissolved oxygen concentrations, both within and between years, must be conducted to provide a better basis for selection of the appropriate return period.

In conducting wasteload allocation and treatment plant design computations, the choice of temperature in the models will be important. Probably the best option would be to use temperatures consistent with those expected in the receiving water over the critical dissolved oxygen period for the biota.

The Criteria and Manipulatable Discharges

If daily minimum DOs are perfectly serially correlated (i.e, if the annual lowest daily minimum dissolved oxygen concentration is adjacent in time to the next lower daily minimum dissolved oxygen concentration, and one of these two minima is adjacent to the third lowest daily minimum dissolved oxygen concentration, and so on), then, to meet the seven-day mean minimum criterion, more than three or four consecutive daily minimum values below the acceptable seven-day mean minimum will not likely occur. Unless the dissolved oxygen pattern is extremely erratic, it is also unlikely that the lowest dissolved oxygen concentration will be appreciably below the acceptable seven-day mean minimum, or that daily minimum values below the seven-day mean minimum will occur in more than one or two weeks each year.

For some discharges, the distribution of dissolved oxygen concentrations can be manipulated to varying degrees. Applying the daily minimum to manipulatable discharges would allow repeated weekly cycles of minimum acutely acceptable dissolved oxygen values, a condition of unacceptable stress, and possible adverse biological effect. For this reason, the application of the one-day minimum criterion to manipulatable discharges must limit either the frequency of occurrence of values below the acceptable seven-day mean minimum or must impose further limits on the extent of excursions below the seven-day mean minimum. For such controlled discharges, the occurrence of daily minima below the acceptable seven-day mean minimum should be limited to three weeks per year or the acceptable one-day minimum should be increased to 4.5 mg/L for coldwater fish and 3.5 mg/L for warmwater fish. Such decisions could be site-specific based upon the extent of control and serial correlation.

(51 F.R. 22978, June 24, 1986)

See Appendix A for Aquatic Life Methodology.

DISSOLVED SOLIDS AND SALINITY

CRITERIA

250 mg/L for chlorides and sulfates in domestic water supplies (welfare).

Introduction

Dissolved solids and total dissolved solids, terms generally associated with freshwater systems, consist of inorganic salts, small amounts of organic matter, and dissolved materials. The equivalent terminology in Standard Methods is "filtrable residue." Salinity, an oceanographic term, is not precisely equivalent to the total dissolved salt content but is related. For most purposes, the terms "total dissolved salt content" and "salinity" are equivalent. The principal inorganic anions dissolved in water include the carbonates, chlorides, sulfates, and nitrates (principally in ground waters); the principal cations are sodium, potassium, calcium, and magnesium.

(Quality Criteria for Water, July 1976) PB-263943
See Appendix D for Methodology.

ENDOSULFAN

115-29-7

CRITERIA

Aquatic Life

The criterion to protect freshwater aquatic life as derived using the guidelines is 0.056 μg/L as a 24-hour average; the concentration should not exceed 0.22 μg/L at any time.

The criterion to protect saltwater aquatic life as derived using the guidelines is 0.0087 μg/L as a 24-hour average; the concentration should not exceed 0.034 μg/L at any time.

Human Health

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60848). Recalculate IRIS values for alpha-endosulfan, beta-endosulfan and endosulfan sulfate are 0.93 μg/L for ingestion of contaminated water and organisms and 2.0 μg/L for ingestion of contaminated aquatic organisms only.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60848, December 22, 1992)
See Appendix B for Aquatic Life Methodology.
See Appendix C for Human Health Mehtodology.

*ENDRIN

72-20-8

CRITERIA

Aquatic Life

The criterion to protect freshwater aquatic life exposed to endrin, as derived using the guidelines, is 0.0023 μg/L as a 24-hour average; the concentration should not exceed 0.18 μg/L at any time.

The criterion to protect saltwater aquatic life exposed to endrin, as derived using the guidelines, is 0.0023 μg/L as a 24-hour average; the concentration should not exceed 0.037 μg/L at any time.

Human Health

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60848). Recalculated IRIS values for both endrin and endrin aldehyde are 0.76 μg/L for ingestion of contaminated water and organisms and 0.81 μg/L for ingestion of contaminated aquatic organisms only.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60848, December 22, 1992)

See Appendix B for Aquatic Life Methodology.

See Appendix C for Human Health Mehtodology.

*Indicates suspension, cancelation, or restriction by U.S. EPA Office of Pesticides and Toxic Substances.

ETHYLBENZENE

100-41-4

CRITERIA

Aquatic Life

The available data for ethylbenzene indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 32,000 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No definitive data are available concerning the chronic toxicity of ethylbenzene to sensitive freshwater aquatic life.

The available data for ethylbenzene indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 430 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of ethylbenzene to sensitive saltwater aquatic life.

Human Health

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60848). Recalculated IRIS values for ethylbenzene are 3,100 µg/L for ingestion of contaminated water and organisms and 29,000 µg/L for ingestion of contaminated aquatic organisms only.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60912, December 22, 1992)
See Appendix C for Human Health Methodology.

FLUORANTHENE

206-44-0

CRITERIA

Aquatic Life

The available data for fluoranthene indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 3,980 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of fluoranthene to sensitive freshwater aquatic life.

The available data for fluoranthene indicate that acute and chronic toxicity to saltwater aquatic life occur at concentrations as low as 40 and 16 µg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested.

Human Health

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60848). Recalculated IRIS values for fluoranthene are 300 µg/L for ingestion of contaminated water and organisms and 370 µg/L for ingestion of contaminated organisms only.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60848, December 22, 1992)
See Appendix C for Human Health Methodology.

GASES, TOTAL DISSOLVED

CRITERIA

Aquatic Life

To protect freshwater and saltwater aquatic life, the total dissolved gas concentrations in water should not exceed 110 percent of the saturation value for gases at the existing atmospheric and hydrostatic pressures.

Rationale

Fish in water containing excessive dissolved gas pressure or tension are killed when dissolving gases in their circulatory system come out of solution to form bubbles (emoli) that block the flow of blood through the capillary vessels. In aquatic organisms this is commonly referred to as "gas bubble disease." External bubbles (emphysema) also appear in the fins, on the opercula, in the skin, and in other body tissues. Aquatic invertebrates are also affected by gas bubble disease but usually at supersaturation levels higher than those lethal to fish.

Percent saturation of water containing a given amount of gas varies with the absolute temperature and the pressure. Because of the pressure changes, percent saturation with a given amount of gas changes with the water depth. Gas supersaturation decreases by 10 percent per meter of increase in water depth due to hydrostratic pressure; a gas that is at 130 percent saturation at the surface would be at 100 percent saturation at 3 meters' depth. Compensation for altitude may be needed because a reduction in atmostpheric pressure changes the water/gas equilibrium, resulting in changes in solubility of dissolved gases.

Total dissolved gas supersaturation can occur in several ways:

1. Excessive biological activity: dissolved oxygen (DO) concentrations can reach supersaturation as a result of excessive algal photosynthesis. Algal blooms are often accompanied by increased water temperatures, which further contribute to supersaturation.

2. Water spillage from hydropower dams causes supersaturation.

3. Gas bubble disease may be induced by discharges from power-generating and other thermal sources. Discharged water becomes supersaturated with gases.

In recent years, gas bubble disease has been identified as a major problem affecting valuable stocks of salmon and trout in the Columbia River system. The disease is caused by high concentrations of dissolved atmospheric gas which enters the river's water during heavy spilling at hydroelectric dams.

Field and laboratory reports result in several conclusions:

1. When either juvenile or adult salmonids are confined to shallow water (1M), substantial mortality occurs at and above 115 percent total dissolved gas saturation.

2. When either juvenile or adult salmonids are free to sound and obtain hydrostatic compensation either in the laboratory or in the field, substantial mortality still occurs when saturation levels (of total dissolved gases) exceed 120 percent saturation.

3. Using survival estimates made in the Snake River from 1966 to 1975, it is concluded that juvenile fish losses ranging from 40 to 95 percent do occur. A major portion of this mortality can be attributed to fish exposure to supersaturation by atmospheric gases during years of high flow.

4. Juvenile salmonids subjected to sublethal periods of exposure to supersaturation can recover when returned to normally saturated water, but adults do not recover and generally die from direct and indirect effects of the exposure.

5. Some species of salmon and trout can detect and avoid supersaturated waters; others may not.

6. Higher survival was observed during periods of intermittent exposure than during continuous exposure.

7. In general, in acute bioassays, salmon and trout were less tolerant than the nonsalmonids.

Interested individuals should review the original document for associated references and reports. This document is available from the National Technical Information Service (NTIS). See Appendix F for ordering information.

No differences are proposed in the criteria for freshwater and marine aquatic life, as available data indicate little difference in overall tolerance between saltwater and freshwater species.

(Quality Criteria for Water, July 1976) PB-263943
See Appendix D for Methodology.

GUTHION

86-50-0

CRITERIA

Aquatic Life

.01 μg/L for freshwater and saltwater aquatic life.

Rationale

Four-day LC50 values for fish exposed to the organophosphate pesticide range from 4 to 4,270 μg/L. Decreased spawning was documented in fathead minnows exposed to low levels during complete life-cycle exposures. The estimated "safe" long-term concentration for this species is 0.3 μg/L to 0.5 μg/L.

Organophosphate pesticides inhibit the enzyme acetylchlolinesterase (AChE), which is essential to nerve impulse transport. Inhibition of 40 to 70 percent of fish brain AChE is usually fatal. Centrarchids are considered one of the most sensitive fish to guthion.

Four-day LC50 values for aquatic invertebrates range from 0.10 μg/L to 22.0 μg/L, indicating an overall greater sensitivity than fish and a narrower spectrum of tolerance across species.

Results of toxicity studies with marine organisms indicate similar responses, with saltwater invertebrates exhibiting LC50s as low as 0.33 μg/L.

A criterion level of 0.01 μg/L for guthion is based on the use of 0.1 as an application factor, applied to the 96-hour LC50 of 0.1 μg/L (for the amphipod, *Gammarus*) and a similar value of 0.3 μg/L, exhibited by the European shrimp.

(Quality Criteria for Water, July 1976) PB-263943
See Appendix D for Methodology.

HALOETHERS

CRITERIA

Aquatic Life

The available data for haloethers indicate that acute and chronic toxicity to freshwater aquatic life occurs at concentrations as low as 360 and 122 µg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested.

No saltwater organisms have been tested with any haloether, and therefore, no statement can be made concerning acute or chronic toxicity.

Human Health

Using the present guidelines, a satisfactory criterion cannot be derived at this time because of insufficient available data for haloethers. See also chloroalkyl ethers.

(45 F.R. 79318, November 28, 1980)
See Appendix B for Aquatic Life Methodology.

HALOMETHANES

CRITERIA

Aquatic Life

The available data for halomethanes indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 11,000 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of halomethanes to sensitive freshwater aquatic life.

The available data for halomethanes indicate that acute and chronic toxicity to saltwater aquatic life occurs at concentrations as low as 12,000 and 6,400 µg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested. A decrease in algal cell numbers occurs at concentrations as low as 11,500 µg/L.

Human Health

For the maximum protection of human health from the potential carcinogenic effects of exposure through the ingestion of contaminated water and aquatic organisms for bromoform, dichlorobromomethane, and methylene chloride, the ambient water concentration should be zero. However, zero levels may not be attainable at the present time. Therefore, the levels that may result in incremental increase of cancer risk over a lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}.

Bromoform (Tribromomethane) 75-25-2

Human health criteria were recalculated using the Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60848). Recalculated IRIS values for bromoform are 4.3 µg/L for ingestion of contaminated water and organisms and 360 µg/L for ingestion of contaminated organisms only. IRIS values are based on a 10^{-6} risk level for carcinogens.

Dichlorobromomethane 75-27-4

Human health criteria were recalculated using the IRIS to reflect available data as of 12/92 (57 F.R. 60848). Recalculated IRIS values for dichlorobromomethane are 0.27 µg/L for ingestion of contaminated water and organisms and 22 µg/L for ingestion of contaminated organisms only. IRIS values are based on a 10^{-6} risk level for carcinogens.

Methylene Chloride 75-09-2

Human health criteria were recalculated using the IRIS to reflect available data as of 12/92 (57 F.R. 60848). Recalculated IRIS values for methylene chloride are 4.7 µg/L for ingestion of contaminated water and organisms and 1,600 µg/L for ingestion of contaminated organisms only. IRIS values are based on a 10^{-6} risk level for carcinogens.

Methyl Chloride (Chloromethane) 74-87-3

Human health criteria have been withdrawn for this compound (57 F.R. 60848, December 22, 1992). However, EPA published a document,

"Ambient Water Quality Criteria for Halomethanes," which includes this compound. This document may contain useful information on human health and was originally noticed on December 22, 1992.

Methyl Bromide (Bromomethane) 74-83-9

Human health criteria were recalculated using the IRIS to reflect available data as of 12/92 (57 F.R. 60848). Recalculated IRIS values for methyl bromide are 48 μg/L for ingestion of contaminated water and organisms and 4000 μg/L for ingestion of contaminated organisms only.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60848, December 22, 1992)
See Appendix C for Human Health Methodology.

HARDNESS

CRITERIA

Introduction

Water hardness is caused by polyvalent metallic ions dissolved in water. In fresh water, these metallic ions are primarily calcium and magnesium, although other metals such as iron, strontium, and manganese can also be present in appreciable concentrations. Commonly, hardness is reported as an equivalent concentration of calcium carbonate ($CaCO_3$).

The concept of hardness comes from water supply practices: it is measured by soap requirements for adequate lather formation and as an indicator of the rate of scale formation in hot water heaters and low pressure boilers. A commonly used classification is given in Table 1.

Table 1.—Classification of water by hardness content.

MAXIMUM CONCENTRATION MG/L AS $CACO_3$	WATER DESCRIPTION
0– 75	soft
75–150	moderately hard
150–300	hard
300 and up	very hard

The principal natural source of hardness is limestone, which is dissolved by percolating rainwater made acid by carbon dioxide. Industrial and industrially related sources of hardness include the inorganic chemical industry and discharges from operating and abandoned mines.

Hardness in fresh water frequently is distinguished in carbonate and noncarbonate fractions: carbonate fractions are chemically equivalent to the bicarbonates present in water. Since bicarbonates generally are measured as alkalinity, the carbonate hardness is usually considered equal to the alkalinity.

Rationale

The determination of hardness in raw waters subsequently treated and used for domestic water supplies is useful as a parameter to characterize the total dissolved solids present and for calculating dosages where lime-soda softening is practiced. Because hardness concentrations in water have not been proven health related, the final level achieved is principally a function of economics. Since hardness in water can be removed with treatment by such processes as lime-soda softening and zeolite or ion exchange systems, a criterion for raw waters used for public water supply is not practical.

The effects of hardness on freshwater fish and other aquatic life appear to be related to the ions causing the hardness rather than hardness. Both the National Technical Advisory Committee (NTAC) and The National Academy of Sciences (NAS) panels have recommended against using the term hardness but suggest including the concentrations of the specific ions. This procedure should avoid confusion in future studies but is not helpful

in evaluating previous studies. For most existing data, it is difficult to determine whether toxicity of various metal ions is reduced because of the formation of metallic hydroxides and carbonates caused by the associated increases in alkalinity, or because of an antagonistic effect of one of the principal cations contributing to hardness — e.g., calcium — or a combination of both effects. One theory presented, without proof, that if cupric ions were the toxic form of copper, whereas copper carbonate complexes were relatively non-toxic, then the observed difference in toxicity of copper between hard and soft waters can be explained by the difference in alkalinity rather than hardness. A review of the literature on toxicity presented data showing that increasing calcium, in particular, reduced the toxicity of other heavy metals. Under usual conditions in fresh water and assuming that other bivalent metals behave similarly to copper, we can assume that both effects occur simultaneously and explain the observed reduction of toxicity of metals in waters containing carbonate hardness. The amount of reduced toxicity related to hardness, as measured by a 40-hour LC50 for rainbow trout, has been estimated to be about four times for copper and zinc when the hardness was increased from 10 to 100 mg/L as $CaCO_3$.

Limits on hardness for industrial uses are quite variable. Table 2 lists maximum values accepted by various industries as a source of raw water. Subsequent treatment generally can reduce harness to tolerable limits, although costs of such treatment are an important factor in determining its desirability for a particular water source.

Hardness is not a determination of concern for irrigation water use. The concentrations of the cations calcium and magnesium, which comprise hardness, are important in determining the exchangeable sodium in a given water. This particular calculation will be discussed under total dissolved solids rather than hardness.

Table 2.—Maximum hardness levels accepted by industry as a raw water source.*

INDUSTRY	MAXIMUM CONCENTRATION MG/L AS $CACO_3$
Electric utilities	5,000
Textile	120
Pulp and paper	475
Chemical	1,000
Petroleum	900
Primary metals	1,000

*Requirements for final use within a process may be essentially zero, which requires treatment for concentration reductions

(Quality Criteria for Water, July 1976) PB-263943
See Appendix D for Methodology.

HEPTACHLOR

76-44-8

CRITERIA

Aquatic Life

The criterion to protect freshwater aquatic life for heptachlor and heptachlor epoxide, as derived using the guidelines, is 0.0038 μg/L as a 24-hour average, and the concentration should not exceed 0.52 μg/L at any time.

The criterion to protect saltwater aquatic life for heptachlor and heptachlor expoxide, as derived using the guidelines, is 0.0036 μg/L as a 24-hour average. The concentration should not exceed 0.053 μg/L at any time.

Human Health

For the maximum protection of human health from potential carcinogenic effects from exposure to heptachlor through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero, based on the nonthreshold assumption for this chemical; however, zero level may not be attainable. Therefore, the levels that may result in incremental increase of cancer risk over a lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}.

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60848). Recaculated IRIS values for heptachlor is 0.00021 μg/L for ingestion of contaminated water and organisms and for ingestion of contaminated aquatic organisms only. IRIS values are based on a 10^{-6} risk level for carcinogens.

Published human health criteria were recalculated using IRIS to reflect available data as of 12/92 (57 F.R. 60848). Recalculated IRIS values for heptachlor epoxide are 0.00010 μg/L for ingestion of contaminated water and organisms and 0.00011 μg/L for ingestion of contaminated aquatic organisms only. IRIS values are based on a 10^{-6} risk level for carcinogens.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60848, December 22, 1992)

See Appendix B for Aquatic Life Methodology.

See Appendix C for Human Health Methodology.

HEXACHLOROBENZENE

118-74-1

CRITERIA

Aquatic Life

As of 8/88, a draft Ambient Water Quality Criteria (AWQC) document for hexachlorobenzene became available. Final rulemaking will eventually be promulgated, but as of this writing, aquatic life criteria for hexachlorobenzene has not been finalized.

Human Health

Refer to Chlorinated Benzenes.

HEXACHLOROBUTADIENE

86-68-3

CRITERIA

Aquatic Life

The available data for hexachlorobutadiene indicate that acute and chronic toxicity to freshwater aquatic life occur at concentrations as low as 90 and 9.3 μg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested.

The available data for hexachlorobutadiene indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 32 μg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of hexachlorobutadiene to sensitive saltwater aquatic life.

Human Health

For the maximum protection of human health from the potential carcinogenic effects of exposure to hexachlorobutadiene through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentrations should be zero, based on the nonthreshold assumption for this chemical; however, zero level may not be attainable. Therefore, the levels that may result in incremental increase of cancer risk over a lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}.

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect avaialable data as of 12/92 (57 F.R. 60848). Recalculated IRIS values for hexachlorobutadiene are 0.44 μg/L for ingestion of contaminated water and organisms and 50 μg/L for ingestion of contaminated aquatic organisms only. IRIS values are based on a 10^{-6} risk level for carcinogens.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60848, December 22, 1992)
See Appendix C for Human Health Methodology.

HEXACHLOROCYCLOHEXANE

58-89-9

CRITERIA

Aquatic Life

Gamma-hexachlorocyclohexane (Lindane) 58-89-9

For gamma-hexachlorocyclohexane (lindane), the criterion to protect freshwater aquatic life as derived using the guidelines is 0.080 µg/L as a 24-hour average. The concentration should not exceed 2.0 µg/L at any time.

For saltwater aquatic life the concentration of lindane should not exceed 0.16 µg/L at any time. No data are available for lindane's chronic toxicity to sensitive saltwater aquatic life.

BHC 680-73-1

The available data for a mixture of isomers of benzene hexachloride (BHC) indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 100 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of a mixture of isomers of BHC to sensitive freshwater aquatic life.

The available data for a mixture of isomers of BHC indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 0.34 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of a mixture of isomers of BHC to sensitive saltwater aquatic life.

Human Health

Alpha-hexachlorocyclohexane 319-84-6

For the maximum protection of human health from the potential carcinogenic effects of exposure to alpha-hexachlorocyclohexane through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentrations should be zero, based on the nonthreshold assumption for this chemical; however, zero level may not be attainable. Therefore, the levels that may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}.

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60848). Recalculated IRIS values for hexachlorocyclohexane-alpha are 0.0039 for ingestion of contaminated water and organisms and 0.013 for ingestion of contaminated aquatic organisms only. IRIS values are based on a 10^{-6} risk level for carcinogens.

Beta-hexachlorocyclohexane 319-85-7

For the maximum protection of human health from the potential carcinogenic effects of exposure to beta-hexachlorocyclohexane through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentrations should be zero, based on the nonthreshold assumption for this chemical; however, zero level may not be attainable. Therefore, the levels that may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}. The corresponding recommended criteria are 163 ng/L, 16.3 ng/L, and 1.63 ng/L, respectively. If these estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 547 ng/L, 54.7 ng/L, and 5.47 ng/L, respectively.

Published human health criteria were recalculated using IRIS to reflect available data as of 3/91. Recalculated IRIS values for hexachlorocyclohexane-beta are 0.014 μg/L for ingestion of contaminated water and organisms and 0.046 μg/L for ingestion of contaminated aquatic organisms only. IRIS values are based on a 10^{-6} risk level for carcinogens.

Gamma-hexachlorocyclohexane (Lindane) 58-89-9

For the maximum protection of human health from the potential carcinogenic effects due to exposure of gamma-hexachlorocyclohexane through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentrations should be zero, based on the nonthreshold assumption for this chemical; however, zero level may not be attainable. Therefore, the levels that may result in incremental increase of cancer risk over a lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}. The corresponding recommended criteria are 186 ng/L, 18.6 ng/L, and 1.86 ng/L, respectively. If these estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 625 ng/L, 62.5 ng/L, and 6.25 ng/L, respectively.

Published human health criteria were recalculated using IRIS to reflect available data as of 3/91. Recalculated IRIS values for hexachlorocyclohexane-gamma (Lindane) are 0.019 μg/L for ingestion of contaminated water and organisms and 0.063 for ingestion of contaminated aquatic organisms only. IRIS values are based on a 10^{-6} risk level for carcinogens.

Technical-hexachlorocyclohexane 319-86-8

Using the present guidelines, satisfactory criteria cannot now be derived for delta and epsilon hexachlorocyclohexane because of insufficient data.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60848, December 22, 1992)

See Appendix B for Aquatic Life Methodology.

See Appendix C for Human Health Methodology.

HEXACHLOROCYCLOPENTADIENE

77-47-4

CRITERIA

Aquatic Life

The available data for hexachlorocyclopentadiene indicate that acute and chronic toxicity to freshwater aquatic life occurs at concentrations as low as 7.0 and 5.2 μg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested.

The available data for hexachlorocyclopentadiene indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 7.0 μg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of hexachlorocyclopentadiene to sensitive saltwater aquatic life.

Human Health

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60848)..Recalculated IRIS values for hexachlorocyclopentadiene are 240.0 μg/L for ingestion of contaminated water and organisms and 17,000. μg/L for ingestion of contaminated aquatic organisms only.

Using available organoleptic data, the estimated level is 1 μg/L to control undesirable taste and odor quality of ambient water. Organoleptic data have limitations as a basis for establishing water quality criteria but no demonstrated relationship to potentially adverse effects on human health.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60848, December 22, 1992)
See Appendix C for Human Health Methodology.

IRON

7439-89-6

CRITERIA

Aquatic Life

0.3 mg/L for domestic water supply (health).
1.0 mg/L for freshwater aquatic life.

Introduction

Of the elements that make up the earth's crust, iron is the fourth most abundant by weight. Common in many rocks, iron is an important component of many soils but especially clay, where it is usually a major constituent. Iron may be present in varying quantities in water, depending upon the area's geology and the waterway's other chemical components.

Iron is an essential trace element required by both plants and animals. In some waters, it may be a limiting factor for the growth of algae and other plants, especially in some marl lakes where it is precipitated by the highly alkaline conditions. Also, iron is a vital oxygen transport mechanism in the blood of all vertebrate and some invertebrate animals.

The ferrous, or bivalent (Fe^{++}), and the ferric, or trivalent (Fe^{+++}), irons are of primary concern in the aquatic environment, although other forms of iron may occur in organic and inorganic wastewater streams. The ferrous (Fe^{++}) form can persist in waters void of dissolved oxygen and originates usually from groundwaters or mines that have been pumped or drained. For practical purposes, the ferric (Fe^{+++}) form is insoluble. Iron can exist in natural organometallic or humic compounds and colloidal forms. Black or brown swamp waters can contain iron concentrations of several mg/L in the presence or absence of dissolved oxygen, but this form of iron has little effect on aquatic life.

Soluble ferrous iron occurs in the deep waters of stratified lakes with anaerobic hypolimnia. During the autumnal or vernal overturns and with aeration of these lakes, it is oxidized rapidly to the ferric ion that precipitates to the bottom sediments as a hydroxide, $Fe(OH)_3$, or with other anions. If hydrogen sulfide (H_2S) is present in anaerobic bottom waters or muds, ferrous sulfide (FeS) may be formed. Ferrous sulfide is a black compound and produces black mineral muds.

Prime iron pollution sources are industrial wastes, mine drainage waters, and iron-bearing groundwaters. In the presence of dissolved oxygen, iron in water from mine drainage is precipitated as a hydroxide, $Fe(OH)_3$. These yellowish or ochre precipitates produce "yellow boy" deposits found in many streams draining coal mining regions of Appalachia. Occasionally, ferric oxide (Fe_2O_3) is precipitated, forming red waters. Both of these precipitates form as gels or flocs that may be detrimental to fishes and other aquatic life when suspended in water. These precipitates can settle to form flocculent materials that cover stream bottoms, thereby destroying bottom-dwelling invertebrates, plants, or incubating fish eggs. With time these flocs can consolidate to form cement-like materials, thus consolidating bottom gravels into pavement-like areas unsuitable as spawning sites for nest-building fishes. This is particularly detrimental to

trout and salmon populations whose eggs are protected in the interstices of gravel and incubated with oxygen-bearing waters passing through the gravel.

Rational

Iron is an objectional constituent in water supplies for both domestic and industrial use. Iron appreciably affects the taste of beverages and can stain laundered clothes and plumping fixtures. A study by the Public Health Service (see original document) indicates that the taste of iron may be detected readily at 1.8 mg/L in spring water and at 3.4 mg/L in distilled water.

The daily nutritional requirement for iron is 1 to 2 mg, but intake of larger quantities is required as a result of poor absorption. Diets contain 7 to 35 mg per day and average 16 mg. The iron criterion in water to prevent objectionable tastes or laundry staining (0.3 mg/L) constitutes only a small fraction of the iron normally consumed and is of aesthetic rather than toxicological significance.

Studies obtain 96-hour LC50 values of 0.32 mg/L iron for mayflies, stoneflies, and caddisflies; all are important fish food organisms. Other studies found iron toxic to carp *(Cyrinus carpio)* at concentrations of 0.9 mg/L when the pH of the water was 5.5. Pike *(Esox lucius)* and trout (species unknown) died at iron concentrations of 1 to 2 mg/L. In an iron polluted Colorado stream, neither trout nor other fish were found until the waters were diluted or the iron had precipitated to effect a concentration of less than 1.0 mg/L, even though other water quality constituents were suitable for the presence of trout.

Ferric hydroxide flocs have been observed to coat the gills of white perch *(Morone americanus)*, minnows, and silversides *(Menidia sp)*. The smothering effects of settled iron precipitates may be particularly detrimental to fish eggs and bottom-dwelling fish food organisms. Iron deposits in the Brule River, Michigan and Wisconsin, were found to have a residual long-term effect on fish food organisms even after the pumping of iron-bearing waters from deep shaft iron mines had ceased. Settling iron flocs have also been reported to trap and carry diatoms downward in waters.

In 69 of 75 study sites with good fish fauna, the iron concentrations were less than 10.0 mg/L. The European Inland Fisheries Commission recommended that iron concentrations not exceed 1.0 mg/L in waters to be managed for aquatic life.

Based principally on field observations, a criterion of 1 mg/L iron for freshwater aquatic life is believed to be adequately protective. As noted, data obtained under laboratory conditions suggest a greater toxicity for iron than that obtained in natural ecosystems. Ambient natural waters will vary according to alkalinity, pH, hardness, temperature, and the presence of ligands, which change the valence state and solubility and therefore the toxicity of the metal.

The effects of iron on marine life have not been investigated adequately to determine a water quality criterion. Dissolved iron readily precipitates in alkaline seawaters. Fears have been expressed that these settled iron flocs may have adverse effects on important benthic commercial mussels and other shellfish resources.

Iron has not been reported to have a direct effect on the recreational uses of water, other than its effect on aquatic life. Suspended iron precipitates may interfere with swimming and be aesthetically objectionable with water deposits as yellow ochre or reddish iron oxides.

Iron at exceedingly high concentrations has been reported to be toxic to livestock and interfere with the metabolism of phosphorus. Dietary supplements of phosphorus can be used to overcome this metabolic deficiency. In aerated soils, iron in irrigation waters is not toxic. Precipitated iron may be complex phosphorus and molybdenum, making them less available as plant nutrients. In alkaline soils, iron may be so insoluble as to be deficient as a trace element and result in chlorosis, an objectionable plant nutrient deficiency disease. A reported reduction in the quality of tobacco was due to precipitate iron oxides on the leaves when the crop was spray irrigated with water containing 5 mg/L of soluble iron.

For some industries, iron concentrations in process waters lower than that required for public water supplies are required or desirable. Examples include high pressure boiler feed waters; scouring, bleaching, and dyeing of textiles; certain types of paper production; some chemicals; some food processing; and leather finishing industries.

(Quality Criteria for Water, July 1976) PB-263943
See Appendix D for Methodology.

ISOPHORONE

78-59-1

CRITERIA

Aquatic Life

The available data for isophorone indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 117,000 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of isophorone to sensitive freshwater aquatic life.

The available data for isophorone indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 12,900 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of isophorone to sensitive saltwater aquatic life.

Human Health

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60848). Recalculated IRIS values for isophorone are 8.4 µg/L for ingestion of contaminated water and organisms and 600 for ingestion of contaminated aquatic organisms only.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60848, December 22, 1992)
See Appendix C for Human Health Methodology.

LEAD

7439-92-1

CRITERIA

Aquatic Life

Saltwater — 1-hour average of 220 μg/L
4-day average of 8.5 μg/L
Freshwater criteria are hardness dependent. See text.

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 10 species range from 142.5 μg/L for an amphipod to 235,900 μg/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 μg/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 μg/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 μg/L for the mummichog to 27,000 μg/L for the soft shell clam. A chronic toxicity test was conducted with a mysid; unacceptable effects were observed at 37 μg/L but not at 17 μg/L; the acute-chronic ratio for this species is 124.8. A species of macroalgae was affected at 20 μg/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 μg/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 μg/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 μg/L, respectively, and the one-hour average concentrations are 34, 82, and 200 μg/L.

The procedures described in the guidelines 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 8.5* μg/L more than

once every three years on the average and if the one-hour average concentration does not exceed 220* μg/L more than once every three years on the average.

The recommended exceedence frequency of three years is the Agency's best scientific judgment of the average time needed for an unstressed system to recover from a pollution event in which lead exposure exceeds the criterion. A stressed system — for example one, in which several outfalls occur in a limited area — would be expected to require more recovery time. 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 selecting of an appropriate wasteload allocation model. Dynamic models are preferred for applying 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 design flow, and 7Q5 or 7Q10 for the Criterion Continuous Concentration design flow in steady-state models for unstressed and stressed systems, respectively. These matters are discussed in more detail in EPA's "Technical Support Document for Water Quality-Based Toxics Control."

Human Health

Human health criteria have been withdrawn for this compound (see 57 F.R. 60885, December 22, 1992). Although the human health criteria are withdrawn, EPA published a document for this compound that may contain useful human health information. This document was originally noticed in 45 F.R. 79331, November 28, 1980.

(45 F.R. 79318 November 28,1980) (50 F.R. 30784, July 29, 1985)
See Appendix A for Aquatic Life Methodology.

* Saltwater lead concentrations are based on a recalculation. See 57 F.R. 60882, December 22, 1992, Comment #45.

MALATHION

121-75-5

CRITERIA

Aquatic Life

0.1 μg/L for freshwater and saltwater aquatic life.

Rationale

Salmonids and centrarachids appear to be the most sensitive freshwater fish to malathion. Documented 96-hour LC50s ranged from 120 to 265 μg/L for four different salmonid species and 101 to 285 μg/L for three species of centrarchids. Reported 96-hour LC50s for rainbow trout (*Onchorhynchus mykiss*), largemouth bass (*Micropterus salmoides*), and chinook salmon, (*Onchorhynchus tshawytcha*) were 86, 50, and 28 μg/L, respectively.

Freshwater invertebrates are generally even more sensitive to malathion than fish. Reported 96-hour LC50s for amphipod *Gammarus lacustris* and *G. fasciutus* were 1.0 and 0.76 μg/L, respectively. The 48-hour LC50s for the cladocerans *Simocephalus serrulatus* and *Daphnia pulex* ranged from 1.8 to 3.5 μg/L, while the 24-hour LC50s for two midge larvae species averaged just over two μg/L in tests with malathion.

In flow-through exposures to malathion with saltwater fish (*Lagodon rhombides*), 575 μg/L result in a 50 percent mortality rate in 3.5 hours and caused about 75 percent brain acetylcholinesterase (AChE) inhibition. Similar mortality and AChE inhibition is documented with other saltwater fish as well. Static 96-hour tests for saltwater teleosts exposed to malathion indicates a broad spectrum of species sensitivities, with LC50 values ranging from 27 to 3,250 μg/L for several different species. For the commercially and economically important striped bass, *Morone saxatilis*, a flow-through 96-hour LC50 of 14 μg/L is documented.

Saltwater invertebrates are comparably sensitive to the less-resistant fish species with 96-hour LC50s ranging from 33 μg/L for sand shrimp (*Crangon septemspinosa*) to 83 μg/L for the hermit crab (*Pagurus longicorpus).*

Malathion enters the aquatic environment primarily as a result of its application as an insecticide. Because it degrades quite rapidly in most waters, depending on pH, its occurrence is sporadic rather than continous. Because malathion's toxicity is exerted through inhibition of the enzyme AChE and because such inhibition may be additive with repeated exposures and may be caused by any of the organophosphorus insecticides, inhibition of AChE by more than 35 percent may be expected to result in damage to aquatic organisms.

An application factor of 0.1 is applied to the 96-hour LC50 data for *Gammarus lacustris, G. fasciatus,* and *Daphnia,* which are all approximately 1.0 μg/L, yielding a criterion of 0.1μg/L.

(Quality Criteria for Water, July 1976) PB-263943

MANGANESE

7439-96-5

CRITERIA

50 μg/L for domestic water supply (health).
100 μg/L for protection of consumers of saltwater molluscs.

Introduction

Manganese does not occur naturally as a metal but is found in various salts and minerals that are frequently in association with iron compounds. The principal manganese-containing substances are manganese dioxide (MnO_2), pyrolusite, manganese carbonate (rhodocrosite), and manganese silicate (rhodonite).

The oxides are the only important minerals mined. Manganese is not mined in the United States, except when contained in iron ores that are deliberately used to form ferro-manganese alloys.

The primary uses of manganese are in metal alloys, dry cell batteries, micro-nutrient fertilizer additives, organic compounds used in paint driers, and as chemical reagents. Permanganates are very strong oxidizing agents of organic materials.

Manganese is a vital micro-nutrient for both plants and animals. When manganese is not present in sufficient quantities, plants exhibit chlorosis (a yellowing of the leaves) or failure of proper leaf development. Inadequate quantities of manganese in domestic animal food results in reduced reproductive capabilities and deformed or poorly maturing young. Livestock feeds usually have sufficient manganese, but beef cattle on a high corn diet may require a supplement.

Rationale

Although inhaled manganese dusts have been reported to be toxic to humans, manganese normally is ingested as a trace nutrient in food. The average human intake is approximately 10 mg per day. Very large doses of ingested manganese can cause some disease and liver damage, but these are not known to occur in the United States. Only a few manganese toxicity problems have been found throughout the world, and these have occurred under unique circimstances (i.e., a well in Japan near a deposit of buried batteries).

It is possible to partially sequester manganese with special treatment, but manganese is not removed in the conventional treatment of domestic waters. Consumer complaints arise when manganese exceeds a concentration of 150 μg/L in water supplies. These complaints are concerned primarily with the brownish staining of laundry and objectionable tastes in beverages. The presence of low concentrations of iron may intensify the adverse effects of manganese. Manganese at concentrations of about 10 to 20 μg/L is acceptable to most consumers. A criterion for domestic water supplies of 50 μg/L should minimize the objectionable qualities.

Ions of manganese are found rarely at concentrations above 1 mg/L in freshwater. The reported tolerance values range from 1.5 mg/L to over 1,000 mg/L. Thus, manganese is not considered to be a problem in fresh

waters. Permanganates have been reported to kill fish in 8 to 18 hours at concentrations of 2.2 to 4.1 mg/L. Permanganates are not persistent because they rapidly oxidize organic materials and are thereby reduced and rendered nontoxic.

Few data are available on the toxicity of manganese to marine organisms. The ambient concentration of manganese is about 2 μg/L. The material is rapidly assimulated and bioconcentrated into nodules that are deposited on the sea floor. The major problem with manganese may be concentration in the edible portions of mollusks, as bioaccumulation factors as high as 12,000 have been reported. In order to protect against a possible health hazard to humans by manganese accumulation in shellfish, a criterion of 100 μg/L is recommended for marine water.

Manganese is not known to be a problem in water consumed by livestock. At concentrations of slighly less than 1 mg/L to a few milligrams per liter, manganese may be toxic to plants from irrigation water applied to soils with pH values lower than 6.0. The problem may be rectified by liming soils to increase the pH. Problems may develop with long-term (20-year) continuous irrigation on other soils with water containing about 10 mg/L of manganese. But as stated previously, manganese rarely is found in surface waters at concentrations greater than 1 mg/L. Thus, no specific criterion for manganese in agricultural waters is proposed. In select areas and where acidophilic crops are cultivated and irrigated, a criterion of 200 μg/L is suggested for consideration.

Most indrustrial users of water can operate successfully where the criterion proposed for public water supplies is observed. However, a more restrictive criterion may be needed to protect or ensure product quality.

(Quality Criteria for Water, July 1976) PB-263943
See Appendix D for Methodology.

*MERCURY

7439-97-6

CRITERIA

Not to exceed 2.4 μg/L in fresh water or 2.1 μg/L in salt water.
0.012 and 0.025 μg/L for freshwater and saltwater aquatic life, respectively.

AQUATIC LIFE SUMMARY

Data are available on the acute toxicity of mercury (II) to 28 genera of freshwater animals. Acute values for invertebrate species range from 2.2 μg/L for *Daphnia pulex* to 2,000 μg/L for three insects. Acute values for fishes range from 30 μg/L for the guppy to 1,000 μg/L for the *Mozambique tilapia*. Few data are available for various organomercury compounds and mercurous nitrate, and they all appear to be 4 to 31 times more acutely toxic than mercury (II).

Available chronic data indicate that methylmercury is the most chronically toxic of the tested mercury compounds. Tests on methylmercury with *Daphnia magna* and brook trout (*Salvelinus fontinalis*) produced chronic values less than 0.07 μg/L. For mercury (II) the chronic value obtained with *D. magna* was about 1.1 μg/L and the acute-chronic ratio was 4.5. In both a life-cycle test and an early life-stage test on mercuric chloride with the fathead minnow (*Pinnephales promelas*), the chronic value was less than 0.26 μg/L, and the acute-chronic ratio was over 600.

Freshwater plants show a wide range of sensitivities to mercury, but the most sensitive appear to be less affected than the most sensitive freshwater animals to both mercury (II) and methylmercury. A bioconcentration factor of 4,994 is available for mercury (II), but the bioconcentration factors for methylmercury range from 4,000 to 85,000.

Data on the acute toxicity of mercuric chloride are available for 29 genera of saltwater animals, including annelids, molluscs, crustaceans, echinoderms, and fishes. Acute values range from 3.5 μg/L for a mysid to 1,678 μg/L for winter flounder (*Pseudo pleuroneotes americanus*). Fishes tend to be more resistant, and molluscs and crustaceans tend to be more sensitive to the acute toxic effects of mercury (II). Results of a life-cycle test with the mysid show that mercury (II) at a concentration of 1.6 μg/L significantly affected time of first spawn and productivity; the resulting acute-chronic ratio was 3.1.

Concentrations of mercury that affected growth and photosynthetic activity of one saltwater diatom and six species of brown algae range from 10 to 160 μg/L. Bioconcentration factors of 10,000 and 40,000 have been obtained for mercuric chloride and methylmercury with an oyster.

National Criteria

Derivation of a water quality criterion for mercury is more complex than for most metals because of methylation of mercury in sediments, fish, and their food chain. Apparently, almost all mercury currently being

*Indicates suspension, cancelation, or restriction by U.S. EPA Office of Pesticides and Toxic Substances.

discharged is mercury (II). Thus mercury (II) should be the only important possible cause of acute toxicity, and the Criterion Maximum Concentrations can be based on its acute values.

The best available data on long-term exposure of fish to mercury (II) indicates that concentrations above 0.23 μg/L statistically affected fathead minnows significantly, causing the concentration of total mercury in the whole body to exceed 1.0 mg/kg. Although it is not known what percent of the mercury in the fish was methylmercury, it is also not known whether uptake from food would increase the concentration in natural situations. Species such as rainbow trout (*Oncorhynchus mykiss*), coho salmon (*Oncorhynchus kisutch*), and especially the bluegill (*Lepomis macrochirus*) might suffer chronic effects and accumulate high residues of mercury as did the fathead minnow.

With regard to long-term exposure to methylmercury, scientists found that brook trout can exceed the FDA action level without suffering statistically significant adverse effects on survival, growth, or reproduction. Thus for methylmercury, the Final Residue Value would be substantially lower than the Final Chronic Value.

Basing a freshwater criterion on the Final Residue Value of 0.012 μg/L derived from the bioconcentration factor of 81,700 for methylmercury with the fathead minnow essentially assumes that all discharged mercury is methylmercury. On the other hand, in field situations uptake from food might add to the uptake from water. Similar considerations apply to the derivation of the saltwater criterion of 0.025 μg/L using the BCF of 40,000 obtained for methylmercury with the eastern oyster. Because the Final Residue Values for methylmercury are substantially below the Final Chronic Values for mercury (II), of lesser importance is that many fishes, including the rainbow trout, coho salmon, bluegill, and haddock (*Melanogrammus aeglefinus*), might not be adequately protected by the freshwater and saltwater Final Chronic Values for mercury (II).

In contrast to the complexities of deriving numerical criteria for mercury, monitoring for unacceptable environmental effects should be relatively straightforward. The most sensitive adverse effect will probably be to exceed the FDA action level. Therefore, existing discharges should be acceptable if the concentration of methylmercury in the edible portion of exposed consumed species does not exceed the FDA action level.

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 of mercury does not exceed 0.012 μg/L more than once every three years on the average and if the one-hour average concentration does not exceed 2.4 μg/L more than once every three years on the average. If the four-day average concentration exceeds 0.012 μg/L more than once in a three-year period, the edible portion of consumed species should be analyzed to determine whether the concentration of methylmercury exceeds the FDA action level.

The procedures described in the guidelines indicate that, except possibly where a localy important species is very sensitive, saltwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of mercury does not exceed 0.025 μg/L more than once every three years on the average and if the one-hour average concentration does not exceed 2.1 μg/L more than once every three years on the average. If the four-day average concentration exceeds 0.025 μg/L

more than once in a three-year period, the edible protion of consumed species should be analyzed to determine whether the concentration of mathylmercury exceeds the FDA action level.

The recommended exceedence frequency of three years is the Agency's best scientific judgment of the average time needed for an unstressed system to recover from a pollution event in which exposure to mercury exceeds the criterion. A stressed system, 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 can 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 design flow and 7Q5 or 7Q10 for the Criterion Continuous Concentration design flow in steady-state models for unstressed and stressed systems, respectively. These matters are discussed in more detail in EPA's "Technical Support Document for Water Quality-Based Toxics Control."

Human Health Criteria

The ambient water criterion is 144 ng/L for the protection of human health from the toxic properties of mercury ingested through water and contaminated aquatic organisms.

For the protection of human health from the toxic properties of mercury ingested through contaminated aquatic organisms alone, the ambient water criterion is 146 ng/L. These values include the consumption of freshwater, estuarine, and saltwater species.

(45 F.R. 79318 November 28,1980) (50 F.R. 30784, July 29, 1985)

See Appendix A for Aquatic Life Methodology.

See Appendix C for Human Health Mehtodology.

METHOXYCHLOR

72-435

CRITERIA

100 μg/L for domestic water supply (health).
0.03 μg/L for freshwater and saltwater aquatic life.

Rationale

Where adequate human data are available for corraboration of the animal results, the total "safe" drinking intake level is assumed to be 1/100 of the "no effect" or "minimal effect" level reported for the most sensitive animal tested — in this case, humans.

Applying the available data and assuming that 20 percent of the toal intake of methoxychlor is from drinking water and that the average person weighs 70 kg and consumes two liters of water per day, the formual for calculating a criterion is 2.0 mg/kg x 0.2 x 70 kg x 1/100 x 1/2 = 0.14 μg/L. A criterion level for domestic water supply of 100 μg/L is recommended.

In tests with aquatic organisms exposed to methoxychlor, reduced hatchability of fathead minnow (*Pimephales promelas*) embryos at 0.125 μg/L and lack of spawning at 2.0 μg/L was observed. Yellow perch (*Perca flavescens*) exposed to 0.6 μg/L methoxychlor for 8 months exhibited reduced growth. The four-day LC50s for the fathead minnow, the yellow perch, and economically important striped bass (*Morone saxatilis*) were 7.5, 22, and 3.3 μg/L, respectively.

The four-day LC50s for aquatic invertebrates were as low as 0.61 μg/L for the scud (*Gammarus pseudolimnaeus*) and ranged from 1.6 to 7 μg/L for three insect larvae and a crayfish. In 28-day tests, reduction in emergence or pupation of aquatic insects was observed at 0.25 to 0.5 μg/L of methoxychlor.

The data indicate that 0.1 μg/L methoxychlor would be just below the chronic effect level for the fathead minnow and 1/5 the acute toxicity level in crayfish. Thus, a criterion level of 0.03 μg/L is recommended. The 96-hour LC50 for the marine pink shrimp (*Penaeus duorarum*) exposed to methoxychlor is 3.5 μg/L. The 30-day LC50 was 1.3 μg/L. Applying an application factor of 0.01 with the pink shrimp acute toxicity of 3.5 μg/L, the recommended criterion for a saltwater environment is also 0.03 μg/L.

(Quality Criteria for Water, July 1976) PB-263943
See Appendix D for Methodology.

MIREX

2385-85-5

CRITERIA

0.001 µg/L for freshwater and saltwater aquatic life.

Rationale

Mirex is generally limited in its use to control the imported fire ant in the southeastern United States, and it is always presented in bait.

In experiments with field-collect crayfish, juvenile *Procumbarus blandingi* were exposed to 1 or 5 µg/L mirex for six to 144 hours and then transferred to clean water and observed for 10 days. After five days, 95 percent of the crayfish exposed to 1 µg/L mirex for 144 hours were dead. Exposure to 5 µg/L for 6, 24, and 58 hours resulted in 26, 50, and 98 percent mortality within the 10-day observation period in clean water. Several similar experiments with other crayfish species revealed comparable mortality levels in exposures to low levels of mirex. For *Promcambarus hayi*, mirex tissue residue accumulations ranged from 940- to 27,210-fold above water concentrations after 48-hour exposures to 0.1 and 0.5µg/L.

In feeding experiments with 108 crayfish, one particle of mirex bait resulted in a 77 percent mortality rate after six days. Comparable experiments yielded similar results, indicating that mirex is extremely toxic to the tested species of crayfish. Mortality and accumulation increased with exposure time. Field studies have shown that mirex is accumulated through the food chain, while additional data reveals that mirex is transported from treated land into marsh and estuarine areas. Mirex residues were found to increase with trophic levels of animals sampled. In addition, further studies documented mirex in areas not treated with the insecticide. Mirex has been reported in fish from Lake Ontario, Canada, although mirex is not registered for use in Canada.

A summary of data available shows a mosaic of effects. Crayfish and channel catfish survival is affected by mirex in the water or by ingestion of the bait particles. Bioaccumulation is well established for a wide variety of organisms. Mirex is very persistent in bird tissue. Considering the extreme toxicity and potential for bioaccumulation, every effort should be made to keep mirex bait particles out of water containing aquatic organisms, and water concentrations should not exeed 0.001 µg/L mirex. This value is based upon an application factor of 0.01 applied to the lowest levels at which effects on crayfish have been observed.

Data on which to base a marine criterion involve several estuarine and marine crustaceans. A concentration of 0.1 µg/L mirex was lethal to juvenile pink shrimp *(Penaeus duorarum)* in a three-week exposure. Reduced survival of the mud crab *(Rhithropanopeus harrisii)* was observed in 0.1 µg/L mirex. In three of four 28-day seasonal flow-through experiments, reduced survival of *Callinectes sapidus, Penaeus duorarum,* and grass shrimp *(Palaemonetes pugio)*, at levels of 0.12 µg/L in summer, 0.06 µg/L in fall, and 0.09 µg/L in winter, was observed.

Effects of mirex on estuarine and marine crustaceans were observed after considerable time had elapsed, so that length of exposure is an

important consideration for this chemical. This may not be the case in fresh water since the crayfish were affected within 48 hours. Therefore, a three- to four-week exposure might be considered acute; and by applying an application factor of 0.01 to reasonable average of toxic effect levels as summarized above, a recommended marine criterion of 0.001 μg/L results.

(Quality Criteria for Water, July 1976) PB-263943
See Appendix D for Methodology.

NAPHTHALENE

91-20-3

CRITERIA

Aquatic Life

The available data for naphthalene indicate that acute and chronic toxicity to freshwater aquatic life occurs at concentrations as low as 2,300 and 620 μg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested.

The available data also indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 2,350 μg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of naphthalene to sensitive saltwater aquatic life.

Human Health

Using the present guidelines, a satisfactory criterion cannot be derived at this time because of insufficient available data.

(45 F.R. 79318, November 28, 1980)

See Appendix B for Aquatic Life Methodology.

See Appendix C for Human Health Methodology.

NICKEL

7440-02-0

CRITERIA

Not to exceed 75 μg/L in salt water.
8.3 μg/L for saltwater aquatic life.
Freshwater criteria are hardness dependent. See text.

Summary

Acute values with 21 freshwater species in 18 genera range from 1,101 μg/L for a cladoceran to 43,240 μg/L for a fish. Fishes and invertebrates are both spread throughout the range of sensitivity. Acute values with four species are significantly correlated with hardness. Data are available concerning the chronic toxicity of nickel to two invertebrates and two fishes in fresh water. Data available for two species indicate that chronic toxicity decreases as hardness increases. The measured chronic values ranged from 14.77 μg/L for *Daphnia magna* in soft water to 526.7 μg/L for the fathead minnow (*Pimephales promelas*) in hard water. Five acute-chronic ratios are available for two species in soft and hard water and range from 14 to 122 μg/L.

Nickel appears to be quite toxic to freshwater algae, with concentrations as low as 50 μg/L producing significant inhibition. Bioconcentration factors for nickel range from 0.8 for fish muscle to 193 for a cladoceran.

Acute values for 23 saltwater species in 20 genera range from 151.7 μg/L for mysid juveniles to 1,100,000 μg/L for juveniles and adults of a clam. The acute values for the four species of fish range from 7,598 to 350,000 μg/L. Nickel's acute toxicity appears to be related to salinity and is species-dependent.

An acceptable chronic test on nickel has been conducted on only one saltwater species, *Mysidopsis bahia*. In it, chronic exposure to 141 μg/L and greater resulted in reduced survival and reproduction; the measured acute-chronic ratio was 5.478.

Bioconcentration factors in saltwater range from 261.8 for an oyster to 675 for a brown alga.

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 of nickel (in μg/L) does not exceed the numerical value given by

$$e^{(0.8460[\ln (\text{hardness})]+1.1645)}$$

more than once every three years on the average and if the one-hour average concentration (in μg/L) does not exceed the numerical value given by

$$e^{(0.8460[\ln (\text{hardness})]+3.3612)}$$

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 nickel are 88, 160, and 280 μg/L, respectively, and the one-hour average concentrations are 790, 1400, and 2500 μg/L.

The procedures described in the guidelines 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 nickel does not exceed 8.3 μg/L more than once every three years on the average and if the one-hour average concentration does not exceed 75 μg/L more than once every three years on the average.

Three years is the Agency's best scientific judgment of the average amount of time aquatic ecosystems should be provided between excursions. The resiliences of ecosystems and their abilities to recover differ greatly, however, and site-specific, allowed excursion frequencies can be established if adequate justification is provided.

When developing water quality-based permit limits and designing waste treatment facilities, criteria must be applied to an appropriate wasteload allocation model. Dynamic models are preferred, but limited data or other considerations might make their use impractical; therefore, regulatory programs must rely on a steady-state model.

Human Health

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60911). Recalculated IRIS values for nickel are 610 μg/L for ingestion of contaminated water and organisms and 4,600 μg/L for ingestion of contaminated aquatic organisms only.

(45 F.R. 79337, November 28, 1980) (51 F.R. 43665, December 3, 1986)
(57 F.R. 60911, December 22, 1992)

See Appendix A for Aquatic Life Methodology.
See Appendix C for Human Health Methodology.

NITRATES/NITRITES

14797-55-8

CRITERIA

10 mg/L nitrate nitrogen (N) for domestic water supply (health).

Introduction

Two gases — molecular nitrogen and nitrous oxide — and five forms of nongaseous, combined nitrogen — amino and amide groups, ammonium, nitrite, and nitrate — are important in the nitrogen cycle. The amino and amide groups are found in soil organic matter and as constituents of plant and animal protein. The ammonium ion either is released from proteinaceous organic matter and urea or is synthesized in industrial processes involving atmospheric nitrogen fixation. The nitrite ion is formed from the nitrate or the ammonium ions by certain microorganisms found in soil, water, sewage, and the digestive tract. The nitrate ion is formed by the complete oxidation of ammonium ions by soil or water microorganisms. This process, known as denitrification, takes place when nitrate-containing soils become anaerobic and the conversion to nitrite, molecular nitrogen, or nitrous oxide occurs; in some instances, ammonium ions are produced. In oxygenated natural water systems, nitrite is rapidly oxidized to nitrate. Growing plants assimilate nitrate or ammonium ions and convert them to protein.

Among the major point sources of nitrogen entry into waterbodies are municipal and industrial wastewaters, septic tanks, and feed lot discharges. Diffuse sources of nitrogen include farm-site fertilizer and animal wastes, lawn fertilizer, leachate from waste disposal in dumps or sanitary landfills, atmospheric fallout, nitric oxide and nitrite discharges from automobile exhausts and other combustion processes, and losses from natural sources such as mineralization of soil organic matter. Water reuse systems in some fish hatcheries employ a nitrification process for ammonia reduction; this can result in exposure of hatchery fish to elevated levels of nitrite.

Rationale

In quantities normally found in food or feed, nitrates become toxic only under conditions in which they are, or may be, reduced to nitrites. Otherwise, at "reasonable" concentrations, nitrates are rapidly excreted in the urine. High intake of nitrates constitutes a hazard primarily to warmblooded animals under conditions favorable to their reduction to nitrite. Under certain circumstances, nitrate can be reduced to nitrite in the gastrointestinal tract, which then reaches the bloodstream and reacts directly with hemoglobin to produce methemoglobin, with consequent impairment of oxygen transport.

The reaction of nitrite with hemoglobin can be hazardous in infants under 3 months of age. Serious and occasionally fatal poisonings in infants have occurred following ingestion of untreated well waters shown to contain nitrate at concentrations greater than 10 mg/L nitrate/nitrogen (N). High nitrate concentrations frequently are found in shallow farm and rural community wells, often as the result of inadequate protection from barnyard drainage or from septic tanks. Increased concentrations of nitrates

also have been found in streams from farm tile drainage in areas of intense fertilization and farm crop production.

The differences in susceptibility to methemoglobina are not yet understood. They appear, however, to be related to a combination of factors including nitrate concentration, enteric bacteria, and the lower acidity characteristic of the digestive system of baby mammals. Methemoglobinemia symptoms and other toxic effects were observed when high nitrate well waters containing pathogenic bacteria were fed to laboratory mammals. Conventional water treatment has no significant effect on nitrate removal from water.

Because of the potential risk of methemoglobinemia to bottle-fed infants, and in view of the absence of substantiated physiological effects at nitrate concentrations below 10 mg/L nitrate/nitrogen, this level is the criterion for domestic water supplies. Waters with nitrite/nitrogen concentrations over 1 mg/L should not be used for infant feeding. Waters with a significant nitrite concentration usually would be heavily polluted and probably bacteriologically unacceptable.

Quality Criteria for Water, July 1976, provides data for exposed fishes. This data concludes that

1. Levels of nitrate/nitrogen at or below 90 mg/L would have no adverse effects on warmwater fish.

2. Nitrite/nitrogen at or below 5 mg/L should be protective of most warmwater fish.

3. Nitrite/nitrogen at or below 0.06 mg/L should be protective of salmonid fishes.

These levels are not known to occur or would be unlikely to occur in natural surface waters. Recognizing that concentrations of nitrate or nitrite that would exhibit toxic effects on warmwater fish could rarely occur in nature, restrictive criteria are not recommended.

(Quality Criteria for Water, July 1976) PB-263943
See Appendix D for Methodology.

NITROBENZENE

98-95-3

CRITERIA

Aquatic Life

The available data for nitrobenzene indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 27,000 μg/L and would occur at lower concentrations among species that are more sensitive than those tested. No definitive data are available concerning the chronic toxicity of nitrobenzene to sensitive freshwater aquatic life.

The available data for nitrobenzene indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 6,680 μg/L and would occur at lower concentrations among species that are more sensitive than those tested. No definitive data are available concerning the chronic toxicity of nitrobenzene to sensitive saltwater aquatic life.

Human Health

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60848). Recalculated IRIS values for nitrobenzene are 17.0 μg/L for ingestion of contaminated water and organisms and 1,900 μg/L for ingestion of contaminated aquatic organisms only.

Using available organoleptic data, the estimated level is 30 μg/L to control undesirable taste and odor qualities of ambient water. Organoleptic data do have limitations as a basis for establishing a water quality criterion, however, but no demonstrated relationship to potentially adverse effects on human health.

The U.S. EPA is currently developing Acceptable Daily Intake (ADI) or Verified Reference Dose (RfD) values for agencywide use for this chemical. The new value should be substituted when it becomes available. The January 1986 draft Verified Reference Dose document cites an RfD of .0005 mg/kg/day for nitrobenzene.

(45 F.R. 79318, November 28, 1980)

See Appendix C for Human Health Methodology.

NITROPHENOLS

CRITERIA

Aquatic Life

The available data for nitrophenols indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 230 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of nitrophenols to sensitive freshwater aquatic life but toxicity to one species of algae occurs at concentrations as low as 150 µg/L.

The available data for nitrophenols indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 4,850 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of nitrophenols to sensitive saltwater aquatic life.

Human Health

2,4-Dinitro-O-Cresol (2-Methyl-4,6-Dinitrophenol) 534-52-1

Values to protect human health from exposure to 2,4-dinitro-o-cresol are 13.4 µg/L through ingestion of contaminated water and organisms and 765 µg/L through ingestion of contaminated organisms only.

2,4-Dinitrophenol 51-28-5

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60848). Recalculated IRIS values for 2,4-dinitrophenol are 70 µg/L for ingestion of contaminated water and organisms and 14,000 µg/L for ingestion of contaminated aquatic organisms only.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60848, December 22, 1992)
See Appendix C for Human Health Methodology.

NITROSAMINES

35576-91-1

CRITERIA

Aquatic Life

The available data for nitrosamines indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 5,850 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of nitrosamines to sensitive freshwater aquatic life.

The available data for nitrosamines indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 3,300,000 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of nitrosamines to sensitive saltwater aquatic life.

Human Health

N-nitrosodiethylamine 55-18-5

For the maximum protection of human health from the potential carcinogenic effects of exposure to N-nitrosodiethylamine and all other nitrosamines, except those listed below, through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentrations should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels that can result in incremental increase of cancer risk over a lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}. The corresponding recommended criteria are 8.0 ng/L, 0.8 ng/L, and 0.08 ng/L, respectively. If these estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 12,400 ng/L, 1,240 ng/L, and 124 ng/L, respectively.

N-nitrosodimethylamine 62-75-9

For the maximum protection of human health from the potential carcinogenic effects of exposure to N-nitrosodimethylamine through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentrations should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time. Published human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60914). Therefore, the levels that may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}. The corresponding recommended criteria are 0.0069 µg/L, 0.00069 µg/L, and 0.000069 µg/L, respectively. If these estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 81 µg/L, 8.1 µg/L, and 0.81 µg/L, respectively.

N-nitrosodibutylamine 924-16-3

For the maximum protection of human health from the potential carcinogenic effects of exposure to N-nitrosodibutylamine through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentrations should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels that may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}. The corresponding recommended criteria are 64 ng/L, 6.4 ng/L, and 0.64 ng/L, respectively. If these estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 5,868 ng/L, 587 ng/L, and 58.7 ng/L, respectively.

N-nitrosopyrrolidine 930-55-2

For the maximum protection of human health from the potential carcinogenic effects of exposure to N-nitrosopyrrolidine through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentrations should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels that may result in incremental increase of cancer risk over a lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}, The corresponding recommended criteria are 160 ng/L, 16 ng/L, and 1.6 ng/L, respectively. If these estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 919,000 ng/L, 91,900 ng/L, and 9,190 ng/L, respectively.

N-nitrosodiphenylamine 86-30-6

For the maximum protection of human health from the potential carcinogenic effects of exposure to N-nitrosodiphenylamine through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentrations should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels that may result in incremental increase of cancer risk over the lifetime are estimated at $10^{-5,}$ 10^{-6}, and 10^{-7}. Published human health criteria were recalculated using IRIS to reflect available data as of 12/92 (57 F.R. 60914). Recalculated IRIS values for N-nitrosodiphenylamine are 5.0 µg/L for ingestion of contaminated water and organisms and 16.0 µg/L for ingestion of contaminated aquatic organisms only.

N-nitrosodi-n-propylamine 62-164-7

For the maximum protection of human health from the potential carcinogenic effects of exposure to N-nitrosodi-n-propylamine through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentrations should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels that may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}. Human health criteria were calculated using IRIS to reflect available data as of 12/92 (57 F.R. 60890). Calculated values are based on 10^{-6} risk level for N-nitrosodi-n-propylamine are 0.005 µg/L for ingestion of contaminated water and organisms and 1.4 µg/L for ingestion of contaminated aquatic organism only.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60890, December 22, 1992.

See Appendix C for Human Health Methodology.

OIL AND GREASE

CRITERIA

Domestic Water Supply

Virtually free from oil and grease, particularly from the tastes and odors that emanate from petroleum products.

Aquatic Life

1. 0.01 of the lowest continuous flow 96-hour LC50 to several important freshwater and marine species, each having a demonstrated high susceptibility to oils and petrochemicals.

2. Levels of oils or petrochemicals in the sediment that cause deleterious effects to the biota should not be allowed.

3. Surface waters shall be virtually free from floating nonpetroleum oils of vegetable or animal origin as well as petroleum-derived oils.

INTRODUCTION

An estimated 5 to 10 million metric tons of oil enter the marine environment annually. A major difficulty encountered in setting criteria for oil and grease is categorization. Chemicals are not divided into categories but include thousands of organic compounds with varying physical, chemical, and toxicological properties. They can be either volatile or nonvolatile, soluble or insoluble, and persistent or easily degraded.

Rationale

Field and laboratory evidence have demonstrated both acute lethal toxicity and long-term sublethal toxicity of oils to aquatic organisms. Events such as the *Tampico Maru* wreck of 1957 in Baja, California, and the No. 2 fuel oil spill in West Falmouth, Massachusetts, in 1969, both of which caused immediate death to a wide variety of organisms, illustrate the lethal toxicity that may be attributed to oil pollution. Similarly, a gasoline spill in South Dakota in November 1969 was reported to have caused immediate death to the majority of freshwater invertebrates and 2,500 fish, 30 percent of which were native species of trout. Because of the wide range of compounds included in the category of oil, establishing meaningful 96-hour LC50 values for oil and grease without specifying the product involved is impossible. The most susceptible category of organisms, the marine larvae, appear to be intolerant of petroleum pollutants, particularly the water soluble compounds, at concentrations as low as 0.1 mg/L.

The long-term sublethal effects of oil pollution refer to interferences with cellular and physiological processes such as feeding and reproduction and do not lead to immediate death of the organism. Disruption of such behavior apparently can result from petroleum product concentrations as low as 10 to 100 μg/L.

Summaries of some of the sublethal toxicities for various petroleum pollutants and aquatic species are contained in the 1976 criteria. In addition to sublethal effects reported at the 10 to 100 μg/L level, petroleum products can harm aquatic life at concentrations as low as 1 μg/L.

Bioaccumulation of petroleum products presents two especially important public health problems: (1) the tainting of edible, aquatic species, and (2) the possibility of edible marine organisms incorporating the high boiling, carcinogenic polycyclic aromatics in their tissues. Research shows that 0.01mg/L of crude oil caused tainting in oysters. Concentrations as low as 1 to 10 μg/L could lead to tainting within very short periods of time. Chemicals responsible for cancer in animals and humans (such as 3,4-benzopyrene) occur in crude oil. Also, marine organisms are capable of incorporating potentially carcinogenic compounds into their body fat where the compounds remain unchanged.

Oil pollutants may also be incorporated into sediments. Evidence shows that once this occurs in the sediments below the aerobic surface layer, petroleum oil can remain unchanged and toxic for long periods, since its rate of baterial degradation is slow. For example, No. 2 fuel oil incorporated into the sediments after the West Falmouth spill persisted for over a year, and even began spreading in the form of oil-laden sediments to more distant areas that had remained unpolluted immediately after the spill. The persistence of unweathered oil within the sediment could have a long-term effect on the structure of the benthic community or cause the demise of specific sensitive important species.

Reports show that 0.01 mg/L oil produced deformed and inactive flatfish larvae and inhibition or delay of cellular division in algae by oil concentrations of 10^{-4} to 10^{-1} mg/L. A reduction in the chemotactic perception of food by the snail, *Nassarius obsoletus*, at kerosene concentrations of 0.001 to 0.004 mg/L was also reported. Decreased survival and fecundity in worms were reported at concentrations of 0.01 to 10 mg/L of detergent.

Because of the great variability in the toxic properties of oil, establishing a numerical criterion applicable to all types of oil is difficult. Thus, an application factor of 0.01 of the 96-hour LC_{50} as determined by using continuous flow with a sensitive resident species should be employed for individual petrochemical components.

Toxicological data is sparse on the ingestion of the components of refinery wastewaters by humans or by test animals. Any tolerable health concentrations for petroleum-derived substances far exceed the limits of taste and odor. Since petroleum derivatives become organoleptically objectionable at concentrations far below the human chronic toxicity, hazards to humans will not likely arise from drinking oil-polluted waters. Oils of animal or vegetable origin generally are nontoxic to humans and aquatic life.

In view of the problem of petroleum oil incorporated in sediments, its persistence and chronic toxic potential, and the present lack of sufficient toxicity data to support specific criteria, oil concentrations in sediments should not approach levels that cause deleterious effects to important species or the bottom community as a whole.

Petroleum and nonpetroleum oils share some similar physical and chemical properties. Because they share common properties, they may cause similar harmful effects in the aquatic environment by forming a sheen, film, or discoloration on the water surface. Like petroleum oils, nonpetroleum oils may occur at four levels of the aquatic environment: (a) floating on the surface, (b) emulsified in the water column, (c) solubilized, and (d) settled on the bottom as a sludge. Analogous to the grease balls

from vegetable oil and animal fats are the tar balls of petroleum origin that have been found in the marine environment or washed ashore on beaches.

Oils of any kind can cause (a) drowning of water fowl because of loss of buoyancy, exposure because of loss of insulating capacity of feathers, and starvation and vulnerability to predators because of lack of mobility; (b) lethal effects on fish by coating epithelial surface of gills, thus preventing respiration; (c) potential fishkills resulting from biochemical oxygen demand; (d) asphyxiation of benthic life forms when floating masses become engaged with surface debris and settle on the bottom; an (e) adverse aesthetic effects of fouled shorelines and beaches. These and other effects have been documented in the U.S. Department of Health, Education and Welfare report on Oil Spills Affecting the Minnesota and Mississippi Rivers in the 1975 Proceedings of the Joint Conference on Prevention and Control of Oil Spills.

Oils of animal or vegetable origin generally are chemically nontoxic to humans or aquatic life; however, floating sheens of such oils result in deleterious environmental effects described in this criterion. Thus, surface waters should be virtually free from floating nonpetroleum oils of vegetable or animal origin.This same recommendation applies to floating oils of petroleum origin, since they, to, may produce similar effects.

(Quality Criteria for Water, July 1976) PB-263943
See Appendix D for Methodology.

PARATHION

56-38-2

CRITERIA

Aquatic Life

Freshwater — 1-hour average of 0.065 µg/L
4-day average of 0.013 µg/L

Summary

The acute values for 37 freshwater species in 31 genera range from 0.04 µg/L for an early instar of a crayfish (*Orconectes nais*) to 5,230 µg/L for two species of tubificid worms. For *Daphnia magna*, the chronic value and acute- chronic ratio are 0.0990 µg/L and 10.10, respectively. Chronic toxicity values are available for two freshwater fish species, the bluegill (*Lepomis macrochirus*) and the fathead minnow (*Pimephales promelas*), with chronic values of 0.24 µg/L and 6.3 µg/L, and acute-chronic ratios of 2,121 and 79.45, respectively. Two freshwater algae were affected by toxaphene concentrations of 30 and 390 µg/L, respectively. Bioconcentration factors determined with three fish species ranged from 27 to 573.

The acute values available for saltwater species are 11.5 and 17.8 µg/L for the Korean shrimp, Palaemon macrodactylus, and 17.8 µg/L for the striped bass (*Morone saxatilis*). No data are available concerning the chronic toxicity of parathion to saltwater species, toxicity to saltwater plants, or bioaccumulation by saltwater species. Some data indicate that parathion is acutely lethal to commercially important saltwater shrimp at concentrations as low as 0.24 µg/L. Measurement of acetylcholinesterase (AChE) in fish tissue might be useful for diagnosing fish kills caused by parathion.

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 of parathion does not exceed 0.013 µg/L more than once every three years on the average and if the one-hour average concentration does not exceed 0.065 µg/L more than once every three years on the average.

The procedures described in the guidelines require the availability of specified data for the derivation of a criterion. A saltwater criterion for parathion cannot be derived because very few of the required data are available.

In the Agency's best scientific judgment, three years is the average time aquatic ecosystems should be provided between excursions. The resiliences of ecosystems and their abilities to recover differ greatly, however, and site-specific allowed excursion frequencies may be established if adequate justification is provided.

When developing water quality-based permit limits and for designing waste treatment facilities, criteria must be based upon an appropriate

wasteload allocation model. Dynamic models are preferred; but if limited data or other considerations might make their use impractical, rely on steady-state models.

(51 F.R. 43665, December 3, 1986)
See Appendix A for Aquatic Life Methodology.

PENTACHLOROPHENOL (PCP)

87-86-5

CRITERIA

Not to exceed 13.0 μg/L in salt water.
7.9 μg/L for saltwater aquatic life.
Freshwater criteria are pH dependent. See text.

Summary

The acute and chronic toxicity of PCP to freshwater animals increased as pH and dissolved oxygen concentration of the water decreased. Generally, toxicity also increases with increased temperature. The estimated acute sensitivities of 36 species at pH = 6.5 ranged from 4.355 μg/L for larval common carp to 43,920 μg/L for a crayfish. At pH = 6.5, the lowest and highest estimated chronic values of < 1.835 and 79.66 μg/L, respectively, were obtained with different cladoceran species. Chronic toxicity to fish was affected by the presence of impurities, with industrial-grade PCP being more toxic than purified samples. Mean acute-chronic ratios for six freshwater species ranged from 0.8945 to 15.79, but the mean ratios for the four most acutely sensitive species only ranged from 0.8945 to 5.035. Freshwater algae were affected by concentrations as low as 7.5 μg/L, whereas vascular plants were affected at 189 μg/L and above. Bioconcentration factors ranged from 7.3 to 1,066 for three species of fish.

Acute toxicity values from tests with 18 species of saltwater animals, representing 17 genera, range from 22.63 μg/L for late yolk-sac larvae of the Pacific herring (*Clupea harengus pallasi*) to 18,000 μg/L for adult blue mussels (*Mytllus edulis*). The embryo and larval stages of invertebrates and the late larval premetamorphosis stage of fish appear to be the most sensitive life stages to PCP. With few exceptions, fish are more sensitive than invertebrates to PCP. Salinity, temperature, and pH have a slight effect on the toxicity of PCP to some saltwater animals.

Life-cycle toxicity tests have been conducted with the sheepshead minnow (*Cyprinodon variegatus*) and the polychaete worm (*Ophryotrocha diadema*). The chronic value for the minnow is 64.31 μg/L and the acute-chronic ratio is 6.873. Unfortunately, no statistical analysis of the worm test data is available.

The EC50s for saltwater plants range from 17.40 μg/L for the diatom, Skeletonema costatum, to 3,600 μg/L for the green alga (*Dunaliella tertiolecta*). Apparent steady-state BCFs are available for the eastern oyster (*Crassostrea virginica*) and two saltwater fishes and range from 10 to 82.

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 μg/L) of pentachlorophenol does not exceed the numerical value given by

$$e^{[1.005(pH)-5.290]}$$

more than once every three years on the average and if the one-hour average concentration (in μg/L) does not exceed the numerical value given by

$$e^{[1.005(pH)-4.830]}$$

more than once every three years on the average. For example, at pH = 6.5, 7.8, and 9.0, the four-day average concentrations of pentachlorophenol are 3.5, 13, and 43 μg/L, respectively, and the one-hour average concentrations are 5.5, 20, and 68 μg/L. At pH = 6.8, a pentachlorophenol concentration of 1.74 μg/L caused a 50 percent reduction in the growth of yearling sockeye salmon (*Oncorhynchus nerka*) in a 56-day test.

The procedures described in the guidelines 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 pentachlorophenol does not exceed 7.9 μg/L more than once every three years on the average and if the one-hour average concentration does not exceed 13 μg/L more than once every three years on the average.

In the Agency's best scientific judgment, three years is the average time aquatic ecosystems should be provided between excursions. The resiliences of ecosystems and their abilities to recover differ greatly, however, and site-specific allowed excursion frequencies may be established if adequate justification is provided.

When developing water quality-based permit limits and designing waste treatment facilities, criteria must be based upon an appropriate wasteload allocation model. Dynamic models are preferred, but if limited data or other considerations might make their use impractical, rely on steady-state models.

(51 F.R. 43665, December 3, 1986)
See Appendix A for Aquatic Life Methodology.

Human Health

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect data available as of 12/92 (57 F.R. 60840). Recalculated IRIS values for pentachlorophenol are 0.28 μg/L for ingestion of contaminated water and organisms and 8.2 μg/L for ingestion of contaminated aquatic organisms only.

Using available organoleptic data, the estimated level is 30 μg/L to control undesirable taste and odor qualities of ambient water. Organoleptic data do have limitations as a basis for establishing a water quality criterion but no demonstrated relationship to potentially adverse effects on human health.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60848, December 22, 1992)
See Appendix C for Human Health Methodology.

pH

CRITERIA

pH Range

- 5-9 for domestic water supplies.
- 6.5-9.0 for freshwater aquatic life.
- 6.5-8.5 for marine aquatic life (but not more than 0.2 units outside of normally occurring range).

Introduction

"pH" measures the hydrogen ion activity in a water sample. It is mathematically related to hydrogen ion activity according to the expression: $pH = -Log_{10} (H^+)$, where (H^+) is the hydrogen ion activity.

The pH of natural waters is a measure of acid base equilibrium achieved by the various dissolved compounds, salts, and gases. The principal system regulating pH in natural waters is the carbonate system, which is composed of carbon dioxide (CO_2), carbonic acid (H_2CO_3), bicarbonate ion (HCO_3), and carbonate ions (CO_3). The interactions and kinetics of this system have been described by scientists.

pH is an important factor in the chemical and biological systems of natural waters. The degree of dissociation of weak acids or bases is affected by changes in pH, which is important because the toxicity of many compounds is affected by the degree of dissociation. One such example is hydrogen cyanide (HCN): cyanide toxicity to fish increases as the pH is lowered because the chemical equlibrium is shifted toward an increased concentration of HCN. Similar results have been shown for hydrogen sulfide (H_2S).

pH also affects the solubility of metal compounds contained in bottom sediments or as suspended material. For example, laboratory equilibrium studies under anaerobic conditions indicated that pH was an important parameter involved in releasing manganese from bottom sediments.

The pH of a waterbody does not indicate ability to neutralize additions of acids or bases without appreciable change. This characteristic, termed "buffering capacity," is controlled by the amounts of alkalinity and acidity present.

(Quality Criteria for Water, July 1976) PB-263943
See Appendix D for Methodology.

PHENOL

108-95-2

CRITERIA

Aquatic Life

The available data for phenol indicate that acute and chronic toxicity to freshwater aquatic life occurs at concentrations as low as 10,200 μg/L and 2,560 μg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested.

The available data for phenol indicate that toxicity to saltwater aquatic life occurs at concentrations as low as 5,800 μg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of phenol to sensitive saltwater aquatic life.

Human Health

Published human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect data available as of 12/92 (57 F.R. 60912). Recalculated IRIS values for phenol are 21,000 μg/L for ingestion of contaminated water and organisms and 4,600,000 μg/L for ingestion of contaminated aquatic organisms only.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60912, December 22, 1992)
See Appendix C for Human Health Methodology.

PHOSPHORUS

7723-14-0

CRITERION

0.10 μg/L yellow (elemental) phosphorus for estuarine and saltwater aquatic life.

Introduction

Phosphorus in its elemental form is particularly toxic and subject to bioaccumulation in much the same way as mercury. Phosphorus as phosphate, however, is one of the major nutrients required for plant nutrition and essential for life. In excess of a critical concentration, phosphates stimulate plant growths.

During the past 30 years, the belief has developed that increasing standing crops of aquatic plants, which often interfere with water uses and are nuisances to humans, frequently are caused by increasing phosphorus supplies. Such phenomena are associated with a condition of accelerated eutrophication or aging of waters. Phosphorus is not the sole cause of eutrophication, but evidence suggests that frequently it is the key element of all elements required by freshwater plants. Generally, it is present in the least amount relative to need. Therefore, an increase in phosphorus allows use of other already present nutrients for plant growth. Further, of all elements required for plant growth in the water environment, phosphorus is most easily controlled by humans.

Large deposits of phosphate rock are found near the western shore of central Florida, as well as in a number of other States. Deposits in Florida are found in the form of pebbles embedded in a matrix of clay and sand that vary in size from fine sand to about the size of a human foot. The phosphate rock beds lie within a few feet of the surface and are mined by using hydraulic water jets and a washing operation that separates the phosphates from waste materials, a process similar to strip-mining. Florida, Idaho, Montana, North Carolina, South Carolina, Tennessee, Utah, Virginia, and Wyoming all mine phosphate.

Phosphates enter waterways from several different sources. The human body excretes about one pound per year of phosphorus, expressed as "P." The use of phosphate detergents and other domestic phosphates increases the per capita contribution to about 3.5 pounds per year of phosphorus as P. Some industries, such as potato processing, have wastewaters high in phosphates. Cropland, forestland, and idle and urban land contribute varying amounts of phosphorus-diffused sources in drainage to watercourses: surface runoff of rainfall, effluent from tile lines, or return flow from irrigation. Other contributing sources are cattle feedlots, concentrations of domestic or wild ducks, tree leaves, and fallout from the atmosphere.

Evidence indicates that high phosphorus concentrations are associated with accelerated eutrophication of waters when other growth-promoting factors are present; aquatic plant problems develop in reservoirs and other standing waters with phosphorus values lower than those critical in flowing streams; reservoirs and lakes collect phosphates from influent streams

and store a portion of them within consolidated sediments, thus serving as a phosphate sink; and phosphorus concentrations critical to noxious plant growth vary. Therefore, nuisance growths may result from a particular concentration of phosphate in one geographical area but not in another.

The amount or percentage of inflowing nutrients that may be retained by a lake or reservoir is variable and will depend upon the following:

- The nutrient loading to the lake or resevoir;
- The volume of the euphotic zone;
- The extent of biological activities;
- The detention time within a lake basin or the time available for biological activities; and
- The level of discharge from the lake or of the penstock from the reservoir.

Once nutrients are combined within the aquatic ecosystem, their removal is tedious and expensive. Phosphates are used by algae and higher aquatic plants and excess may be stored within the plant cell. With decomposition of the plant cell, some phosphorus may be released immediately through bacterial action for recycling within the biotic community, while the remainder may be deposited with sediments. Much of the material that combines with the consolidated sediments within the lake bottom is bound permanently and will not be recycled into the system.

Rationale

Elemental Phosphorus

Isom (1960) reported and LC_{50} of 0.105 mg/L at 48 hours and 0.025 mg/L at 160 hours for bluegill sunfish *(Lepomis macrochirus)* exposed to yellow phosphorus in distilled water at 26°C and pH 7. The 125- and 195-hour LC_{50}s of yellow phosphorus to Atlantic cod *(Gadus morhua)* and Atlantic salmon *(Salmo salar)* smolts in continuous-exposure experiments was 1.89 and 0.79 µg/L, respectively. No evidence of an incipient lethal level was observed since the lowest concentration of P_4 tested was 0.79 µg/L. Salmon that were exposed to elemental phosphorus concentration of 40 µg/L or less developed a distinct external red color and showed signs of extensive hemolysis. The predominant features of P_4 poisoning in salmon were external redness, hemolysis, and reduced hematocrits.

Following the opening of an elemental phosphorus production plant in Long Harbour, Placentia Bay, Newfoundland, divers observed dead fish upon the bottom throughout the harbor. Mortalities were confined to a water depth of less than 18 meters. Visual evidence showed selective mortality among benthos. Live mussels were found within 300 meters of the effluent pipe, while all scallops within this area were dead.

Fish will concentrate elemental phosphorus from water containing as little as 1 µg/L. In one set of experiments, a cod swimming in water containing 1 µg/L elemental phosphorus for 18 hours concentrated phosphorus to 50 µg/L in muscle, 150 µg/kg in fatty tissue, and 25,000 µg/kg in the liver. The experimental findings showed that phosphorus is quite stable in the fish tissues.

The criterion of 0.10 µg/L elemental phosphorus for marine or estuarine waters in 1/10 of demonstrated lethal levels to important marine organisms and of levels found to result in significant bioaccumulation.

Although a total phosphorus criterion to control nuisance aquatic growths is not presented, the following rationale to support such a criterion, which currently is evolving, should be considered.

Total phosphate phosphorus concentrations in excess of 100 μg/L P may interfere with coagulation in water treatment plants. When such concentrations exceed 25 μg/L at the time of the spring turnover on a volume-weighted basis in lakes or reservoirs, they may occasionally stimulate excessive (nuisance) growths of algae and other aquatic plants. Algal growths inpart undesirable tastes and odors to water, interfere with water treatment, become aesthetically unpleasant, and alter the chemistry of the water supply. They contribute to the phenomenon of cultural eutrophication.

To prevent the development of biological nuisances and to control accelerated or cultural eutrophication, total phosphates as phosphorus (P) should not exceed 50 μg/L in any stream at the point where it enters any lake or reservoir, nor 25 μg/L within the lake or reservoir. A desired goal for the prevention of plant nuisances in streams or other flowing waters not discharging directly to lakes or impoundments is 100 μg/L total P. Most relatively uncontaminated lake districts are known to have surface waters that contain from 10 to 30 μg/L total phosphorus as P.

The majority of the Nation's eutrophication problems are associated with lakes or reservoirs. Currently, more data support establishing a limiting phosphorus level in those waters than in streams or rivers that do not directly impact such water. Some natural conditions, also, would dictate whether a more or less stringent phosphorus level should be considered. Eutrophication problems may occur in waters where the phosphorus concentration is less than that indicated previously. Obviously, such waters would need more stringent nutrient limits. Likewise, in some waters phosphorus is not now a limiting nutrient and the need for phosphorus limits is substantially diminished.

Establishing a phosphorus criterion for flowing waters requires two basic needs: one is to control the development of plant nuisances within the flowing water and, in turn, to control and prevent animal pests that may become associated with such plants; the other is to protect the downstream receiving waterway, regardless of its proximity in linear distance. A portion of the phosphorus that enters a stream or other flowing waterway eventually will reach a receiving lake or estuary either as a component of the fluid mass, as bed load sediments carried downstream, or as floating organic materials drifting just above the streambed or floating on the water's surface. Superimposed on the loading from the inflowing waterway, a lake or estuary can receive additional phosphorus as fallout from the air shed or as a direct introduction from shoreline areas.

Another method to control the inflow of nutrients, particularly phosphates, into a lake is that of prescribing an annual loading to the receiving water. Vollenweider suggests total phosphorus loadings in grams per square meter of surface area per year as a critical level for eutrophic conditions within the receiving waterway for a particular water volume where the mean depth of the lake in meters is divided by the hydraulic detention time in years. Vollenweider's data suggest a range of loading values that should result in oligotrophic lake water quality (see Table 1).

In some waterways, higher concentrations or loadings of total phosphorus do not produce eutrophy or lower concentrations or loadings of total phosphorus may produce nuisance organisms. Therefore, waters now

Table 1.—Annual loadings.

MEAN DEPTH/HYDRAULIC DETENTION TIME (meters/year)	OLIGOTROPHIC OR PERMISSIBLE LOADING (grams/meter^2/year)	EUTROPHIC OR CRITICAL LOADING (grams/meter^2/year)
0.5	0.07	0.14
1.0	0.10	0.20
2.5	0.16	0.32
5.0	0.22	0.45
7.5	0.27	0.55
10.0	0.32	0.63
25.0	0.50	1.00
50.0	0.71	1.41
75.0	0.87	1.73
100.0	1.00	2.00

Source: Vollenweider (1973).

containing less than the specified amounts of phosphorus should not be degraded by the introduction of additional phosphates.

The following specific exceptions can reduce the threat of phosphorus as a contributor to lake eutrophy:

- Naturally occurring phenomena may limit the development of plant nuisances;
- Technological or cost-effective limitations may help control introduced pollutants;
- Waters may be highly laden with natural silts or colors that reduce the penetration of sunlight needed for plant photosynthesis;
- Some waters' morphometric features — steep banks, great depth, and substantial flows — contribute to a history of no plant problems;
- Waters may be managed primarily for waterfowl or other wildlife;
- In some waters, a nutrient other than phosphorus limits plant growth; the level and nature of such a limiting nutrient would not be expected to increase to an extent that would influence eutrophication; and
- In some waters, phosphorus control cannot be sufficiently effective under present technology to make phosphorus the limiting nutrient.

No national criterion is presented for phosphate phosphorus for the control of eutrophication.

(Quality Criteria for Water, July 1976) PB-263943
See Appendix D for Methodology.

PHTHALATE ESTERS

CRITERIA

Aquatic Life

The available data for phthalate esters indicate that acute and chronic toxicity to freshwater aquatic life occurs at concentrations as low as and μg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested.

The available data for phthalate esters indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as μg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of phthalate esters to sensitive saltwater aquatic life, but toxicity to one species of algae occurs at concentrations as low as μg/L.

Human Health

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60913). Recalculated IRIS values for diethyl phthalate are 23,000 μg/L for ingestion of contaminated water and organisms and 120,000 μg/L for ingestion of contaminated aquatic organisms only.

Human health criteria were recalculated using IRIS to reflect available data as of 12/92 (57 F.R. 60913). Recalculated IRIS values for dibutyl phthalate are 2,700 μg/L for ingestion of contaminated water and organisms and 12,000 μg/L for ingestion of contaminated aquatic organisms only.

Human health criteria were recalculated using IRIS to reflect available data as of 12/92 (57 F.R. 60890). Recalculated IRIS values for di-2-ethylhexyl phthalate are 1.8 μg/L for ingestion of contaminated water and organisms and 5.9 μg/L for ingestion of contaminated aquatic organisms only. IRIS values are based on a 10^{-6} risk level for carcinogens.

Human health criteria were recalculated using IRIS to reflect available data as of 12/92 (57 F.R. 60913). Recalculated IRIS values for dimethyl phthalate are 313,000 μg/L for ingestion of contaminared water organisms and 2,900,000 μg/L for ingestion of contaminated organisms only. IRIS values are based on a 10^{-6} risk level for carcinogens.

Human health criteria were recalculated using IRIS to reflect available data as of 12/92 (57 F.R. 60890). Recalculated IRIS values for butylbenzyl phthalate are 3,000 μg/L for ingestion of contaminated water and organisms and 5,200 μg/L for ingestion of contaminated organisms only. IRIS values are based on a 10^{-6} risk level for carcingens.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60848, December 22, 1992)
See Appendix C for Human Health Methodology.

POLYCHLORINATED BIPHENYLS (PCBs)

1336-36-3

CRITERIA

Not to exceed 2.0 μg/L in fresh water or 10.0 μg/L in salt water.
0.014 and 0.03 mg/L for freshwater and saltwater aquatic life, respectively.

Aquatic Life

For polychlorinated biphenyls, the criterion to protect freshwater aquatic life as derived using the guidelines is 0.014 μg/L as a 24-hour average. This concentration is probably too high because it is based on bioconcentration factors measured in laboratory studies; however, field studies produce factors at least 10 times higher for fishes. The available data indicate that acute toxicity to freshwater aquatic life probably will occur only at concentrations above 2.0 μg/L; therefore, the 24-hour average should provide adequate protection against acute toxicity.

The criterion to protect saltwater aquatic life for polychlorinated biphenyls as derived using the guidelines is 0.030 μg/L as a 24-hour average. This concentration is probably too high because it is based on bioconcentration factors measured in laboratory studies; however, field studies produce factors at least 10 times higher for fishes. The available data indicate that acute toxicity to saltwater aquatic life probably will only occur at concentrations above 10 μg/L; therefore, 24-hour average criterion should provide adequate protection against acute toxicity.

Human Health

For the maximum protection of human health from the potential carcinogenic effects of exposure to polychlorinated biphenyls through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time.

Published human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60915). Recalculated IRIS values for PCBs 1016, 1221, 1232, 1242, 1248, 1254, and 1260 are estimated at 10^{-6} risk level. The recommended criteria for consumption of contaminated water and organisms is 0.000044 μg/L and 0.000045 μg/L for organisms only.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60915, December 22, 1992)
See Appendix B for Aquatic Life Methodology.
See Appendix C for Human Health Methodology.

POLYNUCLEAR AROMATIC HYDROCARBONS

CRITERIA

Aquatic Life

The limited freshwater database, available mostly from short-term bioconcentration studies with two compounds, does not permit a statement concerning acute or chronic toxicity for polynuclear aromatic hydrocarbons.

The data that are available indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 300 μg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning these hydrocarbons' chronic toxicity to sensitive saltwater aquatic life.

Human Health

For the maximum protection of human health from the potential carcinogenic effects of exposure to polynuclear aromatic hydrocarbons through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be presently attainable. Therefore, the levels that may result in incremental increase of cancer risk over a lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}, with the corresponding recommended criteria 28.0 ng/L, 2.8 ng/L, and 0.28 ng/L, respectively. If these estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 311.0 ng/L, 31.1 ng/L, and 3.11 ng/L, respectively.

Human health criteria for three polynuclear aromatic hydrocarbons — acenaphylene, phenanthrene, and benzo (g,h,i) perylene — have been deleted (see 57 F.R. 60887, December 22, 1992). Although the water quality criteria for these compounds have been deleted, information in the 1980 document may be useful.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60848, December 22, 1992)
See Appendix C for Human Health Methodology.

SELENIUM

7782-49-2

CRITERIA

Not to exceed 20 µg/L in fresh water or 300 µg/L in salt water.
5.0 µg/L and 71 µg/L for freshwater and saltwater aquatic life, respectively.

Implementation

Because of the variety of forms of selenium in ambient water and the lack of definitive information about their relative toxicities to aquatic species, no available analytical measurement is known to be ideal for expressing aquatic life criteria for selenium. Previous aquatic life criteria for metals and metalloids were expressed in terms of the total recoverable measurement, but newer criteria for metals and metalloids were expressed in terms of the acid-soluble measurement. Acid-soluble selenium — operationally defined as the selenium that passes through a 0.45 um membrane filter after the sample has been acidified to a pH between 1.5 and 2.0 with nitric acid — is probably the best measurement at the present time.

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 of selenium does not exceed 5.0 µg/L more than once every three years on the average, and if the one-hour average concentration does not exceed 20 µg/L more than once every three years on the average.

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 selenium does not exceed 71 µg/L more than once every three years on the average and if the one-hour average concentration does not exceed 300 µg/L more than once every three years on the average. If selenium is as toxic to saltwater fishes as it is to freshwater fishes in the field, the status of the fish community should be monitored whenever the concentration of selenium exceeds 5 µg/L in salt water.

Human Health

Human health criteria have been withdrawn for this compound (see 57 F.R. 60885, December 22, 1992). Although the human health criteria are withdrawn, EPA published a document for this compound that may contain useful human health information. This document was originally noticed in 45 F.R. 79331, November 28, 1980.

(45 F.R. 79331, November 28, 1980) (53 F.R. 177, January 5, 1988)
(57 F.R. 60911, December 22, 1992)
See Appendix A for Aquatic Life Methodology.

SILVER

7440-22-4

CRITERIA

Not to exceed 2.3 μg/L in salt water.
Freshwater values are hardness dependent. See text.
0.12 μg/L for freshwater aquatic life.

Aquatic Life

For freshwater aquatic life, the concentration (in μg/L) of total recoverable silver should not exceed the numerical value given by

$$e^{(1.72[\ln(\text{hardness})]-6.52)}$$

at any time. For example, at hardnesses of 50, 100, and 200 mg/L as $CaCO_3$, the concentration of total recoverable silver should not exceed 1.2, 4.1, and 13 μg/L, respectively, at any time. The available data indicate that chronic toxicity to freshwater aquatic life may occur at concentrations as low as 0.12 μg/L.

For saltwater aquatic life, the concentration of total recoverable silver should not exceed 2.3 μg/L at any time. No data are available concerning the chronic toxicity of silver to sensitive saltwater aquatic life.

Human Health

Human health criteria have been withdrawn for this compound (see 57 F.R. 60885, December 22, 1992). Although the human health criteria are withdrawn, EPA published a document for this compound that may contain useful human health information. This document was originally noticed in 45 F.R. 79318, November 28, 1980.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60911, December 22, 1992)
See Appendix B for Aquatic Life Methodology.
See Appendix C for Human Health Methodology.

SOLIDS (SUSPENDED, SETTLEABLE) AND TURBIDITY

CRITERIA

Freshwater Fish and Other Aquatic Life

Settleable and suspended solids should not reduce the depth of the compensation point for photosynthetic activity by more than 10 percent from the seasonally established norm for aquatic life.

Introduction

The term "suspended and settleable solids" describes the organic and inorganic particulate matter in water. The equivalent terminology used for solids in "Standard Methods" is "total suspended matter" for suspended solids, "settleable matter" for settleable solids, "volatile suspended matter" for volatile solids, and "fixed suspended matter" for fixed suspended solids. The term "solids" is used in this discussion because of its more common use in the water pollution control literature.

Rationale

Suspended solids and turbidity are important parameters in both municipal and industrial water supply practices. Finished drinking waters have a maximum limit of 1 turbidity unit where the water enters the distribution system. This limit is based on health considerations as they relate to effective chlorine disinfection. Suspended matter provides areas where microorganisms do not come into contact with the chlorine disinfectant. The ability of common water treatment processes (i.e., coagulation, sedimentation, filtration, and chlorination) to remove suspended matter to achieve acceptable final turbidities is a function of the material's composition as well as its concentration. Because of the variability of such removal efficiency, general raw water criterion for these uses cannot be delineated.

Turbid water interferes with recreational use and aesthetic enjoyment of water. It can be dangerous for swimming, especially if diving facilities are provided, because of the possibility of unseen submerged hazards and the difficulty in locating swimmers in danger of drowning. The less turbid the water, the more desirable it becomes for swimming and other water contact sports. Other recreational pursuits, such as boating and fishing, will be adequately protected by suspended solids criteria developed for protection of fish and other aquatic life.

Fish and other aquatic life requirements concerning suspended solids can be divided into those whose effect occurs in the water column and those whose effect occurs following sedimentation to the bottom of the waterbody. Noted effects are similar for both fresh and marine waters.

The effects of suspended solids on fish have been reviewed by the European Inland Fisheries Advisory Commission. This 1965 review identified four effects on fish and fish food populations:

1. By acting directly on the fish swimming in water in which solids are suspended, and either killing them or reducing their growth rate, resistance to disease, etc.;

2. By preventing the successful development of fish eggs and larvae;
3. By modifying natural movement and migrations of fish; and
4. By reducing the abundance of food available to the fish.

Settleable materials that blanket the bottom of waterbodies damage the invertebrate populations, block gravel spawning beds, and, if organic, remove dissolved oxygen from overlying waters. In a study downstream from the discharge of a rock quarry where inert suspended solids were increased to 80 mg/L, the density of macroinvertebrates decreased by 60 percent; in areas of sediment accumulation, benthic invertebrate populations also decreased by 60 percent regardless of the suspended solid concentrations. Similar effects have been reported downstream from an area that was intensively logged. Major increases in stream suspended solids (25 ppm turbidity upstream, versus 390 ppm downstream) caused smothering of bottom invertebrates, reducing organism density to only 7.3 per square foot, versus 25.5 per square foot upstream.

When settleable solids block gravel spawning beds that contain eggs, high mortalities result although evidence suggests that some species of salmonids will not spawn in such areas.

Silt attached to the eggs may prevent sufficient exchange of oxygen and carbon dioxide between the egg and the overlying water. The important variables are particle size, stream velocity, and degree of turbulence.

Deposition of organic materials to the bottom sediments can cause imbalances in stream biota by increasing bottom animal density, principally worm populations; diversity is reduced as pollution-sensitive forms disappear. Algae likewise flourish in such nutrient-rich areas, although forms may become less desirable.

Plankton and inorganic suspended materials reduce light penetration into the waterbody, reducing the depth of the photic zone. This reduces primary production and decreases fish food. In 1974 the National Academy of Sciences recommended that the depth of light penetration not be reduced by more than 10 percent. Additionally, the near surface waters are heated because of the greater heat absorbency of the particulate material, which tends to stabilize the water column and prevents vertical mixing. Such mixing reductions decrease the dispersion of dissolved oxygen and nutrients to lower portions of the waterbody.

One partially offsetting benefit of suspended inorganic material in water is the sorption of organic materials such as pesticides. Following this sorption process, subsequent sedimentation may remove these materials from the water column into the sediments.

Identifiable effects of suspended solids on irrigation use of water include the formation of crusts on top of the soil, which inhibit water infiltration and plant emergence and impeded soil aeration; the formation of films on plant leaves that blocks sunlight and impedes photosynthesis and that may reduce the marketability of some leafy crops like lettuce; and finally, the adverse effect on irrigation reservoir capacity, delivery canals, and other distribution equipment.

The criteria for freshwater fish and other aquatic life are essentially that proposed by the National Academy of Sciences and the Great Lake Water Quality Board.

(Quality Criteria for Water, July 1976) PB-263943

See Appendix D for Methodology.

SULFIDE - HYDROGEN SULFIDE

7783-06-4

CRITERIA

2 μg/L undissociated H_2S for fish and other aquatic life in either fresh water or salt water.

Introduction

Hyrogen sulfide is a soluble, highly poisonous, gaseous compound that smells like rotten eggs. Humans can detect it in air at a dilution of 0.002 ppm. It will dissolve in water at 4,000 mg/L at 20°C and one atmosphere of pressure. Biologically, hydrogen sulfide is an active compound found primarily as an anaerobic degradation product of both organic sulfur compounds and inorganic sulfates. Sulfides are constituents of many industrial wastes such as those from tanneries, paper mills, chemical plants, and gas works. The anaerobic decomposition of sewage, sludge beds, algae, and other naturally deposited organic material is a major source of hydrogen sulfide.

When soluble sulfides are added to water, they react with hydrogen ions to form HS^- or H_2S, the proportion of each depending on the pH. The toxicity of sulfides derives primarily from H_2S rather than from the hydrosulfide (HS^-) or sulfide (S=) ions.

When hydrogen sulfide dissolves in water it dissociates according to the following reactions:

$$H_2S \Leftrightarrow HS^- + H^+ \text{ and } HS^- \Leftrightarrow S^{-2} + H^+$$

At pH 9, about 99 percent of the sulfide is in the form of HS^-, at pH 7 the sulfide is equally divided between HS^- and H_2S; and at pH 5 about 99 percent of the sulfide is present as H_2S. Investigators have minimized the toxic effects of H_2S on fish and other aquatic life because H_2S is oxidized in well-aerated water by natural biological systems to sulfates or is biologically oxidized to elemental sulfur.

Rationale

The degree of hazard exhibited by sulfide to aquatic animal life is dependent on the temperature, pH, and dissolved oxygen. AT lower pH values, a greater proportion is in the form of the toxic undissociated H_2S. In winter when the pH is neutral or below or when dissolved oxygen levels are low but not lethal to fish, the hazard from sulfides is exacerbated. Fish exhibit a strong avoidance reaction to sulfide. Based on data from experiments with the stickleback, if fish encounter a lethal concentration of sulfide, reasonable chance exists that they will be repelled by it before they are harmed. This, of course, assumes that an escape route is open.

Many past data on the toxicity of hydrogen sulfide to fish and other aquatic life have been based on extremely short exposure periods. Consequently, these early data have indicated that concentrations between 0.3 and 0.4 mg/L permit fish to survive. Recent long-term data, both in field situations and under controlled laboratory condition, demonstrate hydrogen sulfide toxicity at lower concentrations.

Concentrations as high as 0.7 mg/L have been found within 20 mm of the bottom of sludge beds, and the levels of 0.1 to 0.02 mg/L were common within the first 20 mm of water above this layer. Walleye *(Stizostedion vitreum)* eggs held in trays in this zone did not hatch. The hatchability of northern pike *(Esox lucius)* eggs was substantially reduced at 25 μg/L at H_2S; at 47 μg/L, mortality was almost complete. Northern pike fry had 96-hour LC_{50} values that varied from 17 to 32 μg/L at normal oxygen levels of 6.0 mg/L. The highest concentration of hydrogen sulfide that had no observable effect on eggs and fry was 14 and 4 μg/L, respectively. Eggs, fry, and juveniles of walleyes and white suckers *(Catostomus commersoni)* and safe levels in working on walleyes and fathead minnows *(Pimephales promelas)* were found to vary from 2.9 μg/L to 12 μg/L, with eggs being the least sensitive and juveniles being the most sensitive in short-term test. In 96-hour bioassays, fathead minnows and goldfish *(Carassius auratus)* varied greatly in tolerance to hydrogen sulfide with changes in temperature. They were more tolerant at low temperatures (6 to 10°C). In addition, 1.0 mg/L sulfide caused 100 percent mortality in 72 hours with Pacific salmon.

On the basis of chronic tests evaluating growth and survival, the safe H_2S level for bluegill *(Lepomis macrochirus)* juveniles and adults was 2 μg/L. Egg deposition in bluegills was reduced after 46 days in 1.4 μg/L H_2S. White sucker eggs were hatched at 15 μg/L, but juveniles showed growth reductions at 1 μg/L. Safe level for fathead minnows were between 2 and 3 μg/L. Studies showed that safe levels for *Gammarus Pseudolimnaeus* and *Hexagenia limbata* were 2 and 15 μg/L, respectively. Some species typical of normally stressed habitats *(Asellus* spp.) were much more resistant.

Sulfide criteria for domestic or livestock use have not been established because the unpleasant odor and taste would preclude such use at hazardous concentrations.

The hazard from hydrogen sulfide to aquatic life is often localized and transient. Available data indicate that water containing concentrations of 2.0 μg/L undissociated H_2S would not be hazardous to most fish and other aquatic wildlife, but concentrations in excess of 2.0 μg/L would constitute a long-term hazard.

(Quality Criteria for Water, July 1976) PB-263943
See Appendix D for Methodology.

TAINTING SUBSTANCES

CRITERIA

Aquatic Life

Materials should not be present in concentrations that individually or in combination produce undesirable flavors that are detectable by organoleptic tests performed on the edible portions of aquatic organisms.

Rationale

Fish or shellfish with abnormal flavors, colors, tastes, or odors are either not marketable or elicit consumer complaints and possible rejection of the food source, even though subsequent lots of organisms may be acceptable. In some areas, poor product quality can and has seriously affected or eliminated the commercial fishing industry. Recreational fishing also can be affected adversely by off-flavor fish. For the majority of sport fishers, consuming the catch is part of their recreation; off-flavored catches can divert the angler to other waterbodies. This can have serious economic impact on established recreation industries such as tackle and bait sales and boat and cottage rental.

A number of wastewaters and chemical compounds have been found to lower the palatability of fish flesh. Implicated wastewaters included those from 2,4-D manufacturing plants, kraft and neutral sulfite pulping processes, municipal wastewater treatment plants, and slaughterhouses, as well as oily, refinery, and phenolic wastes. The list of implicated chemical compounds is long; it includes cresol and phenol compounds, kerosene, naphthol, styrene, toluene, and exhaust outboard motor fuel.

The susceptibility of fishes to the accumulation of tainting substances is variable and depends on the species, length of exposure, and the pollutant. As little as 0.1 μg/L o-chlorophenol can cause tainting of fish flesh. As little as 5 μg/L of gasoline can impart off-flavors to fish.

(Quality Criteria for Water, July 1976) PB-263943
See Appendix D for Methodology.

TEMPERATURE

CRITERIA

Freshwater Aquatic Life

For any time of year, a location has two upper limiting temperatures (based on the important sensitive species present at that time).

1. One limit, a maximum temperature for short exposures, is time dependent and given by the following species-specific equation:

   Temperature (°C) = (1/b) [Log_{10} [time (min)] -a) - 2°C

   where:

   Log_{10} = logarithm to base 10 (common logarithm)

   a = intercept on the "y" or logarithmic axis of the line fitted to experimental data, which is available for some species from Appendix II-C of the National Academy of Sciences (1974)

   b = slope of the line fitted to experimental data, available for some species from Appendix II-C of the National Academy of Sciences (1974)

2. The second value is a limit on the weekly average temperature that

   a. in cooler months — mid-October to mid-April in the North and December to February in the South — will protect against mortality of important species if the elevated plume temperature is suddenly dropped to the ambient temperature, with the limit being the acclimation temperature minus 2°C, when the lower lethal threshold temperature equals the ambient water temperature (in some regions this limitation may also be applicable in summer);

   b. in the warmer months — April through October in the North and March through November in the South — is determined by adding to the physiological optimum temperature (usually for growth) a factor calculated as one-third of the difference between the ultimate upper incipient lethal temperature and the optimum temperature for the most sensitive important species (and appropriate life state) that normally is found at that location and time;

   c. during reproductive seasons — generally April through June and September through October in the North and March through May and October through November in the South — the limit is a temperature that meets site-specific requirements for successful migration, spawning, egg incubation, fry rearing, and other reproductive functions of important species. These local requirements should supersede all other requirements when they are applicable; or

   d. is site-specific and found necessary to preserve normal species diversity or prevent appearance of nuisance organisms.

Marine Aquatic Life

To assure protection of the characteristic indigenous marine community of a waterbody segment from adverse thermal effects

1. The maximum acceptable increase in the weekly average temperature resulting from artificial sources is 10°C (1.8°F) during all seasons of the year, providing the summer maxima are not exceeded; and
2. Daily temperature cycles characteristic of the waterbody segment should not be altered in either amplitude or frequency.

Summer thermal maxima, which define the upper thermal limits for the communities of the discharge area, should be established on a site-specific basis. Existing studies suggest the following regional limits as shown in Table 1.

Table 1.—Regional Limits.

	SHORT-TERM MAXIMUM	MAXIMUM TRUE DAILY MEAN*
Sub-tropical regions (south of Cape Canaveral and Tampa Bay, Fla., and Hawaii)	32.2°C (90°F)	29.4°C (85°F)
Cape Hatteras, N.C., to Cape Canaveral, Fla.	32.2°C (90°F)	29.4°C (85°F)
Long Island (south shore) to Cape Hatteras, N.C.	30.6°C (87°F)	27.8°C (82°F)

*True daily mean = average of 24 hourly temperature readings.

Baseline thermal conditions should be measured at a site without unnatural thermal addition from any source, which is in reasonable proximity to the thermal discharge (within 5 miles), and which has similar hydrography to that of the receiving waters at the discharge.

Introduction

Human uses of water in and out of its natural situs in the environment are affected by its temperature. Offstream domestic uses and in-stream recreation are both partially temperature-dependent. Likewise, species composition and activity of life in any aquatic environment is regulated by water temperature. Since essentially all of these are so-called "cold blooded" or poikilotherm organisms, the temperature of the water regulates their metabolism and ability to survive and reproduce effectively. Industrial uses for process water and cooling are likewise regulated by the water temperature.

Temperature, therefore, is an important physical parameter that, to some extent, regulates many of the beneficial uses of water. In 1967, the Federal Water Pollution Control Administration called temperature "a catalyst, a depressant, an activator, a restrictor, a stimulator, a controller, a killer — one of the most important and most influential water quality characteristics to life in water."

Rationale

The suitability of water for total body immersion is greatly affected by temperature. In temperate climates, dangers from exposure to low temperatures is more prevalent than exposure to elevated water temperatures. Depending on the amount of activity by the swimmer, comfortable temperatures range from 20°C to 30°C. Short durations of lower and higher temperatures can be tolerated by most individuals. For example, for a 30-minute period, temperatures of 10°C or 35°C can be tolerated without harm.

Temperature also affects the self-purification phenomenon in waterbodies, and therefore, the aesthetic and sanitary qualities that exist. Increased temperatures accelerate the biodegradation of organic material both in the overlying water and in bottom deposits, which makes increased demands on the dissolved oxygen resources of a given system. The typical situation is exacerbated by the fact that oxygen becomes less soluble as water temperature increases. Thus, greater demands are exerted on an increasingly scarce resource that may lead to total oxygen depletion and obnoxious septic conditions.

Indicator enteric bacteria, and presumably enteric pathogens, are likewise affected by temperature. Both total and fecal coliform bacteria die away more rapidly in the environment with increasing temperatures. Likewise, changes from a coldwater fishery to a warmwater fishery can occur because temperature may be directly lethal to adults or fry and cause a reduction of activity or limit reproduction.

Upper and lower limits for temperature have been established for many aquatic organisms. Considerably more data exist for upper than lower limits. Tabulations of lethal temperatures for fish and other organisms are available. Factors such as diet, activity, age, general health, osmotic stress, and even weather contribute to the lethality of temperature. The aquatic species, thermal accumulation state, and exposure time are considered the critical factors.

The effects of sublethal temperatures on metabolism, respiration, behavior, distribution and migration, feeding rate, growth, and reproduction have been summarized. Another study has illustrated that the tolerance zone contains a more restrictive temperature range in which normal activity and growth occur; and an even more restrictive zone exists inside that in which normal reproduction will occur.

Data on the combined effects of increased temperature and toxic materials on fish indicate that toxicity generally increases with increased temperature an that organisms subjected to stress from toxic materials are less tolerant of temperature extremes.

An organisms tolerance to temperature extreme is a function of its genetic ability to adapt to thermal changes.

Temperature effects have been shown for water treatment processes. Lower temperatures reduce the effectiveness of coagulation with alum and subsequent rapid sand filtration. In one study, difficulty was especially pronounced below 5°C. Decreased temperature also decreases the effectiveness of chlorination. Based on studies relating chlorine dosage to temperature, and with a 30-minute contact time, dosages required for equivalent disinfective effect increased by as much as a factor of 3 when temperatures were decreased from 20°C to 10°C. Increased temperature may increase the water's odor because of the increased volatility of odor-causing compounds. Odor problems associated with plankton may also be aggravated.

The effects of temperature on aquatic organisms have been the subject of comprehensive literature reviews and annual literature reviews published by the Water Pollution Control Federation. Only highlights from the thermal effects on aquatic life are presented here.

Temperature changes in waterbodies can alter the existing aquatic community. The dominance of various phytoplankton groups in specific temperature ranges has been shown. For example, from 20°C to 25°C, diatoms predominated; green algae predominated from 30°C to 35°C; and blue-greens predominated above 35°C within their characteristic

temperature range, the acclimation temperature prior to exposure, and the time of exposure to the elevated temperature. The upper incipient lethal temperature or the highest temperature than 50 percent of a sample of organisms can survive is determined on the organism at the highest sustainable acclimation temperature. The lowest temperature that 50 percent of the warm acclimated organisms can survive in is the ultimate lower incipient lethal temperature. True acclimation to changing temperatures requires several days. The lower end of the temperature accommodation range for aquatic life is 0°C in fresh water and somewhat less for saline waters. However, organisms acclimated to relatively warm water, when subjected to reduced temperatures that under other conditions of acclimation would not be detrimental, may suffer a significant mortality caused by thermal shock.

Through the natural changes in climatic conditions, the temperatures of waterbodies fluctuate daily, as well as seasonally. These changes do not eliminate indigenous aquatic populations, but affect the existing community structure and the geographic distribution of species. Such temperature changes are necessary to induce the reproductive cycles of aquatic organisms and to regulate other life factors.

Artificially induced changes, such as the return of cooling water or the release of cool hypolimnetic waters from impoundments, may alter indigenous aquatic ecosystems. Entrained organisms may be damaged by temperature increases across cooling water condensers if the increase is sufficiently great or the exposure period sufficiently long. Impingement upon condenser screens, chlorination for slime control, or other physical insults damage aquatic life. However, data has shown that algae passing through condensers are not injured if the temperature of the outflowing water does not exceed 345°C.

In open waters elevated temperatures may affect periphyton, benthic invertebrates, and fish, in addition to causing shifts in algae dominance. Studies of the Delaware River downstream from a power plant concluded that the periphyton population was considerably altered by the discharge.

The number and distribution of bottom organisms decrease as water temperature increase. The upper tolerance limit for a balanced benthic population structure is approximately 32°C. A large number of these invertebrate species are able to tolerate higher temperatures than those required for reproduction.

In order to define criteria for fresh waters, the following was cited as currently defineable requirements:

1. Maximum sustained temperatures that are consistent with maintaining desirable levels of productivity;

2. Maximum levels of metabolic acclimation to warm temperatures that will permit return to ambient winter temperatures should artificial sources of heat cease;

3. Time-dependent temperature limitations for survival of brief exposures to temperature extremes, both upper and lower;

4. Restricted temperature ranges for various states of reproduction, including (for fish) gametogenesis, spawning migration, release of gametes, development of the embryo, commencement of independent feeding (and other activities) by juveniles, and temperatures required for metamorphosis, emergence, or other activities of lower forms;

5. Thermal limits for diverse species compositions of aquatic communities, particularly where reduction in diversity creates nuisance growths of certain organisms, or where important food sources (food chains) are altered;

6. Thermal requirements of downstream aquatic life (in rivers) where upstream diminution of a coldwater resource will adversely affect downstream temperature requirements.

The major portion of such information that is available, however, is for freshwater fish species rather than lower forms of marine aquatic life.

The temperature-time duration for short-time exposures, such that 50 percent of a given population will survive and extreme temperature, frequently is expressed mathematically by fitting experimental data with a straight line on a semi-logarithmic plot with time on the logarithmic scale and temperature on the linear scale. In equation form, this 50 percent mortality relationship is

$$\text{Log}_{10}(\text{time}^{(\text{minutes})}) = a + b\,(\text{Temperature } (^\circ\text{C}))$$

where:

Log_{10} = logarithm to base 10 (common logarithm)

a = intercept on the "y" or logarithmic axis of the line fitted to experimental data and which is available for some species from Appendix II-C of the National Academy of Sciences document

b = slope of the line fitted to experimental data and which is available for some species from Appendix II-C of the National Academy of Sciences document

To provide a safety factor so that none or only a few organisms will perish, experiments found that a criterion of 2°C below maximum temperature is usually sufficient. To provide safety for all the organisms, the temperature causing a median mortality for 50 percent of the population would be calculated and reduced by 2°C in the case of an elevated temperature. Available scientific information includes upper and lower incipient lethal temperatures, coefficients "a" and "b" for the thermal resistance equation, and information of size, life stage, and geographic source of the particular test species.

Maximum temperatures for an extensive exposure (e.g., more than one week) must be divided into those for warmer periods and winter. Other than for reproduction, the most temperature-sensitive life function appears to be growth. A satisfactory estimate of a limiting maximum weekly mean temperature may be an average of the optimum temperature for growth and the temperature for zero net growth.

Because of the difficulty in determining the temperature of zero net growth, essentially the same temperature can be derived by adding to the optimum essentially to temperature (for growth or other physiological functions) a factor calculated as one-third of the difference between the ultimate upper incipient lethal temperature and the optimum temperature. In equation form

$$\begin{array}{l}\text{Maximum weekly}\\ \text{average temperature}\end{array} = \begin{array}{c}\text{optimum}\\ \text{temperature}\end{array} + \begin{array}{l}\tfrac{1}{3}\text{ (ultimate upper incipient lethal}\\ \text{temperature} - \text{optimum}\\ \text{temperature)}\end{array}$$

Since temperature tolerance varies with various states of development of a particular species, the criterion for a particular location would be

calculated for the most important life form likely to be present during a particular month. One caveat in using the maximum weekly mean temperature is that the limit for short-term exposure must not be exceeded. Example calculations for predicting the summer maximum temperatures for short-term survival and for extensive exposure for various fish species are presented in Table 2. These calculations use the above equations and data from EPA's Environmental Research Laboratory in Duluth.

Table 2.— Example calculated values for maximum weekly average temperatures for growth and short-term maxima for survival for juveniles and adults during the summer (centigrade and Fahrenheit).

SPECIES	GROWTH[a]	MAXIMA[b]
Atlantic salmon	20 (68)	23 (73)
Bigmouth buffalo	—	—
Black crappie	27 (81)	—
Bluegill	32 (90)	35 (95)
Brook trout	19 (66)	24 (75)
Carp	—	—
Channel catfish	32 (90)	35 (95)
Coho salmon	18 (64)	24 (75)
Emerald shiner	30 (86)	—
Freshwater drum	—	—
Lake herring (Cisco)	17 (63)[c]	25 (77)
Largemouth bass	32 (90)	34 (93)
Northern pike	28 (82)	30 (86)
Rainbow trout	19 (66)	24 (75)
Sauger	25 (77)	—
Smallmouth bass	29 (84)	—
Smallmouth buffalo	—	—
Sockeye salmon	18 (64)	22 (72)
Striped bass	—	—
Threadfin shad	—	—
White bass	—	—
White crappie	28 (82)	—
White sucker	28 (82)[c]	—
Yellow perch	29 (84)	—

[a]Calculated according to the equation (using optimum temperature for growth) maximum weekly average temperature for growth = optimum temperature + ½ (ultimate incipient lethal temperature − optimum temperature).

[b]Based on temperature (°C) = 1/b [$\log_{10}$(time in minutes) − a] − 2°C, acclimation at the maximum weekly average temperature for summer growth, and data in Appendix II-C of "Water Quality Criteria," published by National Academy of Sciences.

[c]Based on data for larvae (ERL-Duluth, 1976).

The winter maximum temperature must not exceed the ambient water temperature by more than the amount of change a specimen acclimated to the plume temperature can tolerate. Such a change could occur by a cessation of the source of heat or by the specimen being driven from an area by addition of biocides or other factors. However, data is inadequate to estimate a safety factor for the "no stress" level from cold shocks.

Reviews have been performed on the effects of temperature on aquatic life reproduction and development. Reproductive events are noted as perhaps the most thermally restricted of all life phases, assuming other factors are at or near optimum levels. Natural short-term temperature fluctuations appear to cause reduced reproduction of fish and invertebrates.

Inadequate data are available to quantitate the most temperature-sensitive life stages among various aquatic species. Uniform elevation of temperature a few degrees but still within the spawning range may lead to advanced spawning for spring spawning species and delays for fall spawners. such changes may not be detrimental unless asynchrony occurs between newly hatched juveniles and their normal food source. Such asynchrony may be most pronounced among anadromous species or other migrants who pass from the warmed area to normally chilled,

Table 3.— Summary of reported values for maximum weekly average temperatures for spawning and short-term maxima for embryo survival during the spawning season (centigrade and Fahrenheit).

SPECIES	SPAWNING[a]	EMBRYO SURVIVAL[b]
Atlantic salmon	5 (41)	11 (52)
Bigmouth buffalo	17 (63)	27 (81)[c]
Black crappie	17 (63)	20 (68)[c]
Bluegill	25 (77)	34 (93)
Brook trout	9 (48)	13 (55)
Carp	21 (70)	33 (91)
Channel catfish	27 (81)	29 (84)[c]
Coho salmon	10 (50)	13 (55)[c]
Emerald shiner	24 (75)	28 (82)[c]
Freshwater drum	21 (70)	26 (79)
Lake herring (Cisco)	3 (37)	8 (46)
Largemouth bass	21 (70)	27 (81)[c]
Northern pike	11 (52)	19 (66)
Rainbow trout	8 (46)	15 (59)
Sauger	12 (54)	18 (64)
Smallmouth bass	17 (63)	23 (73)[c]
Smallmouth buffalo	21 (70)	28 (82)[c]
Sockeye salmon	10 (50)	13 (55)
Striped bass	18 (64)	24 (75)
Threadfin shad	18 (64)	34 (93)
White bass	17 (63)	26 (79)
White crappie	18 (64)	23 (73)
White sucker	10 (50)	20 (68)
Yellow perch	12 (54)	20 (68)

[a]The optimum or mean of the range of spawning temperatures reported for the species (ERL-Duluth, 1976).
[b]The upper temperature for successful incubation and hatching reported for the species (ERL-Duluth, 1976).
[c]Upper temperature for spawning.

unproductive area. Reported temperature data on maximum temperatures for spawning and embryo survival have been summarized in Table 3.

Although the limiting effects of thermal addition to estuarine and marine waters are not as conspicuous in the fall, winter, and spring as during the summer season of maximum heat stress, nonetheless crucial thermal limitations do exist. Hence, thermal additions to the receiving waters must be minimized during all seasons. Size of harvestable stocks of commercial fish and shellfish, particularly near geographic limits of the fishery, appear to be markedly influenced by slight changes in the long-term temperature regime.

Studies on the relationship between temperature and annual variation in seven-year catch data for winter flounder *(Pseudopleuronected americanus)* in Narragansett Bay, Rhode Island, revealed that a 78 percent decrease in annual catch correlated closely with a 0.5°C increase in the average temperature over the 30-month period between spawning and recruitment in to the fishery. Sissenwine's 1874 model predicts a 68 percent reduction of recruitment in yellowtail flounder *(Limanda ferruginea)* with a 1°C long-term elevation in southern New England waters.

Community balance can be influenced strongly by such temperature-dependent factors as rates of reproduction, recruitment, and growth of each component population. A few degrees elevation in average monthly temperature can appreciably alter a community through changes in interspecies relationships. A 50 percent reduction in the softshell clam fishery in Maine by the green crab *(Carcinus maenus)* illustrates how an increase in winter temperatures can establish new predator-prey relationships. Over a period of four years, temperature naturally ameliorated and the monthly mean for the coldest month of each year did not fall below 2°C. This apparently precluded appreciable ice formation and winter cold kill of the green crab and permitted a major expansion of its population, with resulting increased predation of the softshell clam.

Temperature is a primary factor controlling reproduction and can influence many events of the reproductive cycle from gametogenesis to spawning. Among marine invertebrates, initiation of reproduction (gametogenesis) is often triggered during late winter by attainment of a minimum environmental threshold temperature. In some species, availability of adequate food is also a requisite. Elevated temperature can limit gametogenesis by preventing accumulation of nutrients in the gonads. This problem could be acute during the winter if food availability and feeding activity is reduced. Most marine organisms spawn during the spring and summer; gametogenesis is usually initiated during the previous fall. Some species spawn only during the fall (herring), while others spawn during the winter and early spring. At the higher latitudes, winter breeders include such estuarine community dominants as acorn barnacles *(Balanus balanus* and *B. balanoides)*, the edible blue mussel *(Mytilus edulis)*, sea urchin *Strongylocentrotus drobachiensis)*, sculpin, and the winter flounder *(Pseudopleuronectes americanus)*. The two boreal barnacles require temperatures below 10°C before egg production will be initiated. Adaptations for reproduction exist that are dependent on temperature condition closes to the natural cycle.

Juvenile and adult fish usually thermoregulate behaviorally by moving to water having temperatures closest to their thermal preference. This provides a thermal environment that approximates the optimal temperature for many physiological functions, including growth. As a consequence, fishes usually are attracted to heated water during the fall, winter, and spring. Avoidance will occur as warmer temperature exceeds the preferendum by 1 to 3°C. This response precludes problems of heat stress for juvenile and adult fishes during the summer, but several potential problems exist during the other seasons. The possibility of cold shock and death of plume-entrained fish resulting from winter plant shutdown is well recognized. Also, increased incidence of disease and a deterioration of physiological condition has been observed among plume-entrained fishes, perhaps because of insufficient food. A weight loss of approximately 10 percent for each 1°C rise in water temperature has been observed in fish when food is absent. Indirect adverse effects may also occur on the indigenous community because of increased predation pressure if thermal addition leads to a concentration of fish that are dependent on this community for their food.

Fish migration is often linked to natural environmental temperature cycles. In early spring, fish employ temperature as their environmental cue to migrate northward (e.g., menhaden, bluefish) or to move inshore (winter flounder). Likewise, water temperature strongly influences timing of spawning runs of anadromous fish into rivers. In the autumn, a number of juvenile marine fishes and shrimp are dependent on a drop in temperature to trigger their migration from estuarine nursery grounds for oceanic dispersal or southward migration.

Thermal discharges should not alter diurnal and tidal temperature variations normally experienced by marine communities. Laboratory studies show thermal tolerance to be enhanced when animals are maintained under a diurnally fluctuating temperature regime rather than at a constant temperature. A daily cyclic regime can be protective additionally as it reduces duration of exposure to extreme temperatures.

Summer thermal maxima should be established to protect the various marine communities within each biogeographic region. During the summer, naturally elevated temperatures may be of sufficient magnitude to cause death or emigration. This more commonly occurs in tropical and

warm temperate zone waters, but has been reported for enclosed bays and shallow waters in other regions as well. Summer heat stress also can contribute to increased incidence of disease or parasitism; reduced or blocked sexual maturation; inhibited or blocked embryonic cleavage of larval development; reduced feeding and growth of juveniles and adults; increased predation; and reduced productivity of macroalgae and seagrasses. The general ceilings set forth here are derived from studies delineating limiting temperatures for the more thermally sensitive species or communities of a biogeographic region.

(Quality Criteria for Water, July 1976) PB-263943
See Appendix D for Methodology.

2,3,7,8-TETRACHLORODIBENZO- P-DIOXIN (TCDD) (DIOXIN)

CRITERIA

Aquatic Life

Not enough data are available concerning the effects of 2,3,7,8-TCDD on aquatic life and its uses to derive national criteria. The available information indicates that acute values for some freshwater animal species are 1.0 μg/L; some chronic values are μg/L; and the chronic value for rainbow trout *(Oncorhynchus mykiss)* is 0.001 μg/L. Because exposures of some species of fishes to 0.01 μg/L for < 6 days resulted in substantial mortality several weeks later, derivation of aquatic life criteria for 2,3,7,8-TCDD may require special consideration.

Predicted bioconcentration factors (BCFs) for 2,3,7,8-TCDD range from 3,000 to 900,000, but the available measured BCFs range from 390 to 13,000. If the BCF is 5,000, concentrations 0.00001 μg/L should result in concentrations in edible freshwater and saltwater fish and shellfish that exceed levels identified in a Food and Drug Administration health advisory. If the BCF is 5,000 or if uptake in a field situation is greater than in laboratory tests, the value of 0.00001 μg/L will be too high.

Human Health

For the maximum protection of human health from the potential carcinogenic effects of 2,3,7,8-TCDD exposure through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero. This criterion is based on the nonthreshold assumption for 2,3,7,8-TCDD. However, zero may not be an attainable level at this time. Therefore, the levels that may result in an increase of cancer risk over the lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}, and the corresponding recommended criteria are 1.3×10^{-7} μg/L, 1.3×10^{-8} μg/L, and 1.3×10^{-9} μg/L, respectively.

If the above estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 1.4×10^{-7} μg/L, 1.4×10^{-8} μg/L, and 1.4×10^{-9} μg/L, respectively. If these estimates are made for comsumption of water only, the levels are 2.2×10^{-6} μg/L, 2.2×10^{-7} μg/L, and 2.2×10^{-8} μg/L, respectively.

(49 F.R. 5831, February 15, 1984)
See Appendix A for Aquatic Life Methodology.

TETRACHLOROETHYLENE

127-18-4

CRITERIA

Aquatic Life

For freshwater aquatic life, the available data for tetrachloroethylene indicate that acute and chronic toxicity occurs at concentrations as low as 5,280 and 840 μg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested.

For saltwater aquatic life, the available data for tetrachloroethylene indicate that acute and chronic toxicity occurs at concentrations as low as 10,200 and 450 μg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested.

Human Health

For the maximum protection of human health from the potential carcinogenic effects of exposure to tetrachloroethylene through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentrations should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be attainable. Therefore, the levels that may result in incremental increase of cancer risk over the lifetime are estimated at 10-5, 10^{-6}, and 10^{-7}. The corresponding recommended criteria are 8.0 μg/L, 0.80 μg/L, and 0.08 μg/L, respectively. If these estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 88.5 μg/L, 8.85 μg/L, and 0.88 μg/L, respectively.

(45 F.R. 79318, November 28, 1980)
See Appendix C for Human Health Methodology.

THALLIUM

7440-28-0

CRITERIA

Aquatic Life

For freshwater aquatic life, the available data for thallium indicate that acute and chronic toxicity occurs at concentrations as low as 1,400 and 40 μg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested. Toxicity to one species of fish occurs at concentrations as low as 20 μg/L after 2,600 hours of exposure.

For saltwater aquatic life, the available data for thallium indicate that acute toxicity occurs at concentrations as low as 2,130 μg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of thallium to sensitive saltwater aquatic life.

Human Health

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60848). Recalculated IRIS values for thallium are 1.7 μg/L for ingestion of contaminated water and organisms and 6.3 μg/L for ingestion of contaminated aquatic organisms only.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60848, December 22, 1992)
See Appendix C for Human Health Methodology.

TOLUENE

108-88-3

CRITERIA

Aquatic Life

The available data for toluene indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 17,500 µg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of toluene to sensitive freshwater aquatic life.

The available data for toluene indicate that acute and chronic toxicity to saltwater aquatic life occurs at concentrations as low as 6,300 and 5,000 µg/L, respectively, and would occur at lower concentrations among species that are more sensitive than those tested.

Human Health

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60848). Recalculated IRIS values for toluene are 6,800 µg/L for ingestion of contaminated water and organisms and 200,000 µg/L for ingestion of contaminated organisms only.

(45 F.R. 79318, November 28, 1980) (57 F.R. 60848, December 22, 1992)
See Appendix C for Human Health Methodology.

TOXAPHENE

8001-35-2

CRITERIA

Aquatic Life

Freshwater — 4-day average of 0.0002 µg/L
1-hour average of 0.73 µg/L
Saltwater — 4-day average of 0.0002 µg/L
1-hour average of 0.21 µg/L

Summary

The acute sensitivities of 36 freshwater species in 28 genera ranged from 0.8 µg/L to 500 µg/L. Such important fish species as the channel catfish, largemouth bass, chinook and coho salmon, brook, brown, and rainbow trout, striped bass, and bluegill had acute sensitivities ranging from 0.8 µg/L to 10.8 µg/L. Chronic values for four freshwater species range from less than 0.039 µg/L for the brook trout (*Salvelinus fontinalis*) to 0.1964 µg/L for the channel catfish (*Ictalurus punctatus*).

The growth of algae was affected at 100 to 1,000 µg/L, and bioconcentration factors from laboratory tests ranged from 3,100 to 90,000. Concentrations in lake trout in the Great Lakes have frequently exceeded the Food and Drug Administration (FDA) action level of 5 mg/kg, even though the concentrations in the water seem to be only 1 to 4 ng/L. These concentrations in lake water are thought to have resulted from toxaphene being transported to the Great Lakes from remote sites, the locations of which are not well known.

The acute toxicity of toxaphene to 15 species of saltwater animals ranges from 0.53 for pinfish (*Lagodon rhomoides*) to 460,000 µg/L for the adults of the clam (*Rangia cuneata*). Except for resistant species tested at concentrations greater than toxaphene's water solubility, acute values for most species were within a factor of 10. The toxicity of toxaphene was found to decrease slightly with increasing salinity for adult blue crabs (*Callinectes sapidus*), whereas no relationship between toxicity and salinity was observed with the threespine stickleback (*Gasterosteus aculeatus*). Temperature significantly affected the toxicity of toxaphene to blue crabs.

Early life-stage toxicity tests have been conducted with the sheepshead minnow (*Cyprinodon variegatus*) and the longnose killifish (*Fundulus similis*), whereas life-cycle tests have been conducted with the sheepshead minnow and a mysid. For the sheepshead minnow, chronic values of 1.658 µg/L from the early life-stage test and 0.7141 µg/L from the life-cycle toxicity test are similar to the 96-hour LC50 of 1.1 µg/L. Killifish are more chronically sensitive, with effects noted at 0.3 µg/L. In the life-cycle test with the mysid, no adverse effects were observed at the highest concentration tested, which was only slightly below the 96-hour LC50, resulting in an acute-chronic ratio of 1.132.

Toxaphene is bioconcentrated by an oyster, *Crassostrea virginica*, and two fishes, *C. variegatus* and *F. similis*, to concentrations that range from 9,380 to 70,140 times that in the test solution.

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 of toxaphene does not exceed 0.0002 μg/L more than once every three years on the average and if the one-hour average concentration does not exceed 0.73 μg/L more than once every three years on the average. If the concentration of toxaphene does exceed 0.0002 μg/L, the edible portions of consumed species should be analyzed to determine whether the concentration of toxaphene exceeds the FDA action level of 5 mg/kg. If the channel catfish is as acutely sensitive as some data indicate it might be, it will not be protected by this criterion.

The procedures described in the guidelines 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 toxaphene does not exceed 0.0002 μg/L more than once every three years on the average and if the one-hour average concentration does not exceed 0.21 μg/L more than once every three years on the average. If the concentration of toxaphene does exceed 0.0002 μg/L, the edible portions of consumed species should be analyzed to determine whether the concentration of toxaphene exceeds the FDA action level of 5 mg/kg.

In the EPA's best scientific judgment, three years is the average amount of time aquatic ecosystems should be provided between excursions. The resiliences of ecosystems and their abilities to recover differ greatly, however, and site-specific allowed excursion frequencies may be established if adequate justification is provided.

Criteria for developing water quality-based permit limits and for designing waste treatment facilities must be applied to an appropriate wasteload allocation model; dynamic models are preferred. Limited data or other considerations might make their use impractical; in which case, rely on a steady-state model.

Human Health

Human health criteria were recalculated using Integrated Risk Information System (IRIS) to reflect available data as of 12/92 (57 F.R. 60848). Recalculated IRIS values for toxaphene are 0.00073 μg/L for ingestion of contaminated water and organisms and 0.00075 μg/L for ingestion of contaminated organisms only.

(51 F.R. 43665, December 3, 1986) (57 F.R. 60848, December 22, 1992)

See Appendix A for Aquatic Life Methodology.

TRICHLOROETHYLENE

79-01-6

CRITERIA

Aquatic Life

The available data for trichloroethylene indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 45,000 μg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of trichloroethylene to sensitive freshwater aquatic life, but the behavior of one species is adversely affected at concentrations as low as 21,900 μg/L.

The available data for trichloroethylene indicate that acute toxicity to saltwater aquatic life occurs at concentrations as low as 2,000 μg/L and would occur at lower concentrations among species that are more sensitive than those tested. No data are available concerning the chronic toxicity of trichloroethylene to sensitive saltwater aquatic life.

Human Health

For the maximum protection of human health from the potential carcinogenic effects of exposure to trichloroethylene through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentration should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels that may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}. The corresponding recommended criteria are 27 μg/L, 2.7 μg/L, and 0.27 μg/L, respectively. If these estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 807 μg/L, 80.7 μg/L, and 8.07 μg/L, respectively.

(45 F.R. 79318, November 28, 1980)

See Appendix C for Human Health Methodology.

VINYL CHLORIDE

75-01-4

CRITERIA

Aquatic Life

No freshwater or saltwater organisms have been tested with vinyl chloride; therefore, no statement can be made concerning acute or chronic toxicity.

Human Health

For the maximum protection of human health from the potential carcinogenic effects of exposure to vinyl chloride through ingestion of contaminated water and contaminated aquatic organisms, the ambient water concentrations should be zero, based on the nonthreshold assumption for this chemical. However, zero level may not be attainable at the present time. Therefore, the levels that may result in incremental increase of cancer risk over the lifetime are estimated at 10^{-5}, 10^{-6}, and 10^{-7}. The corresponding recommended criteria are 20 μg/L, 2.0 μg/L, and 0.2 μg/L, respectively. If these estimates are made for consumption of aquatic organisms only, excluding consumption of water, the levels are 5,246 μg/L, 525 μg/L, and 52.5 μg/L, respectively.

(45 F.R. 79318, November 28, 1980)
See Appendix C for Human Health Methodology.

ZINC

7440-66-6

CRITERIA

Aquatic Life

Saltwater — 4-day average of 86 μg/L
1-hour average of 95 μg/L
Freshwater criteria are hardness dependent. See text.

Summary

Acute toxicity values available for 43 species of freshwater animals and data for eight species indicate that acute toxicity decreases as hardness increases. When adjusted to a hardness of 50 mg/L, sensitivities range from 50.70 μg/L for *Ceriodaphnia reticulata* to 88,960 μg/L for a damselfly. Additional data indicate that toxicity increases as temperature increases.

Chronic toxicity data are available for nine freshwater species. Chronic values for two invertebrates ranged from 46.73 μg/L for *Daphnia magna* to 5,243 μg/L for the caddisfly (*Clistoronia magnificia*). Chronic values for seven fish species ranged from 36.41 μg/L for the flagfish (*Jordanella floridae*) to 854.7 μg/L for the brook trout (*Salvelinus fontinalis*). Acute-chronic ratios ranged from 0.2614 to 41.20, but the ratios for the sensitive species were all less than 7.3.

The sensitivity range of freshwater plants to zinc is greater than that for animals. Growth of the alga (*Selenastrum capriocornutum*) was inhibited by 30 μg/L. On the other hand, with several other species of green algae, four-day EC50s exceeded 200,000 μg/L. Zinc was found to bioaccumulate in freshwater animal tissues from 51 to 1,130 times the concentration present in the water.

Acceptable acute toxicity values for zinc are available for 33 species of saltwater animals including 26 invertebrates and 7 fish. LC50s range from 191.5 μg/L for cabezon (*Scorpanichthys marmoratus*) to 320,400 μg/L for adults of another clam, *Macoma balthica*. Early life stages of saltwater invertebrates and fishes are generally more sensitive to zinc than juveniles and adults. Temperature has variable and inconsistent effects on the sensitivity of saltwater invertebrates to zinc. The sensitivity of saltwater vertebrate animals to zinc decreases with increasing salinity, but the magnitude of the effect is species-specific.

A life-cycle test with the mysid (*Mysidopsia bahia*) found unacceptable effects at 120 μg/L but not at 231 μg/L, and the acute-chronic ratio was 2.997. Seven species of saltwater plants were affected at concentrations ranging from 19 to 10,100 μg/L. Bioaccumulation data for zinc are available for seven species of saltwater algae and five species of saltwater animals. Steady-state zinc bioconcentration factors for the 12 species range from 3.692 to 23,820.

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 of zinc (in μg/L) does not exceed the numerical value given by

$$e^{(0.8473[\ln (\text{hardness})]+0.7614)}$$

more than once every three years on the average and if the one-hour average concentration (in μg/L) does not exceed the numerical value given by

$$e^{(0.8473[\ln (\text{hardness})]+0.8604)}$$

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 zinc are 59, 110 and 190 μg/L, respectively, and the one-hour average concentrations are 65, 120, and 210 μg/L. If the striped bass is as sensitive as some data indicate, it will not be protected by this criterion.

The procedures described in the guidelines 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 zinc does not exceed 86 μg/L more than once every three years on the average and if the one-hour average concentration does not exceed 95 μg/L more than once every three years on the average.

Criteria for developing water quality-based permit limits and for designing waste treatment facilities must be applied to an appropriate wasteload allocation model; dynamic models are preferred. Limited data or other considerations might make their use impractical, in which case rely on a steady-state model.

(52 F.R. 6213, March 2, 1987)

See Appendix A for Aquatic Life Methodology.

APPENDIX A

Derivation of the 1985 Aquatic Life Critera

The following is a summary of the Guidelines for Derivation of Criteria for Aquatic Life. The complete text is found in "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses," available from National Technical Information Service – PB85-227049.

Derivation of numerical national water quality criteria for the protection of aquatic organisms and their uses is a complex process that uses information from many areas of aquatic toxicology. When 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 water quality characteristics 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-day 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 (FDA), or if such residues might harm wildlife that consumes 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 exists, 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.

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 exist only in large quantities as commercial mixtures of the various compounds and apparently have similar biological, chemical, physical, and toxicological properties.

B. For chemicals that do ionize substantially in most natural waterbodies (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 in chemical equilibrium should usually be considered one material. Each different oxidation state of a metal and each different non-ionizable 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 sample. Thus, it is also necessary to reference or describe one 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, compatible with the available toxicity and bioaccumulation data without making overly hypothetical extrapolations, and 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 be used. This compromise measurement must fit with the general approach: 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 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. 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.

The analytical chemistry of the material might need to be considered 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 toxicity to, and bioaccumulation by, aquatic animals and plants; FDA action levels and 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) with enough supporting information to indicate that acceptable test procedures were used and that the results are probably reliable. In some cases, additional written information from the investigator may be needed. Information that is confidential, privileged, or otherwise not available for distribution should not be used.

C. Questionable data, whether published or unpublished, should not be used. Examples would be data 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 may not be used.

E. For some highly volatile, hydrolyzable, or degradable materials, only use data from flow-through tests in which the concentrations of test material were measured often enough with acceptable analytical methods.

F. Data should be rejected if obtained by using:

- Brine shrimp — because they usually occur naturally only in water with salinity greater than 35 g/kg;
- Species that do not have reproducing wild populations in North America; or
- 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 nonresident species 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 to indicate the sensitivities of appropriate untested species. However, since procedures for conducting tests with aquatic plants and interpreting the results are not as well developed, fewer data concerning toxicity are required. Finally, data concerning bioaccumulation by aquatic organisms are required only when relevant information on the significance of residues in aquatic organisms is available.

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 including all of the following:
   - The family Salmonidae in the class Osteichthyes.
   - A second family in the class Osteichthyes, preferably a commercially or recreationally important warmwater species, such as bluegill or channel catfish.
   - A third family in the phylum Chordata (may be in the class Osteichthyes or may be an amphibian, etc.).
   - A planktonic crustacean such as a cladoceran or copepod.
   - A benthic crustacean (ostracod, isopod, amphipod, crayfish, etc.).
   - An insect (mayfly, dragonfly, damselfly, stonefly, caddisfly, mosquito, midge, etc.).
   - A family in a phylum other than Arthropoda or Chordata, such as Rotifera, Annelida, Mollusca.
   - 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:
   - At least one is a fish;
   - At least one is an invertebrate; and
   - 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 the plants are among the aquatic organisms that are most sensitive to the material, test data on 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, including all of the following:
   - Two families in the phylum Chordata;
   - A family in a phylum other than Arthropoda or Chordata;
   - Either the Mysidae or Penaeidae family;
   - Three other families not in the phylum Chordata (may include Mysidae or Penaeidae, whichever was not used previously); and
   - 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; and
   - At least one is an acutely sensitive saltwater species (the other may be an acutely sensitive 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 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 10 with no apparent pattern. Also, if a criterion is to be related to a water quality characteristic T (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 the 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 genera used in acceptable acute tests 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 to protect that important species.

B. Acute toxicity tests should have been conducted using acceptable procedures such as those described in

1. ASTM Standard E 729-80, Practice for Conducting Acute Toxicity Tests with Fishes, Macroinvertebrates, and Amphibians. American Society for Testing and Materials, 1916 Race Street, Philadelphia, PA 19103.

2. ASTM Standard E 724-80, Practice for Conducting Static Acute Toxicity Tests with Larvae of Four Species of Bivalve Molluscs. American Society for Testing and Materials, 1916 Race Street, Philadelphia, PA 19103.

C. Except for tests with saltwater annelids and mysids, do not use results of acute tests during which test organisms were fed, 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 the organic carbon or particulate matter does not affect toxicity.

E. Acute values should be based on endpoints that 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 stressed with second- or third-instar larvae. The result should be the 48-hour EC50 based on percentage of organisms immobilized plus percentage of organisms killed. If such an EC50 is not available from a test, the 48-hour LC50 should be used in place of the desired 48-hour EC50. An EC50 or LC50 of longer than 48 hours 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-hour 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-hour EC50, based on the percentage of organisms with incompletely developed shells and the 96-hour LC50 should be used in place of the desired 96-hour EC50. If the duration of the test was between 48 and 96 hours, 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-hour 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-hour LC50 should be used in place of the desired 96-hour 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 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 2 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 be considered protected from acute toxicity only 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 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 Species Mean Acute Value should be calculated as the geometric mean of all available acute values — i.e., results of flow-through tests in which the concentrations were not measured and results of static and renewal tests based on initial concentrations of test material. (Nominal concentrations are acceptable for most test materials if measured concentrations are not available.)

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

NOTE: The geometric mean of N numbers is the N^{th} 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 individual organisms' sensitivities in toxicity tests on most materials, and the distributions of species' sensitivities 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. The Genus Mean Acute Value should be calculated as the geometric mean of the Species Mean Acute Values available for each genus.

K. Order the Genus Mean Acute Value from high to low.

L. Assign ranks, R, to the Genus Mean Acute Value from "1" for the lowest to "N" for the highest. If two or more Genus Mean Acute Values are identical, arbitrarily assign them successive ranks.

M. Calculate the cumulative probability, P, for each Genus Mean Acute Value as R/(N+1).

N. Select the four Genus Mean Acute Values that have cumulative probabilities closest to 0.05. (If there are less than 59 Genus Mean Acute Values, these will always be the four lowest Genus Mean Acute Values).

O. Using the selected Genus Mean Acute Values and Ps, calculate:

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

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

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

$$FAV = e^A$$

(See original document, referenced at beginning of this appendix, for development of the calculation procedure and for an example calculation 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 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 section IV, steps B through G, or using analysis of covariance. 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 tests. 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 percent confidence limits for each species.

NOTE: Because the best documented relationship is that between hardness and acute toxicity of metals in freshwater 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.

C. Decide whether the data for each species are 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 based only on data for a very narrow range of water quality characteristic values. A slope based on only two data points, however, might be useful if 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, probably some or all of the values should be rejected. 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 these acute values by the mean for the species. This normalizes the 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 percent confidence limits will be identical to those obtained in step B. However, now, 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 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 percent 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.)

I. For each species, calculate the logarithm, Y, of the Species Mean Acute Value 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 Species Mean Acute Values at Z can be obtained by skipping step H, 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, steps J through O.

L. If the Species Mean Acute Value at Z of a commercially or recreationally important species is lower than the calculated Final Acute Value at Z, then that Species Mean Acute Value 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:

$$\text{Final Acute Value} = e^{(V[\ln(\text{water quality characteristic})] + \ln A - V[\ln Z])}$$

where

V = pooled acute slope

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 Ratio 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 1; 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 chronic tests in which the concentrations of test material in the test solutions were properly measured at appropriate times during the test. (Exception: renewal, which is acceptable for daphnids.)

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 (and so forth) 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 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 seven 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 in a fish species to a 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 two 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 fish species 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 total 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 should not be used because these results are possibly not good predictions of the results of comparable life-cycle or partial life-cycle tests.

F. A chronic value can 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 in an acceptable chronic test that did not cause an unacceptable amount of adverse effect on any of the specified biological measurements and below which no tested concentration caused an unacceptable effect. An upper chronic limit is the lowest tested concentration in an acceptable chronic test that did cause an unacceptable amount of adverse effect on one or more of the specified biological measurements and 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 percent) might be considered acceptable even if it is statistically significantly different from the control, whereas a large percent reduction (e.g., 30 percent) might be considered unacceptable even if it is not statistically significant.

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 should also 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; appropriate Genus Mean Chronic Values should also be calculated. The Final Chronic Value should then be obtained using the procedure described in section III, steps J through 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 acute tests in the same dilution water and in which the concentrations were measured. (Exception: static is acceptable for daphnids.)

   For fish, the acute test(s) should have been conducted with juveniles and 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 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 ratio seems to increase or decrease as the Species Mean Acute Value increases, the Final Acute-Chronic Ratio should be calculated as the geometric mean of the acute-chronic ratios for species whose Species Mean Acute Values 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 10, 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 the sensitivities of embryos and larvae would likely determine the results of life-cycle tests. Thus, if the lowest available Species Mean Acute Values 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 will result in the chronic slope being the same as the acute slope. The procedure described in steps B through N usually will result in the chronic slope being different from the acute 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 steps B through G or using analysis of covariance. The two methods are equivalent and produce identical results. The manual method described in the following paragraphs 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.

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 percent 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 are 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 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, probably some or all of the values should be rejected.

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, the chronic slope is probably 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 its mean. 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 percent confidence limits will be identical to those obtained in section B. 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 percent 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.)

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, 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, steps J through 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 Chronic Value should be used as the Final Chronic Value at Z instead of the calculated Final Chronic Value.

N. The Final Chronic Equation is written as:

$$\text{Final Chronic Value} = e^{(L[\ln(\text{water quality characteristic})] + \ln S - L[\ln Z])}$$

where

L = pooled chronic slope

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-hour 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 prevent concentrations in commercially or recreationally important aquatic species from affecting marketability because they exceed applicable FDA action levels and to 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 level (Compliance Policy Guide, U.S. Food & Drug Admin. 1981) for fish oil or for the edible portion of fish or shellfish, or 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 must 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 net uptake from both food and water in a real-world situation. A BAF almost must be measured in a field situation in which predators accumulate the material directly from water and by consuming prey that 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 adequate measurements must be made of the material's concentration in water to ascertain if it was reasonably constant for a long enough 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 derive from the material and its metabolites that are structurally similar and are not much more soluble in water than the parent material.

   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 2 days or 16 percent of the length of the exposure, whichever is longer. The BCF used from a test should be the highest of the apparent steady-state BCF, if apparent steady state was reached; the highest BCF obtained, if apparent steady state was not reached; and 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 invertebrates.

5. If more than one acceptable BCF is available for a species, the geometric mean of the available values should be used; however, if the BCFs are from different lengths of exposure and the BCF increases with length of exposure, then 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 that constitutes or represents 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, calculating additional residue values is possible. 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, 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 1 percent lipid basis by dividing it by the percent lipids. This adjustment to a 1 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 they 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.,

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

- For an FDA action level for fish oil, the appropriate percent lipids value is 100.
- 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 species-by-species 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.
- 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 that 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 a FDA action level will probably result in an average concentration in the edible portion of a fatty species 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 that 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 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 they 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 Criterion Continuous Concentration 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 both the Criterion Maximum Concentration and the Criterion Continuous Concentration to two significant digits.

E. The criterion is stated as follows:

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) µg/L more than once every three years on the average, and if the one-hour average concentration does not exceed (4) µg/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 10 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 not, another criterion — either higher or lower — should be derived using appropriate modifications of these guidelines.

APPENDIX B

Derivation of the 1980 Aquatic Life Criteria

This version of the Guidelines provides clarifications, additional details, and technical and editorial changes in the last version published in the Federal Register [44 FR 15970 (March 15, 1979)]. It incorporates changes resulting from comments on previous versions and from experience gained during EPA's use of the previous versions. Future versions of the Guidelines will incorporate new ideas and data as their usefulness is demonstrated.

Criteria may be expressed in several forms. The numerical form is commonly used, but descriptive and procedural forms can be used if numerical criteria are not possible or desirable. The purpose of these Guidelines is to describe an objective, internally consistent and appropriate way of deriving numerical water quality criteria for the protection of the uses and presence of aquatic organisms.

A numerical criterion is an estimate of the highest concentration of a substance in water that does not present a significant risk to the aquatic organisms in the water and their uses. Thus the Guidelines are intended to derive criteria that will protect aquatic communities by protecting most of the species and their uses most of the time, but not necessarily all of the species all of the time. Aquatic communities can tolerate some stress and occasional adverse effects on a few species, and so total protection of all species all of the time is not necessary. Rather, the Guidelines attempts to provide a reasonable and adequate amount of protection with only a small possibility of considerable overprotection or underprotection. Within these constraints, it seems appropriate to err on the side of overprotection.

The numerical aquatic life criteria derived using the Guidelines are expressed as two numbers, rather than the traditional one number, so that the criteria can more accurately reflect toxicological and practical realities. The combination of both a maximum value and a 24-hour average value is designed to provide adequate protection of aquatic life and its uses from acute and chronic toxicity to animals, toxicity to plants, and bioconcentration by aquatic organisms without the restrictions of a one-number criterion to provide the same amount of protection. The only way to assure the same degree of protection with a one-number criterion would be to use the 24-hour average as a concentration that is not to be exceeded at any time in any place.

The two-number criterion is intended to identify an average pollutant concentration that will produce a water quality generally suited to the maintenance of aquatic life and its uses, while restricting the extent and duration of excursions over the average so that the total exposure will not cause unacceptable adverse effects. Merely specifying an average value over a time period is insufficient, unless the period of time is rather short, because concentration higher than the average value can kill or cause substantial damage in short periods. Furthermore, for some substances the effect of intermittent high exposures is cumulative. Therefore, placing an upper limit on pollutant concentrations to which aquatic organisms might be exposed is necessary, especially when the maximum value is not much higher than the average value. For some substances, the maximum may be so much higher than the 24-hour average that in any real-world situation the maximum will never be reached if the 24-hour average is achieved. In such cases, the 24-hour average will be limiting and the maximum will have no practical significance, except to indicate that elevated concentrations are acceptable as long as the 24-hour average is achieved.

These Guidelines have been developed on the assumption that the results of laboratory tests are generally useful for predicting what will happen in field situations. The resulting criteria are meant to apply to most U.S. bodies of water, except for the Great Salt Lake. All aquatic organisms and their common uses are meant to be considered, but not necessarily protected, if relevant data are available,

with at least one specific exception. This exception is the accumulation of residues of organic compounds in the siscowet subspecies of lake trout which occurs in Lake Superior and contains up to 67 percent fat in the fillets (Thurston, C.E., 1962, Physical Characteristics and Chemical Composition of Two Subspecies of Lake Trout, J. Fish. Res. Bd. Canada 19:39-44). Neither siscowet nor organisms in the Great Salt Lake are intentionally protected by these Guidelines because both may be too atypical.

With appropriate modifications, these Guidelines can be used to derive criteria for any specified geographical area, body of water (such as the Great Salt Lake), or group of similar bodies of water. Thus, with appropriate modifications, the Guidelines can be used to derive national, state, or local criteria if adequate information is available concerning the effects of the substance of concern on appropriate species and their uses. However, the basic concepts described in the Guidelines should be modified only when sound scientific evidence indicates that a criterion produced using the Guidelines would probably significantly overprotect or underprotect the presence or uses of aquatic life.

Criteria produced by these Guidelines are not enforceable numbers. They may be used in developing enforceable numbers, such as water quality standards and effluent standards. However, the development of standards may take into account additional factors such as social, legal, economic, and hydrological considerations, the environmental and analytical chemistry of the substance, the extrapolation from laboratory data to field situations, and the relationship between the species for which data are available and the species to be protected.

Because fresh water and salt water (including both estuarine and marine waters) have basically different chemical compositions and because freshwater and saltwater species rarely inhabit the same water simultaneously, separate criteria should be derived for these two kinds of waters. However, for some substances sufficient data may not be available to allow derivation of one or both of these criteria using the Guidelines.

These Guidelines are meant to be used after a decision is made that a criterion is needed for a substance. The Guidelines do not address the rationale for making that decision. If the potential for adverse effects on aquatic life and its uses are part of the basis for deciding whether or not a criterion is needed for a substance, these Guidelines may be helpful in collecting and interpretating relevant data.

I. Define the Substance for Which the Criterion is to be Derived

A. Each separate chemical that would not ionize significantly in most natural bodies of water should usually be considered a separate substance, except possibly for structurally similar organic compounds that only differ in the number and location of atoms of a specific halogen, and only exist in large quantities as commercial mixtures of the various compounds, and apparently have similar chemical, biological, and toxicological properties.

B. For chemicals that would ionize significantly in most natural bodies of water, such as inorganic salts, organic acids and phenols, all forms that would be in chemical equilibrium should usually be considered one substance. For metals, each different valence and each different covalently bonded organometallic compound should usually be considered a separate substance.

C. The definition of the substance may also need to take into account the analytical chemistry and fate of the substance.

II. Collect and Review Available Data

A. Collect all available data on the substance concerning

1. toxicity to, and bioaccummulation by, aquatic animals and plants
2. FDA action levels
3. chronic feeding studies with wildlife

B. Discard all data that are not available in hard copy (publication, manuscript, letter, memorandum, etc.) with enough supporting information to indicate that acceptable test procedures were used and that the results are reliable. Do not assume that all published data are acceptable.

C. Discard questionable data. For example, discard data from tests for which no control treatment existed, in which too many organisms in the control treatment died or showed signs of stress or disease, or in which distilled or deionized water was used as the dilution water for aquatic organisms. Discard data on formulated mixtures and emulsifiable concentrates of the substance of concern, but not necessarily data on technical grade material.

D. Do not use data 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 resident in — but not necessarily native to — North America. Resident North American species of fishes are defined as those listed in "A List of Common and Scientific Names of Fishes from the United States and Canada," 3rd ed., Special Publication No. 6, American Fisheries Society, Washington, D.C., 1970. Data obtained with non-resident species can be used to indicate relationships and possible problem areas, but cannot be used in the derivation of criteria.

3. Organisms that were previously exposed to significant concentrations of the test material or other pollutants.

III. Minimum Data Base

A. A minimum amount of data should be available to help ensure that each of the four major kinds of possible adverse effects receives some consideration. Results of acute and chronic toxicity tests with a reasonable number and variety of aquatic animals are necessary so that data available for tested species can be considered a useful indication of the sensitivities of the numerous untested species. The requirements concerning toxicity to aquatic plants are less stringent because procedures for conducting tests with plants are not as well developed and the interpretation of the results is more questionable. Data concerning bioconcentration by aquatic organisms can only be used if other relevant data are available.

B. To derive a criterion for freshwater aquatic life, the following should be available:

1. Acute tests (see section IV) with freshwater animals in at least eight different families provided that of the eight species at least one
   - Is a salmonid fish
   - Is a non-salmonid fish
   - Is a planktonic crustacean
   - Is a benthic crustacean
   - Is a benthic insect
   - Of the benthic species is a detritivore

2. Acute-chronic ratios (see section VI) for at least three species of aquatic animals provided that of the three species at least one is
   - A fish
   - An invertebrate
   - A freshwater species (the other two may be saltwater species)

3. At least one test with a freshwater alga or a chronic test with a freshwater vascular plant (see section VIII). If plants are among the aquatic organisms that are most sensitive to the substance, tests with more than one species should be available.

4. At least one acceptable bioconcentration factor determined with an aquatic animal species, if a maximum permissible tissue concentration is available (see section IX).

C. To derive a criterion for saltwater aquatic life, the following should be available:

1. Acute tests (see section IV) with saltwater animals in at least eight different families provided that of the eight species
   - At least two different fish families are included
   - At least five different invertebrate families are included
   - Either the Mysidae or Penaeidae family or both are included
   - At least one of the invertebrate families is in a phylum other than Arthropoda

2. Acute-chronic ratios (see section VI) for at least three species of aquatic animals provided that of the three species at least one is
   - A fish
   - An invertebrate
   - A saltwater species (the other two may be freshwater species)

3. At least one test with a saltwater vascular plant (see section VIII). If plants are among the aquatic organisms most sensitive to the substance, tests with more than one species should be available.

4. At least one acceptable bioconcentration factor determined with an aquatic animal species, if a maximum permissible tissue concentration is available (see section IX).

D. If all the requirements of the minimum data base are met, a criterion can usually be derived, except in special cases. For example, a criterion might not be possible if the acute-chronic ratios vary greatly 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 the minimum data requirements are not satisfied, generally a criterion should not be derived, except in special cases. One such special case would be when less than the minimum amount of acute and chronic data are available, but the available data clearly indicate that the Final Residue Value would be substantially lower then either the Final Chronic Value or the Final Plant Value.

IV. Final Acute Value

A. Appropriate measures of the acute (short-term) toxicity of the substance to various species of aquatic animals are used to calculate the Final Acute Value. If acute values are available for fewer than 20 species, the Final Acute Value probably should be lower than the lowest value. On the other hand, if acute values are available for more than 20 species, the Final Acute Value probably should be higher than the lowest value, unless the most sensitive species is an important one. Although the procedure used to calculate the Final Acute Value has some limitations, it apparently is the best of the procedures currently available.

B. Acute toxicity tests should be conducted using procedures such as those described in

1. ASTM Standard E 729-80, Practice for Conducting Acute Toxicity Tests with Fishes, Macroinvertebrates, and Amphibians. American Society for Testing and Materials, 1916 Race Street, Philadelphia, PA 19103.

2. ASTM Standard E 724-80, Practice for Conducting Static Acute Toxicity Tests with Larvae of Four Species of Bivalve Molluscs. American Society for Testing and Materials, 1916 Race Street, Philadelphia, PA 19103.

C. Results of acute tests in which food was added to the test solutions should not be used, because this may unnecessarily affect the results of the test.

D. Results of acute tests conducted with embryos should not be used (but see section IV.E.2), because this is often an insensitive life stage.

E. Acute values should be based on endpoints and lengths of exposure appropriate to the life stage of the species tested. Therefore, only the following kinds of data on acute toxicity to aquatic animals should be used:

1. 48-hour EC50 values based on immobilization and 48-hour LC50 values for first-instar (less than 24 hours old) daphnids and other cladocerans, and second- or third-instar midge larvae.

2. 48- to 96-hour EC50 values based on incomplete shell development and 48- to 96-hour LC50 values for embryos and larvae of barnacles, bivalve molluscs (clams, mussels, oysters, and scallops), sea urchins, lobsters, crabs, shrimps, and abalones.

3. 96-hour EC50 values based on decreased shell deposition for oysters.

4. 96-hour EC50 values on immobilization or loss of equilibrium or both and 96-hour LC50 values for aquatic animals, except for cladocerans, midges, and animals whose behavior or physiology allows them to avoid exposure to toxicant or for whom the acute adverse effect of the exposure cannot be adequately measured. Such freshwater and saltwater animals include air-breathing molluscs, unionid clams, operculate snails, and bivalve molluscs, except for some species that cannot "close up" and thus prevent exposure to toxicant, such as the bay scallop (*Argopecten irradians*).

F. For the use of LC50 or EC50 values for durations shorter and longer than those listed above, see section X.

G. If the acute toxicity of the substance to aquatic animals has been shown to be related to a water quality characteristic such as hardness for freshwater organisms or salinity for saltwater organisms, a Final Acute Equation should be derived based on that water quality characteristic. Go to section V.

H. If the acute toxicity of the substance has not been adequately shown to be related to a water quality characteristic, for each species for which at least one acute value is available, calculate the geometric mean of the results of all flow-through tests in which the toxicant concentrations were measured. For a species for which no such result is available, calculate the geometric mean of all available acute values — i.e., results of flow-through tests in which the toxicant concentrations were not measured and results of static and renewal tests based on initial total toxicant concentrations.

NOTE: The geometric mean of N numbers is obtained by taking the N^{th} root of the product of 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 can also be calculated as the square root of the product of the two numbers. 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.

I. Count the number = N of species for which a species mean acute value is available.

J. Order the species mean acute values from low to high. Take the common logarithms of the N values (log mean values).

K. The intervals (cell widths) for the lower cumulative proportion calculations are 0.11 common log units apart, starting from the lowest log value. The value of 0.11 is an estimate of average precision and was calculated from replicate species acute values.

L. Starting with the lowest log mean value, separate the N values into intervals (or cells) calculated in step IV.K.

M. Calculate cumulative proportions for each non-empty interval by summing the number of values in the present and all lower intervals and dividing by N. These calculations only need to be done for the first three non-empty intervals (or cells).

N. Calculate the arithmetic mean of the log mean values for each of the three intervals.

O. Using the two interval mean acute values and cumulative proportions closest to 0.05, linearly extrapolate or interpolate to the 0.05 log concentration. The Final Acute Value is the antilog of the 0.05 concentration.

In other words, where

Prop (1) and conc (1) are the cumulative proportion and mean log value for the lowest non-empty interval.

Prop (2) and conc (2) are the cumulative proportion and mean log value for the second lowest non-empty interval.

A = Slope of the cumulative proportions

B = The 0.05 log value

then:

$A = [0.05 - \text{Prop (1)}] / [\text{Prop(2)} - \text{Prop (1)}]$

$B = \text{conc (1)} + A [\text{conc (2)} - \text{conc (1)}]$

Final Acute Value = 10^5

P. If for an important species, such as a recreationally or commercially important species, the geometric mean of the acute values from flow-through tests in which the toxicant concentrations were measured is lower than the Final Acute Value, then that geometric mean should be used as the 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 affected by a water quality characteristic, this effect can be taken into account as described below. Pooled regression analysis should produce similar results, although data available for individual species would be weighted differently.

B. For each species for which comparable acute toxicity values are available at two or more different values of a water quality characteristic which apparently affects toxicity, perform a least squares regression of the natural logarithms of the acute toxicity values on the natural logarithms of the values of the water quality characteristic. Natural logarithms (logarithms to the base e, denoted as ln) are used herein merely because they are easier to use on some hand calculators and computers than common logarithms (logarithms to the base 10). Consistent use of either will produce the same result. No transformation or a different transformation may be used if it fits the data better, but appropriate changes will be necessary throughout this section.

C. Determine whether or not each acute slope is meaningful, taking into account the range and number of values of the water characteristic tested. For example, a slope based on four data points may be of limited value if it is based only on data for a narrow range of values of the water quality characteristic. On the other hand, a slope based on only two data points may be meaningful if it is consistent with other information and if the two points cover a broad enough range of the water quality characteristic. If meaningful slopes are not available for at least two species, or if the available slopes are not similar, return to section IV.H., using the results of tests conducted under conditions and in water similar to those commonly used for toxicity tests with the species.

D. Calculate the mean acute slope (V) as the arithmetic average of all the meaningful acute slopes for individual species.

E. For each species calculate the geometric mean (W) of the acute toxicity values and the geometric mean (X) of the related values of the water quality characteristics.

F. For each species calculate the logarithmic intercept (Y) using the equation:

$$Y = \text{in } W - V(\ln X).$$

G. For each species calculate the species mean acute intercept as the antilog of Y.

H. Obtain the Final Acute Intercept by using the procedure described in section IV.I-O, except insert "Intercept" for "Value."

I. If for an important species, such as a recreationally or commercially important species, the intercept calculated only from results of flow-through tests in which the toxicant concentrations were measured is lower than the Final Acute Intercept, then that intercept should be used as the Final Acute Intercept.

J. The Final Acute Equation is written as

$$e^{(v[\ln(\text{water quality characteristic})+\ln z)}$$

where

V = mean acute slope

Z = Final Acute Intercept.

VI. Final Chronic Value

A. The Final Chronic Value can be calculated in the same manner as the Final Acute Value or by dividing the Final Acute Value by the Final Acute Chronic Ratio, depending on the data available. In some cases it will not be possible to calculate a Final Chronic Value.

B. Use only the results of flow-through (except renewal is acceptable for daphnids) chronic tests in which the concentrations of toxicant in the test solutions were measured.

C. Do not use the results of any chronic test in which survival, growth, or reproduction among the controls was unacceptably low.

D. Chronic values should be based on endpoints and lengths of exposure appropriate to the species. Therefore, only the results of the following kinds of chronic toxicity tests should be used:

1. Life-cycle toxicity tests consisting of exposures of each of several groups of individuals of a species to a different concentration of the toxicant throughout a life cycle. To ensure that all life stages and life processes are exposed, the test should begin with embryos or newly hatched young less than 48 hours old (less than 24 hours old for daphnids), continue through maturation and reproduction, and with fish should end not less than 24 days (90 days for salmonids) after the hatching of the next generation. For fish, data should be obtained and analyzed on survival and growth of adults and young, maturation of males and females, embryos spawned per female, embryo viability (salmonids only) and hatchability. For daphnids, data should be obtained and analyzed on survival and young per female.

2. Partial life-cycle toxicity tests consisting of exposures of each of several groups of individuals of a species of fish to a different concentration of the toxicant through most portions of a life cycle. Partial life-cycle tests are conducted with fish species that require more than a year to reach sexual maturity, so that the test can be completed in less than 15 months, but still expose all major life stages to the toxicant. Exposure to

the toxicant begins with immature juveniles at least 2 months prior to active gonad development, continues through maturation and reproduction, and ends 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, embryos spawned per female, embryo viability (salmonids only), and hatchability.

3. Early-life-stage toxicity tests consisting of 28- to 32-days (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.

E. Do not use the results of an early-life-stage test if results of a life-cycle or partial life-cycle test with the same species are available.

F. A chronic value is obtained by calculating the geometric mean of the lower and upper chronic limits from a chronic test. A lower chronic limit is the highest tested concentration

1. in an acceptable chronic test,
2. which did not cause the occurrence (which was statistically significantly different from the control at $p = 0.05$) of a specified adverse effect, and
3. below which no tested concentration caused such an occurrence.

An upper chronic limit is the lowest tested concentration

1. in an acceptable chronic test,
2. which did cause the occurrence (which was statistically significantly different from the control at $p = 0.05$) of a specified adverse effect, and
3. above which all tested concentrations caused such an occurrence.

NOTE: Various authors have used a variety of terms of definitions to interpret the results of chronic tests, so reported results should be reviewed carefully.

G. If the chronic toxicity of the substance to aquatic animals has been adequately shown to be related to a water quality characteristic such as hardness for freshwater organisms or salinity for saltwater organisms, a Final Chronic Equation should be derived based on that water quality characteristic. Go to section VII.

H. If chronic values are available for eight species as described in section III. B.1 or C.1, a species' mean chronic value should be calculated for each species for which at least one chronic value is available by calculating the geometric mean of all the chronic values for the species. The Final Chronic Value should then be obtained using the procedures described in section IV.I-O. Then go to section VI.M.

I. For each chronic value for which at least one appropriate acute value is available, calculate an acute-chronic ratio, using for the numerator the arithmetic average of the results of all standard flow-through acute tests in which the concentrations were measured and which are from the same study as the chronic test. If such an acute test is not available, use for the numerator the results of a standard acute test performed at the same laboratory with the same species, toxicant, and dilution water. If no such acute test is available, use the species mean acute value for the numerator.

NOTE: If the acute toxicity or chronic toxicity or both of the substance have been adequately shown to be related to a water quality characteristic, the numerator and the denominator must be based on tests performed in the same water.

J. For each species, calculate the species mean acute-chronic ratio as the geometric mean of all the acute-chronic ratios available for that species.

K. For some substances the species mean acute-chronic ratio seems to be the same for all species; but for other substances the ratio seems to increase as the species mean acute value increases. Thus the Final Acute-Chronic Ratio can be obtained in two ways, depending on the data available.

1. If no major trend is apparent and the acute-chronic ratios for a number of species are within a factor of 10, 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.

2. If the species' mean acute-chronic ratio seems to increase as the species' mean acute value increases, the value of the acute-chronic ratio for species whose acute values are close to the Final Acute Value should be chosen as the Final Acute-Chronic Ratio.

L. Calculate the Final Chronic Value by dividing the Final Acute Value by the Final Acute-Chronic Ratio.

M. If the species mean chronic value of an important species, such as a commercially or recreationally important species, is lower than the Final Chronic Value, then that species' mean chronic value should be used as the Final Chronic Value.

N. Go to section VIII.

VII. Final Chronic Equation

A. For each species for which comparable chronic toxicity values are available at two or more different values of a water quality characteristic that apparently affects chronic toxicity, perform a least squares regression of the natural logarithms of the chronic toxicity values on the natural logarithms of the water quality characteristic values. No transformation or a different transformation may be used if it fits the data better, but 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.

B. Determine whether or not each chronic slope is meaningful, taking into account the range and number of values of the water quality characteristic tested. For example, a slope based on four data points may be of limited value if it is based only on data for a narrow range of values of the water quality characteristic. On the other hand, a slope based on only two data points may be meaningful if it is consistent with other information and if the two points cover a broad enough range of the water quality characteristic. If a meaningful chronic slope is not available for at least one species, return to section VI.H.

C. Calculate the mean chronic slope (L) as the arithmetic average of all the meaningful chronic slopes for individual species.

D. For each species calculate the geometric mean (M) of the toxicity values and the geometric mean (P) of the related values of the water quality characteristic.

E. For each species calculate the logarithmic intercept (Q) using the equation:

$$Q = \ln M - L(\ln P).$$

F. For each species calculate a species mean chronic intercept as the antilog of Q.

G. Obtain the Final Chronic Intercept by using the procedure described in section IV. I-O, except insert "Intercept" for "Value."

H. If the species mean chronic intercept of an important species, such as a commercially or recreationally important species, is lower than the Final Chronic Intercept, then that species' mean chronic intercept should be used as the Final Chronic Intercept.

I. The Final Chronic Equation is written as

$$e^{(L[\ln(\text{water quality characteristic})]+\ln R)}$$

where

L = mean chronic slope

R = Final Chronic Intercept.

VIII. Final Plant Value

A. Appropriate measures of the toxicity of the substance to aquatic plants are used to compare the relative sensitivities of aquatic plants and animals.

B. A value is a concentration that decreased growth (as measured by dry weight, chlorophyll, etc.) in a 96-hour or longer test with an alga or in a chronic test with an aquatic vascular plant.

C. Obtain the Final Plant Value by selecting the lowest plant value from a test in which the toxicant concentrations were measured.

IX. Final Residue Value

A. The Final Residue Value is derived in order to

1. Prevent commercially or recreationally important aquatic organisms from exceeding relevant FDA action levels, and

2. Protect wildlife, including fishes and birds, that eat aquatic organisms from demonstrated adverse effects.

A residue value is calculated by dividing a maximum permissible tissue concentration by an appropriate bioconcentration factor (BCF), where the BCF is the quotient of the concentration of a substance in all or part of an aquatic organism divided by the concentration in water to which the organism has been exposed. A maximum permissible tissues concentration is either

1. An action level from the FDA Administrative Guidelines Manual for fish oil or for the edible portion of fish or shellfish, or

2. A maximum acceptable dietary intake based on observations on survival, growth, or reproduction in a chronic wildlife feeding study.

If no maximum permissible tissue concentration is available, go to section X because no Final Residue Value can be derived.

B. Bioconcentration factor

1. A BCF determined in a laboratory test should be used only if it was calculated based on measured concentrations of the substance in the test solution and was based on an exposure that continued until either 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. If a steady-state BCF is not available for a species, the available BCF for the longest exposure over 28 days should be used for that species.

2. A BCF from a field exposure should be used only when it is known that the concentration of the substance was reasonably constant for a long enough period of time over the range of territory inhabited by the organisms.

3. If BCF values from field exposures are consistently lower or higher than those from laboratory exposures, then only those values from field exposures should be used if possible.

4. A BCF should be calculated based on the concentration of the substance and its metabolites, which are structurally similar and are not much more soluble in water than the parent compound, in appropriate tissue and should be corrected for the concentration in the organisms at the beginning of the test.

5. A BCF value obtained from a laboratory or field exposure that caused an observable adverse effect on the test organism may be used only if it is similar to that obtained with unaffected organisms at lower concentrations in the same test.

6. Whenever a BCF is determined for a lipid-soluble substance, the percent lipids should also be determined in the tissue for which the BCF was calculated.

C. A BCF calculated using dry tissue weights must be converted to a wet tissue weight basis by multiplying the dry weight BCF value by 0.1 for plankton and by 0.2 for individual species of fishes and invertebrates.

D. If enough pertinent data exist, several residue values can be calculated by dividing maximum permissible tissue concentrations by appropriate BCF values.

1. For each available maximum acceptable dietary intake derived from a chronic feeding study with wildlife, including birds and aquatic organisms, the appropriate BCF is based on the whole body of aquatic species that constitute or represent a major portion of the diet of the tested wildlife species.

2. For an FDA action level, 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 living 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.

E. For lipid-soluble substances, it may be possible to calculate additional residue values. Because steady-state BCF values for a lipid-soluble chemical seem to be proportional to percent lipids from one tissue to another and from one species to another, extrapolations can be made from tested tissues or species to untested tissues or species on the basis of percent lipids.

1. For each BCF where the percent lipids is known for the same tissue which the BCF was measured, the BCF should be normalized to a 1 percent lipid basis by dividing the BCF by the percent lipids. This adjustment to a 1 percent lipid basis makes all the measured BCF values comparable, regardless of the species or tissue for which the BCF is measured.

2. Calculate the geometric mean normalized BCF. Data for both saltwater and freshwater species can be used to determine the mean normalized BCF, because the normalized BCF seems to be about the same for both kinds of organisms.

3. Residue values can then be calculated by dividing the maximum permissible tissue concentrations by the mean normalized BCF and by a percent lipids value appropriate to the maximum permissible tissue concentration.

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

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 15 for freshwater criteria and 16 for saltwater criteria because FDA action levels are applied species-by-species to commonly consumed species. The edible portion of the freshwater lake trout averages about 15 percent lipids, and the edible portion

of the saltwater Atlantic herring averages about 16 percent lipids (Sidwell, V.D., et al. 1974 Composition of the Edible Portion of Raw (Fresh or Frozen) Crustaceans, Finfish, and Mollusks. I. Protein, Fat, Moisture, Ash, Carbohydrate, Energy Value, and Cholesterol, Marine Fisheries Review 36:21-35).

c. For a maximum acceptable dietary intake derived from a chronic feeding study with wildlife, the appropriate percent lipids is the percent lipids of an aquatic species or group of aquatic species that constitutes a major portion of the diet of the wildlife species.

F. The Final Residue Value is obtained by selecting the lowest of the available residue values. In many cases the Final Residue Value will not be low enough. For example, a residue value calculated from an FDA action level would result in an average concentration in the edible portion of a fatty species that is at the action level. On the average half, of the individuals of the species would have concentrations above the FDA action level. Also, the results of many chronic feeding studies are concentrations that cause adverse effects.

X. Other Data

Pertinent information that could not be used in earlier sections may be available concerning adverse effects on aquatic organisms and their uses. The most important of these are data on flavor impairment, reduction in survival, growth, or reproduction, or any other adverse effect that has been shown to be biologically significant. Especially important are data for species for which no other data are available. Data from behavioral, microcosm, field, and physiological studies may also be available.

X. Criterion

A. The criterion consists of two concentrations, one that should not be exceeded on the average in a 24-hour period and one that should not be exceeded at any time during the 24-hour period. This two-number criterion is intended to identify water quality conditions that should protect aquatic life and its uses from acute and chronic adverse effects of both cumulative and noncumulative substances without being as restrictive as a one-number criterion would have to be to provide the same degree of protection.

B. The maximum concentration is the Final Acute Value or is obtained from the Final Acute Equation.

C. The 24-hour average concentration is obtained from the Final Chronic Value, the Final Plant Value, and the Final Residue Value by selecting the lowest available value, unless other data (see section X) from tests in which the toxicant concentrations were measured show that a lower value should be used. If toxicity is related to a water quality characteristic, the 24-hour average concentration is obtained from the Final Chronic Equation, the Final Plant Value, and the Final Residue Value by selecting the one that results in the lowest concentrations in the normal range of the water quality characteristic, unless other data (see section X) from tests in which the toxicant concentrations were measured show that a lower value should be used.

D. The criterion is (the 24-hour average concentration) as a 24-hour average and the concentration should not exceed (the maximum concentration) at any time.

XII. Review

A. 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 the Guidelines.

APPENDIX C

Derivation of the 1980 Human Health Criteria

(Excerpted from 45 F.R. 79347, November 28, 1980)

Guidelines and Methodology Used in the Preparation of Health Effect Assessment Chapters of the Consent Decree Water Criteria Documents

I. Objective

The objective of the health effect assessment chapters of the ambient water criteria documents is to estimate ambient water concentrations that do not represent a significant risk to the public. These assessments should constitute a review of all relevant information on individual chemicals or chemical classes in order to derive criteria that represent, in the case of suspect or proven carcinogens, various levels of incremental cancer risk, or, in the case of other pollutants, estimates of no-effect levels.

Ideally, ambient water quality criteria should represent levels for compounds in ambient water that do not pose a hazard to the human population. However, in any realistic assessment of human health hazard, a fundamental distinction must be made between absolute safety and recognizing some risk. Criteria for absolute safety would have to be based on detailed knowledge of dose-response relationships in humans, including all sources of chemical exposure, the types of toxic effects elicited, the existence of thresholds for the toxic effects, the significance of toxicant interactions, and the variances of sensitivities and exposure levels within the human population. In practice, such absolute criteria cannot be established because of deficiencies in both the available data and the means of interpreting this information. Consequently, the individual human health effects chapters propose criteria that minimize or specify the potential risk of adverse human effects due to substances in ambient water. Potential social or economic costs and benefits are not considered in the formulation of the criteria.

II. Types of Criteria

Ambient water quality criteria are based on three types of biological endpoints: carcinogenicity, toxicity (i.e., all adverse effects other than cancer), and organoleptic effects.

For the purpose of deriving ambient water quality criteria, carcinogenicity is regarded as a non-threshold phenomenon. Using this assumption, "safe" or "no effect" levels for carcinogens cannot be established because even extremely small doses must be assumed to elicit a finite increase in the incidence of the response. Consequently, water quality criteria for carcinogens are presented as a range of pollutant concentrations associated with corresponding incremental risks.

For compounds that do not manifest any apparent carcinogenic effect, the threshold assumption is used in deriving a criterion. This assumption is based on the premise that a physiological reserve capacity exists within the organism that is thought to be depleted before clinical disease ensues. Alternatively, the rate of damage will likely be insignificant over the life span of the organism. Thus, ambient water quality criteria are derived for non-carcinogenic chemicals, and presumably result in no observable-adverse-affect levels (NOAELs) in the exposed human population.

In some instances, criteria are based on organoleptic characteristics (i.e., thresholds for taste or odor). Such criteria are established when insufficient information is available on

toxicologic effects or when the estimate of the level of the pollutant in ambient water based on organoleptic effects is lower than the level calculated from toxicologic data. Criteria based solely on organoleptic effects do not necessarily represent approximations of acceptable risk levels for human health.

Several ambient water quality criteria documents deal with classes of compounds that include chemicals exhibiting varying degrees of structural similarity. Because prediction of biological effects based solely on structural parameters is difficult, the derivation of compound-specific criteria is preferable to a class criterion. A compound-specific criterion is defined as a level derived from data on each individual subject compound that does not represent a significant risk to the public. For some chemical classes, however, a compound-specific criterion cannot be derived for each member of a class. In such instances, it is sometimes justifiable to derive a class criterion in which available data on one member of a class may be used to estimate criteria for other chemicals of the class because a sufficient data base is not available for those compounds.

For some chemicals and chemical classes, the data base was judged to be insufficient for the derivation of a criterion. In those cases, deficiencies in the available information are detailed.

III. Approach

The human health effects chapters attempt to summarize all information on the individual chemicals or classes of chemicals that might be useful in the risk assessment process to develop water quality criteria. Although primary emphasis is placed on identifying epidemiologic and toxicologic data, these assessments typically contain discussions on four topics: existing levels of human exposure, pharmacokinetics, toxic effects, and criterion formulation.

For all documents, an attempt is made to include the known relevant information. Review articles and reports are often used in the process of data evaluation and synthesis. Scientific judgment is exercised in the review and evaluation of the data in each document and in the identification of the adverse effects against which protective criteria are sought. In addition, each of these documents is reviewed by a peer committee of scientists familiar with the specific compound(s). These work groups evaluate the quality of the available data, the completeness of the data summary, and the validity of the derived criterion.

In the analysis and organization of the data, an attempt is made to be consistent in the format and application of acceptable scientific principles. Evaluation procedures used in the hazard assessment process follow the principles outlined by the National Academy of Sciences in *Drinking Water and Health* (1977) and the guidelines of the Carcinogen Assessment Group of the U.S. EPA.

A. Exposure

The exposure section of the health effects chapters reviews known information on current levels of human exposure to the individual pollutant from all sources. Much of the data was obtained from monitoring studies of air, water, food, soil, and human or animal tissue residues. The major purpose of this section is to provide background information on the contribution of water exposure relative to all other sources. Consequently, the exposure section includes subsections reviewing different routes of exposure, including water and food ingestion, inhalation, and dermal contact.

Information on exposure can be valuable in developing and assessing a water quality criterion. In these documents, exposure from consumption of contaminated water and contaminated fish and shellfish products is used in criterion formulation. Data for all modes of exposure are useful in relating total intake to the expected contribution from contaminated water, fish, and shellfish. In addition, information for all routes of exposure, not limited to drinking water and fish and shellfish ingestion, can be used to justify or assess the feasibility of the formulation of criteria for ambient water.

The use of fish consumption as an exposure factor requires the quantitation of pollutant residues in the edible portions of the ingested species. Accordingly, bioconcentration factors (BCFs) are used to relate pollutant residues in aquatic organisms to the pollutant concentration in the ambient waters in which they reside.

To estimate the average per capita intake of a pollutant due to consumption of contaminated fish and shellfish, the results of a diet survey were analyzed to calculate the average consumption of freshwater and estuarine fish and shellfish. A species is considered to be a consumed freshwater or estuarine fish and shellfish species if at some stage in its life cycle it is harvested from fresh or estuarine water for human consumption in significant quantities.

Three different procedures are used to estimate the weighted average BCF, depending upon the lipid solubility of the chemical and the availability of bioconcentration data.

For lipid-soluble compounds, the average BCF is calculated from the weighted average percent lipids in the edible portions of consumed freshwater and estuarine fish and shellfish, which was calculated from data on consumption of each species and its corresponding percent lipids to be 3 percent. Because the steady-state BCFs for lipid-soluble compounds are proportional to percent lipids, bioconcentration factors for fish and shellfish can be adjusted to the average percent lipids for aquatic organisms consumed by Americans. For many lipid-soluble pollutants, there exists at least one BCF for which the percent lipid value was measured for the tissues for which the BCF is determined.

With 3 percent as the weighted average percent lipids for freshwater and estuarine fish and shellfish in the average diets, a BCF, and a corresponding percent lipid value, the weighted average bioconcentration factor can be calculated.

Example:

Weighted average percent lipids for average diet = 3.0 percent
Measured BCF of 17 for trichloroethylene with bluegills at 4.8 percent lipids
Weighted average BCF for average diet equals

$$17 \times \frac{3.0\%}{4.8\%} = 10.6$$

As an estimate, 10.6 is used for the BCF.

In those cases where an appropriate bioconcentration factor is not available, the equation "Log BCF = (0.85 Log P)- 0.70" can be used to estimate the BCF for aquatic organisms containing about 7.6 percent lipids from the octanol/water partition coefficient P. An adjustment for percent lipids in the average diet versus 7.6 percent is made in order to derive the weighted average bioconcentration factor.

For non-lipid-soluble compounds, the available BCFs for the edible portion of consumed freshwater and estuarine fish and shellfish are weighted according to consumption factors to determine a weighted BCF representative of the average diet.

B. Pharmacokinetics

This section summarizes the available information on the absorption, distribution, metabolism, and elimination of the compound(s) in humans and experimental mammals. Conceptually, such information is useful in validation of inter- and intraspecies extrapolations, and in characterizing the modes of toxic actions. Sufficient information on the absorption and excretion in animals — together with a knowledge of ambient concentrations in water, food, and air — could be useful in estimating body burdens of chemicals in the human population. Distribution data that suggest target organs or tissues are desirable for interspecies comparison techniques. For derivation of criteria, pharmacokinetic data are essential to estimate equivalent oral doses based on data from inhalation or other routes of exposure.

C. Effects

This section summarizes information on biological effects in both humans and experimental mammals resulting in acute, subacute, and chronic toxicity; synergism and/or antagonism; teratogenicity, mutagenicity, or carcinogenicity.

The major goal of this section is to survey the suitability of the data for use in assessment of hazard and to determine which biological end-point (i.e., non-threshold, threshold, or organoleptic) should be selected for use in criterion formulation.

Because this section attempts to assess potential human health effects, data on documented human effects are thoroughly evaluated. However, several factors inherent in human epidemiological studies usually preclude the use of such data in generating water quality criteria. These problems, as summarized by the National Academy of Sciences, are as follows:

1. Epidemiology cannot tell what effects a material will have until after humans have been exposed. One must not conduct what might be hazardous experiments on man.
2. If exposure has been ubiquitous, assessing the effects of a material may be impossible because no control group is unexposed. Statistics of morbidity obtained before using a new material can sometimes be useful; but when latent periods are variable and times of introduction and removal of materials overlap, historical data on chronic effects are usually unsatisfactory.
3. Determining doses in human exposure is usually difficult.
4. Usually, it is hard to identify small changes in common effects, which may nonetheless be important if the population is large.
5. Interactions in a "nature-designed" experiment usually cannot be controlled.

Although these problems often prevent the use of epidemiological data in quantitative risk assessments, qualitative similarities or differences between documented effects in humans and observed effects in experimental mammals are extremely useful in testing the validity of animal-to-man extrapolations. Consequently, in each case, an attempt is made to identify and utilize both epidemiological and animal dose-response data. Criteria derived from such a confirmed data base are considered to be reliable.

The decision to establish a criterion based on a non-threshold model is made after evaluating all available information on carcinogenicity and supportive information on mutagenicity. The approach and conditions for the qualitative decision of carcinogenicity are outlined in the U.S. EPA Interim Cancer Guidelines (41 FR 21402), in a report by Albert, et al. (1977), and in the Interagency Regulatory Liaison Group (IRLG) guidelines on carcinogenic risks (IRLG, 1979). A substance that induces a statistically significant carcinogenic response in animals is assumed to have the capacity to cause cancer in humans. A chemical that has not induced a significant cancer response in humans or experimental animals is not identified as a carcinogen, even though its metabolites or close structural analogues might induce a carcinogenic response or it was shown to be mutagenic in an *in vitro* system.

Some potential human carcinogens may not be identified by the guidelines given above. For example, compounds that have plausible but weak qualitative evidence of carcinogenicity in experimental animal systems (such as data from mouse skin painting or strain A mouse pulmonary adenoma) would be included in this category. The derivation of a criterion for human consumption from these studies in not valid, regardless of the qualitative outcome. In addition, certain compounds (e.g., nickel and beryllium) shown to be carcinogenic in humans after inhalation exposure by chemical form have induced, thus far, no response in animals or humans via ingesting their soluble salts.

Nevertheless, nonthreshold criterion is developed for beryllium because tumors have been produced in animals at a site removed from the site of administration. In contrast, a threshold criterion is recommended for nickel because no evidence exists of tumors at distant sites resulting from administration of nickel solutions by either ingestion or injection.

For those compounds not reported to induce carcinogenic effects or for which carcinogenic data are lacking or insufficient, an attempt is made to estimate a no-effect level. In many respects, the hazard evaluation from these studies is similar to that of bioassays for carcinogenicity. In order to more closely approximate conditions of human exposure, preference is given to chronic studies involving oral exposures in water or diet over a significant portion of the animal life span. Greatest confidence is placed in those studies that demonstrate dose-related adverse effects as well as no-effect levels.

The biological endpoints used to define a no-effect level vary considerably. They may range from gross effects — such as mortality — to more subtle biochemical, physiological, or pathological changes. Teratogenicity, reproductive impairment, and behavioral effects are significant toxic consequences of environmental contamination. In instances where carcinogenic or other chronic effects occur at exposure levels below those causing teratogenicity, reproductive impairment, or behavioral effects, the former are used in deriving the criterion. For most of the compounds evaluated thus far, teratogenicity and reproductive impairment occur at doses near maximum tolerated levels with dose administration schedules well above estimated environmental exposure levels. Moreover, information on behavioral effects, which could be of significance, is not available for most of the compounds under study. Consequently, most NOAELs derived from chronic studies are based either on gross toxic effects or on effects directly related to functional impairment or defined pathological lesions.

For compounds on which adequate chronic toxicity studies are not available, studies on acute and subacute toxicity assume greater significance. Acute toxicity studies usually involve single exposures at lethal or near lethal doses. Subacute studies often involve exposure exceeding 10 percent of the life span of the test organism, (e.g., 90 days for the rat with an average life span of 30 months). Such studies are useful in establishing the nature of the compound's toxic effects and other parameters of compound toxicity, such as target organ effects, metabolic behavior, physiological/biochemical effects, and patterns of retention and tissue distribution. The utility of acute and subacute studies in deriving environmentally meaningful NOELs is uncertain, although McNamara (1976) has developed application factors for such derivations.

In some cases where adequate data are not available from studies utilizing oral routes of administration, no-effect levels for oral exposures may be estimated from dermal or inhalation studies. Such estimates involve approximations of the total dose administered based on assumptions about breathing rates and/or magnitude of absorption.

D. Criterion Rationale

This section reviews existing standards for the chemical(s), summarizes data on current levels of human exposure, attempts to identify special groups at risk, and defines the basis for the recommended criterion.

Information on existing standards is included primarily for comparison with the proposed water quality criteria. Some of the present standards, such as those recommended by the Occupational Safety and Health Administration (OSHA) or the American Conference of Governmental Industrial Hygienists (ACGIH), are based on toxicologic data but are intended as acceptable levels for occupational rather than environmental exposure. Other levels, such as those recommended by the National Academy of Sciences in *Drinking Water and Health* (1977) or in the U.S. EPA Interim Primary Drinking Water Standards, are more closely related to proposed water quality criteria. Emphasis is placed on detailing the basis for the existing standards wherever possible.

Summaries of current levels of human exposure, presented in this section, specifically address the suitability of the data to derive water quality criteria. The identification of special groups at risk, either because of geographical or occupational differences in exposure or biological differences in susceptibility to the compound(s), focuses on the impact that these groups should have on the development of water quality criteria.

The basis for the recommended criteria section summarizes and qualifies all of the data used in developing the criteria.

IV. Guidelines for Criteria Derivation

The derivation of water quality criteria from laboratory animal toxicity data is essentially a two-step procedure. First, a total daily intake for humans must be estimated that establishes either a defined level of risk for nonthreshold effects or a noeffect level for threshold effects. Secondly, assumptions must be made about the contribution of contaminated water and the

consumption of fish/shellfish to the total daily intake of the chemical. These estimates are then used to establish the tolerable daily intake and consequently the water quality criterion.

A. Nonthreshold Effects

After the decision has been made that a compound can potentially cause cancers in humans and that data exist that permit the derivation of a criterion, the water concentration estimated to cause a lifetime carcinogenic risk of 10^{-5} is determined. The lifetime carcinogenicity risk is the probability that a person would get cancer sometime in his or her life assuming continuous exposure to the compound. The water concentration is calculated by using the low-dose extrapolation procedure proposed by Crump (1980). This procedure is an improvement on the multistage low-dose extrapolation procedure by Crump, et al. (1977).

The data used for quantitative estimates are of two types: (1) lifetime animal studies, and (2) human studies where excess cancer risk has been associated with exposure to the agent. In animal studies it is assumed, unless evidence exists to the contrary, that if a carcinogenic response occurs at the dose levels used in the study, then proportionately lower responses will also occur at all lower doses, with an incidence determined by the extrapolation model discussed below.

1. Choice of Model.

No really solid scientific basis exists for any mathematical extrapolation model that relates carcinogen exposure to cancer risks at the extremely low levels of concentration present to deal with in evaluating the environmental hazards. For practical reasons, such low levels of risk cannot be measured directly either using animal experiments or epidemiologic studies. We must, therefore, depend on our current understanding of the mechanisms of carcinogenesis for guidance as to which risk model to use.

At the present time, the dominant view of the carcinogenic process involves the concept that most agents which cause cancer also cause irreversible damage to DNA. This position is reflected by the fact that a very large proportion of agents that cause cancer are also mutagenic. The quantal type of biological response characteristic of mutagenesis is likely associated with a linear nonthreshold dose-response relationship. Indeed, substantial evidence from mutagenesis studies with both ionizing radiation and a wide variety of chemicals suggests that this type of dose-response model is the appropriate one to use. This is particularly true at the lower end of the dose-response curve; at higher doses, there can be an upward curvature, probably reflecting the effects of multistage processes on the mutagenic response.

The linear nonthreshold dose-response relationship is also consistent with the relatively few epidemiological studies of cancer responses to specific agents that contain enough information to make the evaluation possible (e.g., radiation-induced leukemia, breast and thyroid cancer, skin cancer induced by arsenic in drinking water, and liver cancer induced by aflatoxin in the diet). Some animal experiments are consistent with the linear nonthreshold hypothesis (e.g., liver tumors induced in mice by 2-acetylaminofluorene in the large scale ED_{01} study at the National Center of Toxicological Research, and the initiation stage of the two-stage carcinogenesis model in the rat liver and the mouse skin).

Because it has the best, albeit limited, scientific basis of any of the current mathematical extrapolation models, the linear nonthreshold model has been adopted as the primary basis for risk extrapolation to low levels of the dose-response relationship. The risk assessments made with this model should be regarded as conservative, representing the most plausible upper limit for the risk (i.e., the true risk is not likely to be higher than the estimate, but it could be smaller).

The mathematical formulation chosen to describe the linear, nonthreshold dose-response relationship at low doses is the improved multistage model developed by Crump (1980). This model employs enough arbitrary constants to be

able to fit almost any monotonically increasing dose-response data, and it incorporates a procedure for estimating the largest possible linear slope (in the 95 percent confidence limit sense) at low extrapolated doses consistent with the data at all dose levels of the experiment. For this reason, it may be called a "linearized" multistage model.

2. Procedure of Low-Dose Extrapolation Based on Animal Carcinogenicity Data.

A. Description of the Extrapolation Model

Let P(d) represent the lifetime risk (probability) of cancer at dose d. The multistage model has the form

$$P(d) = 1 - \exp[-(q_0 + q_1 d + q_2 d^3 + \ldots + q_k d^k)]$$

where:

$q_i > 0$, and $i = 0, 1, 2, \ldots k$

Equivalently,

$$A(d) = 1 - \exp[-(q_1 d + q_2 d^2 + \ldots + q_k d^k)]$$

where:

$$A(d) = \frac{P(d) - P(o)}{1 - P(o)}$$ is the extra risk over background rate at dose d.

The point estimate of the coefficients q_i i = 0, 1, 2,...,k, and consequently the extra risk function A(d) at any given dose d, is calculated by maximizing the likelihood function of the data.

The point estimate and the 95 percent upper confidence limit of the extra risk A(d) are calculated by using the computer program GLOBAL 79 developed by Crump and Watson (1979). Upper 95 percent confidence limits on the extra risk and lower 95 percent confidence limits on the dose producing a given risk are determined from a 95 percent upper confidence limit q_1^*, on parameter q_1. Whenever $q_1 \neq 0$, at low doses extra risk A(d) has approximately the form $A(d) = q_2 \times d$. Therefore, $q_2 \times d$ is a 95 percent upper confidence limit on the extra risk and R/q_1^* is a 95 percent lower confidence limit on the dose producing an extra risk of R. Let L_0 be the maximum value of the log-likelihood function. The upper limit q_1^* is calculated buy increasing q_1 to a value q_1^* such that when the log-likelihood is again maximized subject to this fixed value q_1^* for the linear coefficient, the resulting maximum value of the log-likelihood L_1 satisfies the equation $2(L_0 - L_1) = 2.70554$ where 2.70554 is the cumulative 90 percent point of the chi-square distribution with one degree of freedom, which corresponds to a 95 percent upper limit (one-sided). This approach of computing the upper confidence limit for the extra risk A(d) is an improvement on the Crump, et al. (1977) model. The upper confidence limit for the extra risk calculated at low doses is always linear. This is conceptually consistent with the linear nonthreshold concept discussed earlier. The slope q_1^* is taken as an upper bound of the potency of the chemical in inducing cancer at low doses.

In fitting the dose-response model, the number of terms in the polynomial g is chosen equal to (h-1), where h is the number of dose groups in the experiment, including the control group.

Whenever the multistage model does not fit the data sufficiently, data at the highest dose is deleted and the model is refitted to the rest of the data. This is continued until an acceptable fit to the data is obtained. To determine whether or not a fit is acceptable, the chi-square statistic:

$$X^2 = \sum_{i=1}^{h} \frac{(X_i - N_i P_i)^2}{N_i P_i (1 - P_i)}$$

is calculated, where N_1 is the number of animals in the i^{th} dose group, X_i is the number of animals in the i^{th} dose group with a tumor response, P_1 is the probability

of a response in the i^{th} dose group estimated by fitting the multistage model to the data, and h is the number of remaining groups.

The fit is determined to be unacceptable whenever chi-square (X^2) is larger than the cumulative 99 percent point of the chi-square distribution with f degrees of freedom, where f equals the number of dose groups minus the number of non-zero multistage coefficients.

3. Selection and Form of Data used to Estimate Parameters in the Extrapolation Model.

For some chemicals, several studies in different animal species, strains, and sexes each conducted at several doses and different routes of exposure are available. A choice must be made as to which of the data sets from several studies are to be used in the model. It is also necessary to correct for metabolism differences between species and for differences in absorption via different routes of administration. The procedures used in evaluating these data, listed below, are consistent with the estimate of a maximum-likely-risk.

a. The tumor incidence data are separated according to organ sites or tumor types. The set data (i.e., dose and tumor incidence) used in the model is set where the incidence is statistically significantly higher than the control for at least one test dose level and/or where the tumor incidence rate shows a statistically significant trend with respect to dose level. The data set that gives the highest estimate of lifetime carcinogenic risk q_1^* is selected in most cases. However, efforts are made to exclude data sets that produce spuriously high risk estimates because of a small number of animals. That is, if two sets of data show a similar dose-response relationship and one has a very small sample size, the set of data that has the larger sample size is selected for calculating the carcinogenic potency.

b. If there are two more data sets of comparable size that are identical with respect to species, strain, sex, and tumor sites, the geometric mean of q_1^*, estimated from each of these data sets, is used for risk assessment. The geometric mean of numbers $A_1, A_2, ..., A_m$ is defined as $(A_1 \times A_2 \times ... A_m)^{1/m}$

c. If sufficient data exist for two or more significant tumor sites in the same study, the number of animals with at least one of the specific tumor sites under consideration is used as incidence data in the model.

d. Following the suggestion of Mantel and Schneiderman (1975), we assume that mg/surface area/day is an equivalent dose between species. Since to a close approximation the surface area is proportional to the ⅔rds power of the weight as would be the case for a perfect sphere, the exposure in mg/⅔rds power of the body weight/day is similarly considered to be an equivalent exposure. In an animal experiment, this equivalent dose is computed in the following manner:

Let:

L_e = duration of experiment

l_e = duration of exposure

m = average dose per day in mg during administration of the agent (i.e., during l_e)

W = average weight of the experimental animal.

Then, the lifetime average exposure is $d = \frac{l_e \times m}{L_e \times W^{2/3}}$

Often exposures are not given in units of mg/day, and it becomes necessary to convert the given exposures to mg/day. For example, in most feeding studies, exposure is expressed as ppm in the diet. In this case the exposure (mg/day) is derived by m = ppm $\times$ F $\times$ r, where ppm is parts per million of the carcinogenic agent in the diet, F is the weight of the food consumed per day in kgms, and r is the absorption fraction.

In the absence of any data to the contrary, r is assumed to be one. For a uniform diet, the weight of the food consumed is proportional to the calories required, which, in turn, is proportional to the surface area or the ⅔rds power of the weight, so that $m_a$ppm $x\ W^{2/3}\ x\ r$ or

$$\frac{m}{r\ W^{2/3}}\ a^{ppm}$$

As a result, ppm in the diet is often assumed to be an equivalent exposure between species. However, this is not justified since the calories/kg of food is significantly different in the diet of man vs. laboratory animals, primarily due to moisture content differences. Instead, an empirically derived food factor, f=F/W, is the fraction of a species body weight that is consumed per day as food. The rates are

Species	W	f
Man	70	0.028
Rat	0.35	0.06
Mice	0.03	0.13

Thus, when the exposure is given as a certain dietary concentration in ppm, the exposure in $mg/W^{2/3}$ is

$$\frac{m}{r\ x\ W^{2/3}} = \frac{ppm\ x\ F}{W^{2/3}} =$$

$$\frac{ppm\ x\ fxW}{W^{2/3}} = ppm\ x\ f\ x\ W^{1/3} =$$

When exposure is given in terms of mg/kg/day = m/Wr = s the conversion is simply:

$$\frac{m}{r\ W^{2/3}} = s\ x\ W^{1/3}$$

When exposure is via inhalation, the calculation of dose can be considered for two cases where (1) the carcinogenic agent is either a completely water-soluble gas or an aerosol and is absorbed proportionally to the amount of air breathed in, and (2) where the carcinogen is a poorly water-soluble gas that reaches an equilibrium between the air breathed and the body compartments. After equilibrium is reached, the rate of absorption of these agents is expected to be proportional to metabolic rate, which in turn is proportional to the rate of oxygen consumption, which in turn is a function of surface area.

B. Threshold Effects

1. Use of Animal Toxicity Data (oral).

In developing guidelines for deriving criteria based on noncarcinogenic responses, five types of response levels are considered:

NOEL—No Observed-Effect-Level

NOAEL—No Observed-Adverse-Effect-Level

LOEL—Lowest-Observed-Effect-Level

LOAEL—Lowest-Observed-Adverse-Effect-Level

FEL—Frank-Effect-Level

Adverse effects are defined as any effects that result in functional impairment and/or pathological lesions which may affect the performance of the whole organism or reduce an organism's ability to respond to an additional challenge.

One of the major problems encountered in considering these concepts regards the reporting of "observed effect levels" as contrasted to "observed adverse effect

levels." The terms "adverse" versus "not adverse" are at times satisfactorily defined. But due to increasingly sophisticated testing protocols, more subtle responses are being identified, resulting in a need for judgment regarding the exact definition of adversity.

The concepts listed above (NOEL, NOAEL, LOEL, LOAEL) have received much attention because they represent landmarks that help define the threshold region in specific experiments. Thus, if a single experiment yields a NOEL, a NOAEL, a LOAEL, and a clearly defined FEL in closely spaced doses, the threshold region has been well defined; such data are very useful for the purpose of deriving a criterion. On the other hand, a clearly defined FEL has little utility in establishing criteria when it stands alone, because such a level gives no indication how far removed the data point is from the threshold region. Similarly, a free-standing NOEL has little utility without an indication of its proximity to the LOEL, since a free-standing NOEL may be many orders of magnitude below the threshold region.

Based on the above dose-response classification system, the following guidelines for deriving criteria have been adopted:

a. A free-standing FEL is unsuitable for the derivation of criteria.
b. A free-standing NOEL is unsuitable for the derivation of criteria. If multiple NOELs are available without additional data on LOELs, NOAELs, or LOAELs, the highest NOEL should be used to derive a criterion.
c. A NOAEL, LOEL, OR LOAEL can be suitable for criteria derivation. A well-defined NOAEL from a chronic (at least 90-day study) may be used directly, applying the appropriate uncertainty factor. For a LOEL, a judgment needs to be made as to whether it actually corresponds to a NOAEL or a LOAEL. In the case of a LOAEL, an additional uncertainty factor is applied; the magnitude of the additional uncertainty factor is judgmental and should lie in the range of 1 to 10. Caution must be exercised not to substitute "Frank-Effect-Levels" for "Lowest-Observable-Adverse-Effect-Levels."
d. If for reasonably closely spaced doses only a NOEL and a LOAEL of equal quality are available, then the appropriate uncertainty factor is applied to the NOEL.

In using this approach, the selection and justification of uncertainty factors are critical. The basis definition and guidelines for using uncertainty factors has been given by the National Academy of Sciences (1977). "Safety Factor" or "Uncertainty Factor" is defined as a number that reflects the degree or amount of uncertainty that must be considered when experimental data in animals are extrapolated to man. When the quality and quantity of experimental data are satisfactory, a low uncertainty factor is used: when data is judged to be inadequate or equivocal, a larger uncertainty factor is used. The following general guidelines have been adopted in establishing the uncertainty factors:

a. Valid experiment results from studies on prolonged ingestion by man, with no indication of carcinogenicity. Uncertainty Factor = 10
b. Experimental results of studies of human ingestion not available or scanty (e.g., acute exposure only) with valid results of long-term feeding studies on experimental animals, or in the absence of human studies, valid animal studies on one or more species. No indication of carcinogenicity. Uncertainty Factor = 100
c. No long-term or acute human data. Scanty results on experimental animals with no indication of carcinogenicity. Uncertainty Factor = 1,000

Considerable judgment must be used in selecting the appropriate safety factors for deriving a criterion. In those cases where the data do not completely fulfill the conditions for one category and appear to be intermediate between two categories, an intermediate uncertainty factor is used. Such as intermediate uncertainty factor may be developed based on a logarithmic scale (e.g., 33 being halfway between 10 and 100 on a logarithmic scale).

In determining the appropriate use of the uncertainty factors, the phrase "no indication of carcinogenicity" is interpreted as the absence of carcinogenicity data from animal experimental studies or human epidemiology. Available short-term carcinogenicity screening tests are reported in the criteria documents, but they are used neither for derivation of numerical criteria nor to rule out the uncertainty factor approach.

Because of the high degree of judgment involved in the selection of a safety factor, the criterion derivation section of each document should provide a detailed discussion and justification for both the selection of the safety factor and the data to which it is applied. This discussion should reflect a critical review of the available data base. Factors to be considered include number of animals, species, and parameters tested; quality of controls; dose levels; route; and dosing schedules. An effort should be made to differentiate between results that constitute a toxicologically sufficient data base and data which may be spurious in nature.

2. Use of Acceptable Daily Intake (ADI).

For carcinogens, the assumption of low-dose linearity precludes the necessity for defining total exposure in the estimation of increased incremental risk. For non-carcinogens, ADIs and criteria derived therefrom are calculated from total exposure data that include contributions from the diet and air. The equation used to derive the criterion (C) is:

$$C = ADI - (DT+IN) \;/\; [2l + (0.0065\ kg \times R)]$$

where

2l =	assumed daily water consumption
0.00065 kg =	assumed daily fish consumption
R =	bioconcentration factor in units of l/kg
DT =	estimated non-fish dietary intake
IN =	estimated daily intake by inhalation.

If estimates of IN and DT cannot be provided from experimental data, an assumption must be made concerning total exposure. The inability to estimate DT and IN due to either lack of data or the wide variability in DT and IN in different states may add an additional element of uncertainty to the criterion formulation process. To achieve scientific validity, the accurate estimate of the Acceptable Daily Intake is the major factor in satisfactory derivation of water quality criteria.

3. Use of Threshold Limit Values or Animal Inhalation Studies.

Threshold Limit Values (TLVs) are established by the American Conference of Governmental and Industrial Hygienists (ACGIH) and represent 8-hour time-weighted average concentrations in air that are intended to protect workers from various adverse health effects over a normal working lifetime. Similar values are set by NIOSH (criteria) and OSHA (standards) for 10- and 8-hour exposures, respectively. To the extent that these values are based on sound toxicologic assessments and have been protective in the work environment, they provide useful information for deriving or evaluating water quality criteria. However, each TLV must be carefully examined to determine if the basis of the TLV contains data that can be used directly to derive a water quality criterion using the uncertainty factor approach. In addition, the history of each TLV must be examined to assess the extent to which it has assured worker safety. In each case, the types of effects against which TLVs are designed to protect are examined relative to exposure from water. It must be demonstrated that the chemical is not a localized irritant and that the effect at the site of entry is not significant irrespective of the routes of exposure (i.e., oral or inhalation).

If the TLV or similar value is recommended as the basis of the criterion, consideration of the previous points is explicitly stated in the document's criterion

derivation section. Particular emphasis is placed on the quality of the TLV relative to the available toxicity data that normally is given priority over TLVs or similar established values. If the TLV can be justified as the basis for the criterion, then the problems associated with estimating acceptable oral doses from inhalation data must be addressed.

Estimating equivalencies of dose-response relationships from one route of exposure to another introduces an additional element of uncertainty in the criteria derivation. Consequently, whenever possible, ambient water quality criteria should be based on data involving oral exposures. If oral data are insufficient, data from other routes of exposure may be useful in the criterion derivation process.

Inhalation data, including TLVs or similar values, are the most common alternatives to oral data. Estimates of equivalent doses can be based upon (1) available pharmacokinetic data for oral and inhalation routes, (2) measurements of absorption efficiency from ingested or inhaled chemicals, or (3) comparative excretion data when the associated metabolic pathways are equivalent to those following oral ingestion or inhalation. Given that sufficient pharmacokinetic data are available, the use of accepted pharmacokinetic models provides the most satisfactory approach for dose conversions. However, if available pharmacokinetic data are marginal or of questionable quality, pharmacokinetic modeling is inappropriate.

The Stokinger and Woodward (1958) approach, or similar models based on assumptions of breathing rate and absorption efficiency, represents possible alternatives when data are not sufficient to justify pharmacokinetic modeling. Such alternative approaches, however, provide less satisfactory approximations because they are not based on pharmacokinetic data. Consequently, in using the Stokinger and Woodward or related models, the basis and uncertainties inherent in each assumption must be clearly stated in the derivation of the criterion.

The use of data pertaining to other routes of exposure to derive water quality criteria may also be considered. As with inhalation data, an attempt is made to use accepted toxicologic and pharmacokinetic principles to estimate equivalent oral doses. If simplifying assumptions are used, their bases and limitations must be clearly specified.

Because of the uncertainties involved in extrapolating from one route of exposure to another and the consequent limitations that this may place on the derived criterion, the decision to disallow such extrapolation and recommend no criterion is highly judgmental and must be made on a case-by-case basis. A decision for or against criteria derivation must balance the quantity and quality of the available data against a perceived risk to the human population.

If the Stokinger and Woodward (1958) approach is used to calculate an ADI from a TLV, the general equation is

$$ADI = TLV \times BR \times DE \times d \times A_A / (A_o \times SF)$$

where:

ADI = Acceptable daily intake in mg
TLV = Concentration in air in mg/m^3
DE = Duration of exposure in hours per day
d = 5 days/7 days
A_A = Efficiency of absorption from air
A_o = Efficiency of absorption from oral exposure
SF = Safety factor following guidelines given above
BR = Amount of air breathed per day; assume 10 m^3

For deriving an ADI from animal toxicity data, the equation is

$$ADI = C_A \times D_E \times d \times A_A \times BR \times 70\,kg / (BW_A \times A_o \times SF)$$

where:

ADI = Acceptable daily intake in mg
C_A = Concentration in air in mg/m^3
D_E = Duration of exposure in hours per day
d = Number of days exposed/number of days observed
A_A = Efficiency of absorption from air
BR = Volume of air breathed per day in m^3
70 kg = Assumed human body weight
BW_A = Body weight of experimental animals in kg
A_o = Efficiency of absorption from oral exposure
SF = Safety factor following guidelines given above

More formal pharmacokinetic models must be developed on a compound-by-compound basis.

Safety factors used in the above formulae are intended to account for species variability. Consequently, the mg surface area/day conversion factor is not used in the derivation of toxicity based criterion.

C. Organoleptic Criteria

Organoleptic criteria define concentrations of materials that impart undesirable taste and/or odor to water. In developing and utilizing such criteria two factors must be appreciated: the limitations of most organoleptic data and the human health significance of organoleptic properties.

The publications that report taste and odor thresholds are, with few exceptions, cryptic in their descriptions of test methodologies, number of subjects tested, concentration/response relationships, and sensory characteristics at specific concentrations above threshold. Thus, the quality of organoleptic data is often significantly less than that of toxicologic data used in establishing other criteria. Consequently, a critical evaluation of the available organoleptic data must be made and selection of the most appropriate data base for the criterion based on sound scientific judgment.

Organoleptic criteria are not based on toxicologic information and have no direct relationship to potential adverse human health effects. Sufficiently intense organoleptic characteristics could result in depressed fluid intake that, in turn, might aggravate a variety of functional disease states (i.e., kidney and circulatory diseases). However, such effects are not used in the derivation process or organoleptic criteria unless available data would indicate an indirect human health effect via decreased fluid consumption; criteria derived solely from organoleptic data are based upon aesthetic qualities only.

Since organoleptic and human health effects criteria are based on different endpoints, a distinction must be made between these two sets of information. In criteria summaries involving both types of data, the following format is used:

For comparison purposes, two approaches were used to derive criterion levels for _____. Based on available toxicity data, for the protection of public health the derived level is _____. Using available organoleptic data, for controlling undesirable taste and odor quality of ambient water the estimated level is _____. It should be recognized that organoleptic data as a basis for establishing a water quality criteria have no demonstrated relationship to potential adverse human health effects.

In those instances where a level to limit toxicity cannot be derived, the following statement is to be appropriately inserted:

Sufficiently data are not available for _____ to derive a level that would protect against the potential toxicity of this compound.

D. Criteria for Chemical Classes

A chemical class is broadly defined as any group of chemical compounds that are reviewed in a single risk assessment document. In criterion derivation, isomers should be regarded as a part of a chemical class rather than as a single compound. A class criterion is an estimate of risk/safety that applies to more than one member of a class. It uses available data on one or more chemicals of a class to derive criteria for other compounds of the same class in the event that insufficient data are available to derive compound-specific criteria.

A class criterion usually applies to each member of a class rather than to the sum of the compounds within the class. While the potential hazards of multiple toxicant exposure are not to be minimized, a criterion, by definition, most often applies to an individual compound. Exceptions may be made for complex mixtures that are produced, released, and toxicologically tested as mixtures (e.g., toxaphene and PCBs). For such exceptions, some attempt is made to assess the effects of environmental partitioning (i.e., different patterns of environmental transport and degradation) on the criterion's validity. If these effects cannot be assessed, an appropriate statement of uncertainty should accompany the criterion.

Since relatively minor structural changes within a class of compounds can have pronounced effects on their biological activities, reliance on class criteria should be minimized. Whenever sufficient toxicologic data are available on a chemical within a class, a compound-specific criterion should be derived. Nonetheless, for some chemical classes, scientific judgment may suggest a sufficient degree of similarity among chemicals within a class to justify a class criterion applicable to some of all members of a class.

The development of a class criterion takes into consideration the following:

1. A detailed review of the chemical and physical properties of chemicals within the group should be made. A close relationship within the class with respect to chemical activity would suggest a similar potential to reach common biological sites within tissues. Likewise, similar lipid solubilities would suggest the possibility of comparable absorption and tissue distribution.

2. Qualitative and quantitative data for chemicals within the group are examined. Adequate toxicologic data on a number of compounds within a group provides a more reasonable basis for extrapolation to other chemicals of the same class than minimal data on one chemical or a few chemicals within the group.

3. Similarities in the nature of the toxicologic response to chemicals in the class provides additional support for the prediction that the response to other members of the class may be similar. In contrast, where the biological response has been shown to differ markedly on a qualitative and quantitative basis for chemicals within a class, the extrapolation of a criterion to other members of that class is not appropriate.

4. Additional support for the validity of extrapolation of a criterion to other members of a class could be provided by evidence of similar metabolic and pharmacokinetic data for some members of the class.

Based on the above considerations, in some cases a chemical class may be divided into various subclasses. Such divisions could be based on biological endpoints (e.g., carcinogens/non- carcinogens), potency, and/or sufficiency of data (e.g., a criterion for some members of a class but no criterion for others). While no *a priori* limits can be placed on the extent of subclassification, each subclassification must be explicitly justified by the available data.

Class criteria, if properly derived and supported, can constitute valid scientific assessments of potential risk/safety. Conversely, the development of a class criterion from an insufficient data base can lead to serious errors in underestimating or overestimating risk/safety and should be rigorously avoided. Although scientific judgment has a proper role in the development of class criteria, such criteria are useful and defensible only if based on adequate data and scientific reasoning. The definition of sufficient data on similarities in physical, chemical, pharmacokinetic, or toxicologic properties to justify a

class criterion may vary markedly, depending on the degree of structural similarity and the gravity of the perceived risk.

Consequently, the criterion derivation section of each document in which a class criterion is recommended must explicitly address each of the key issues discussed above, and define, as clearly as possible, the limitations of the proposed criterion as well as the type of data needed to generate a compound-specific criterion.

A class criterion should be abandoned when sufficient data is available to derive a compound-specific criterion that protects against the biological effect of primary concern; e.g., the availability of a good subchronic study would not necessarily result in the abandonment of a class criterion based on potential carcinogenicity.

The inability to derive a valid class criterion does not, and should not, preclude regulation of a compound or group of compounds based on concern for potential human health effects. The failure to recommend a criterion is simply a statement that the degree of concern cannot be quantified based on the available data and risk assessment methodology.

E. Essential Elements

Some chemicals, particularly certain metals, are essential to biological organisms at low levels but may be toxic and/or carcinogenic at high levels. Because of potential toxic effects, criteria should be established for such essential elements. However, criteria must consider essentiality and cannot be established at levels that would result in deficiency of the element in the human population.

Elements are accepted as essential if listed by NAS Food and Nutrition Board or a comparably qualified panel. Elements not yet determined to be essential but for which supportive data on essentiality exists need to be further reviewed by such a panel.

To modify the toxicity and carcinogenicity based criteria, essentiality must be quantified either as a "recommended daily allowance" (RDA) or "minimum daily requirement" (MDR). These levels are then compared to estimated daily doses associated with the adverse effect of primary concern. The difference between the RDA or MDR and the daily doses causing a specified risk level for carcinogens or ADIs for non-carcinogens defines the spread of daily doses from which the criterion may be derived. Because errors are inherent in defining both essential and maximum tolerable levels, the criterion is derived from dose levels near the center of such a dose range. The decision to use either the MDR or RDA is guided by the spread of the doses and the quality of the essentiality and toxicity estimates.

The modification of criteria by consideration of essentiality must take into account all routes of exposure. If water is a significant source of the MDR or RDA, the criterion must allow for attainment of essential intake. Conversely, even when essentiality may be attained from nonwater sources, standard criteria derivation methods may be adjusted if the derived criterion represents a small fraction of the ADI or MDR. On a case-by-case basis, the modification in the use of the guidelines may include the use of different safety factors for non-carcinogens or other modifications which can be explicitly justified.

F. Use of Existing Standards

For some chemicals for which criteria are to be established, drinking water standards already exist. These standards represent not only a critical assessment of literature, but also a body of human experience since their promulgation. Therefore, accepting the existing standard is valid unless compelling evidence exists to the contrary. This decision should be made after considering the existing standards versus new scientific evidence that has accumulated since the standards have been established. In several instances, the peer review process recommended usage of the present drinking water standards.

APPENDIX D

Derivation of 1976 Philosophy of Aquatic Life Criteria

Water quality criteria specify concentrations of water constituents that, if not exceeded, are expected to support an organic ecosystem suitable for the higher uses of water. Such criteria are derived from scientific facts obtained from experimental or in situ observations that depict organic responses to a defined stimulus or material under identifiable or regulated environmental conditions for a specified time period.

Water quality criteria are not intended to offer the same degree of strategy for survival and propagation at all times to all organisms within a given ecosystem. They are intended not only to protect essential and significant life in water and the direct users of water but also to protect life that is dependent on water for its existence or may consume intentionally or unintentionally any edible portion of such life.

The criteria levels for domestic water supply incorporate available data for human health protection. Such values are different from the criteria levels necessary for protection of aquatic life. The Agency's interim primary drinking water regulations (40 Federal Register 59566, December 24, 1975), as required by the Safe Drinking Water Act (42 U.S.C. 300f, et seq.), incorporate applicable domestic water supply criteria. Where pollutants are identified in both the quality criteria for domestic water supply and the Drinking Water Standards, the concentration levels are identical. Water treatment may not significantly affect the removal of certain pollutants.

What is essential and significant life in water? Do *Daphnia* or stonefly nymphs qualify as such life? Why does 1/100th of a concentration that is lethal to 50 percent of the test organisms (LC50) constitute a criterion in some instances, whereas 1/2 or 1/10th of some effect levels constitutes a criterion in other instances? These are questions often asked of those who undertake the task of criteria formulation.

The universe of organisms composing life in water is great in both kinds and numbers. As in the human population, physiological variability exists among individuals of the same species in response to a given stimulus. A much greater response variation exists among species of aquatic organisms. Thus, aquatic organisms do not exhibit the same degree of harm, individually or by species, from a given concentration of a toxicant or potential toxicant within the environment.

In establishing a level or concentration of a quality constituent as a criterion, it is necessary to ensure a reasonable degree of safety for those more sensitive species that are important to the functioning of the aquatic ecosystem, even though data on such species' response to the quality constituent under consideration may not be available. The aquatic food web is an intricate relationship of predator and prey organisms. A water constituent that may in some way destroy or eliminate an important segment of that food web would, in all likelihood, destroy or seriously impair other organisms associated with it.

Although experimentation relating to the effects of particular substances under controlled conditions began in the early 1900s, the effects of any substance on more than a few of the vast number of aquatic organisms have not been investigated. Certain test animals have been selected by investigators for intensive investigation because of their importance to man, their availability to the researcher, and their physiological responses to the laboratory environment. As general indicators of organism responses, such test organisms are representative of the expected results for other associated organisms. In this context, *Daphnia* or stoneflies or other associated organisms indicate the general levels of toxicity to be expected among untested species. In addition, test organisms are

themselves vital links within the food web that result in the fish population in a particular waterway.

The ideal data base for criteria development would consist of information on a large percentage of aquatic species and would show the community response to a range of concentrations for a tested constituent during a long time period. This data is not available, but investigators are beginning to derive such information for a few water constituents. Where only 96-hour bioassay data are available, judgmental prudence dictates that a substantial safety factor be employed to protect all life stages of the test organism in waters of varying quality, as well as associated organisms within the aquatic environment that have not been tested and that may be more sensitive to the test constituent.

Application factors have been used to provide the degree of protection required. Safe levels for certain chlorinated hydrocarbons and heavy metals were estimated by applying an 0.01 application factor to the 96-hour LC50 value for sensitive aquatic organisms. Flow-through bioassays have been conducted for some test indicator organisms over a substantial period of their life history. In a few other cases, information is available for the organism's natural life or for more than one generation of the species. Such data may indicate a minimal effect level, as well as a no-effect level.

The word "criterion" should not be used interchangeably with or as a synonym for the word "standard." The word "criterion" represents a constituent concentration or level associated with a degree of environmental effect upon which scientific judgment may be based. As it is currently associated with the water environment, it has come to mean a designated concentration of a constituent that, when not exceeded, will protect an organism, an organism community, or a prescribed water use or quality with an adequate degree of safety. A criterion, in some cases, may be a narrative statement instead of a constituent concentration. On the other hand, a standard connotes a legal entity for a particular reach of waterway or for an effluent. A water quality standard may use a water quality criterion as a basis for regulation or enforcement, but the standard may differ from a criterion because of prevailing local natural conditions — such as naturally occurring organic acids — or because of the importance of a particular waterway, economic considerations, or the degree of safety to a particular ecosystem that may be desired.

Toxicity to aquatic life generally is expressed in terms of acute (short-term) or chronic (long-term) effects. Acute toxicity refers to effects occurring in a short time period: often death is the end point. Acute toxicity can be expressed as the lethal concentration for a stated percentage of organisms tested, or the reciprocal, which is the tolerance limit of a percentage of surviving organisms. Acute toxicity for aquatic organisms generally has been expressed for 24- to 96-hour exposures.

Chronic toxicity refers to effects through an extended time period. Chronic toxicity may be expressed in terms of an observation period equal to the lifetime of an organism or to the time span of more than one generation. Some chronic effects may be reversible, but most are not.

Chronic effects often occur in the species population rather than in the individual. If eggs fail to develop or the sperm does not remain viable, the species would be eliminated from an ecosystem because of reproductive failure. Physiological stress may make a species less competitive with others and may result in a gradual population decline or absence from an area.

The elimination of a microcrustacean that serves as a vital food during the larval period of a fish's life could result ultimately in the elimination of the fish from an area. The phenomenon of bioaccumulation of certain materials may result in chronic toxicity to the ultimate consumer in a food chain. Thus, fish may mobilize lethal toxicants from their fatty tissues during periods of physiological stress; egg shells of predatory birds may be weakened to a point of destruction in the nest; and bird chick embryos may have increased mortality rates. There may be a hazard to the health of man if aquatic organisms with toxic residues are consumed.

The fact that living systems (i.e., individuals, populations, species, and ecosystems) can take up, accumulate, and bioconcentrate human-made and natural toxicants is well documented. In aquatic systems, biota are exposed directly to pollutant toxicants through submersion in a relatively efficient solvent (water) and are exposed indirectly through food webs and other biological, chemical, and physical interactions. Initial toxicant levels, if not immediately toxic and damaging, may accumulate in the biota or sediment over time and increase to levels that are lethal or

sublethally damaging to aquatic organisms or to consumers of these organisms. Water quality criteria reflect a knowledge of the capacity for environmental accumulation, persistence, and effects of specific toxicants in specific aquatic systems.

Ions of toxic materials frequently cause adverse effects because they pass through the semipermeable membranes of an organism. Molecular diffusion through membranes may occur for some compounds such as pesticides, polychlorinated biphenyls, and other toxicants. Some materials may not pass through membranes in their natural or waste-discharged state, but in water they may be converted to states that have increased ability to affect organisms. For example, certain microorganisms can methylate mercury, thus producing a material that more readily enters physiological systems. Some materials may have multiple effects: for example, an iron salt may not be toxic; an iron floc or gel may be an irritant or clog fish gills to effect asphyxiation; iron at low concentrations can be a trace nutrient but at high concentrations it can be a toxicant.

Materials also can affect organisms if their metabolic byproducts cannot be excreted. Unless otherwise stated, criteria are based on the total concentration of the substance because an ecosystem can produce chemical, physical, and biological changes that may be detrimental to organisms living in or using the water.

In prescribing water quality criteria, certain fundamental principles dominate the reasoning process. Establishing a level or concentration as a criterion for a given constituent assumed that other factors within the aquatic environment are acceptable to maintain the integrity of the water. Interrelationships and interactions among organisms and their environment, as well as the interrelationships of sediments and the constituents they contain to the water above, are recognized as fact.

Antagonistic and synergistic reactions among many quality constituents in water also are recognized as fact. The precise definition of such reactions and their relative effects on particular segments of aquatic life have not been identified with scientific precision. Historically, much of the data to support criteria development was of an ambient concentration-organism response nature. Recently, data are becoming available on long-term chronic effects on particular species. Studies now determine carcinogenic, teratogenic, and other insidious effects of toxic materials.

Some unpolluted waters in the Nation may exceed designated criteria for particular constituents. The natural quality of water is variable and certain organisms become adapted to that quality, which may be considered extreme in other areas. Likewise, a single criterion cannot identify minimal quality for the protection of the integrity of water for every aquatic ecosystem in the Nation. Providing an adequate degree of safety to protect against long-term effects may result in a criterion that cannot be detected with present analytical tools. In some cases, a mass balance calculation can assure that the integrity of the waterway is not being degraded.

Water quality criteria do not have direct regulatory impact, but they form the basis for judgment in several U.S. Environmental Protection Agency programs that are derived from water quality considerations. For example, water quality standards developed by the States under section 303 of the Act and approved by EPA are to be based on the water quality criteria, appropriately modified to take account of local conditions. The local conditions to be considered include actual and projected uses of the water, natural background levels of particular constituents, the presence or absence of sensitive important species, characteristics of the local biological community, temperature and weather, flow characteristics, and synergistic or antagonistic effects of combinations of pollutants.

Similarly, by providing a judgment on desirable levels of ambient water quality, water quality criteria are the starting point in deriving toxic pollutant effluent standards pursuant to section 307(a) of the Act. Other EPA programs that use water quality criteria involve drinking water standards, the ocean dumping program, designation of hazardous substances, dredge spoil criteria development, removal of in-place toxic materials, thermal pollution, and pesticide registration.

To provide the water resource protection for which they are designed, quality criteria should apply to virtually all of the Nation's navigable waters with modifications for local conditions as needed. To violate quality criteria for any substantial length of time or in any substantial portion of a waterway may result in an adverse affect on aquatic life and perhaps a hazard to humans or other consumers of aquatic life.

Quality criteria have been designed to provide long-term protection. Thus, they may provide a basis for effluent standards, but criteria values need not become effluent standards. Certain substances can be applied to the aquatic environment with the concurrence of a governmental agency for the precise purpose of controlling or managing a portion of the aquatic ecosystem; aquatic herbicides and pesticides are examples of such substances. For such occurrences, criteria obviously do not apply.

In addition, pesticides applied according to official label instructions to agricultural lands and forestlands may be washed to a receiving waterway by a torrential rainstorm. Under such conditions, such diffuse source inflows should receive consideration similar to that of a discrete effluent discharge, and in such instances the criteria should be applied to the principal portion of the waterway rather than to that peripheral portion receiving the diffuse inflow.

The format for presenting water quality criteria includes a concise statement of the dominant criterion or criteria for a particular constituent followed by a narrative introduction and a listing of the references cited within the rationale. An effort has been made to restrict supporting data to those that either have been published or are in press awaiting publication. A particular constituent may have more than one criterion to ensure more than one water use or condition, i.e., hard or soft water where applicable, suitability as a drinking water supply source, protection of human health when edible portions of selected biota are consumed, provision for recreational bathing or waterskiing, and permitting an appropriate factor of safety to ensure protection for essential warm- or coldwater associated biota.

Criteria are presented for those substances that may occur in water where data indicate the potential for harm to aquatic life, or to water users, or to the consumers of the water or aquatic life. Presented criteria do not represent an all-inclusive list of constituent contaminants. Omissions from criteria should not be construed to mean that an omitted quality constituent is either unimportant or nonhazardous.

Excerpted from "Quality Criteria for Water," 1976, available from National Technical Information Service, #PB-263-943.

Appendix E

Bioconcentration Factors

Used in the Calculation of Human Health Criteria

CHEMICAL NAME	BCF (3% lipid; 1/kg)
Acenaphthene	242
Acrolein	215
Acrylonitrile	30
Aldrin	4,670
Anthracene	30
Antimony	1
Arsenic	44
Benzene	5.2
Benzidine	87.5
Benzofluoranthene, 3,4-	30
Benzo(A) Anthracene	30
Benzo(A) Pyrene	30
Benzo(K) Fluoranthene	30
Bromoform	3.75
Butylbenzyl Phthalate	414
Cadmium	64
Carbon Tetrachloride	18.75
Chlordane	14,100
Chlorobenzene	10.3
Chlorodibromomethane	3.75
Chloroform	3.75
Chloronaphthalene, 2-	202
Chlorophenol, 2-	134
Chrysene	30
Copper	36
Cyanide	1
DDT	53,600
DDD	53,600
DDE	53,600
Dibenzo(A,H) Anthracene	30
Di-N-Butyl Phthalate	89
Dichlorobenzene, 1,2-	55.6
Dichlorobenzene, 1,3-	55.6
Dichlorobenzene, 1,4-	55.6
Dichlorobenzidine, 3,3-	312
Dichlorobromethane	3.75

CHEMICAL NAME	BCF (3% lipid; 1/kg)
Dichloroethane, 1,1-	—
Dichloroethane, 1,2-	1.2
Dichloroethylene, 1,1-	5.6
Dichloroethylene, trans, 1,2-	1.58
Dichlorophenol, 2,4-	40.7
Dichloropropane, 1,2-	4.11
Dichloropropylene, 1,3-	1.91
Dieldrin	4,670
Diethyl Phthalate	73
Dimethylphenol, 2,4-	93.8
Dimethyl Phthalate	36
Dinitrophenol, 2,4-	1.5
2-Methyl-4,6-Dinitrophenol	5.5
Dinitrotoluene, 2,4-	3.8
Dioxin (2,3,7,8-TCDD)	5,000
Diphenylhydrazine, 1,2-	24.9
Di-2-Ethylhexyl Phthalate	130
Endosulfan-Alpha	270
Endosulfan-Beta	270
Endosulfan Sulfate	270
Endrin	3,970
Endrin Aldehyde	3,970
Bis(2-Chloroethyl) Ether	6.9
Bis(2-Chloroisopropyl) Ether	2.47
Bis(Chloromthyl) Ether	0.63
Ethylbenzene	37.5
Fluoranthene	1,150
Fluorene	30
Heptachlor	11,200
Heptachlor Epoxide	11,200
Hexachlorobenzene	8,690
Hexachlorobutadiene	2.78
Hexachlorocyclohexane-Alpha (A-BHC)	130
Hexachlorocyclohexane-Beta (B-BHC)	130
Hexachlorocyclohexane-Gamma (G-BHC)	130
Hexachlorocyclohexane-Delta (D-BHC)	130
Hexachlorocyclopentadiene	4.34
Hexachloroethane	86.9
Indeno (1,2,3-CD) Pyrene	30
Isophorone	4.38
Mercury	5,500 Freshwater
	3,760 Esturine
	9,000 Marine
Methyl Bromide	3.75
Methylene Chloride	0.9
Naphthalene	10.5

CHEMICAL NAME	BCF (3% lipid; 1/kg)
Nickel	47
Nitronenzene	2.89
N-Nitrosodimethylamine	0.026
N-Nitrosodiphenylamine	136
N-Nitrosodi-N-Propylamine	1.13
PCB-1016	31,200
PCB-1221	31,200
PCB-1232	31,200
PCB-1242	31,200
PCB-1248	31,200
PCB-1254	31,200
PCB-1260	31,200
Pentachlorophenol	11
Phenanthrene	30
Phenol	1.4
Pyrene	30
Selenium	6
Silver	0.5
Tetrachlorobenzene, 1,2,4,5-	1,125
Tetrachlorethane, 1,1,2,2-	5
Tetrachloroethylene	30.6
Thallium	116
Toluene	10.7
Toxaphene	13,100
Trichloroethane, 1,1,1-	5.6
Trichloroethane, 1,1,2-	10.6
Trichlorophenol, 2,4,6-	150
Vinyl Chloride	1.17
Zinc	47

NOTE: The bioconcentration factors (BCFs) listed here were used in the calculation of the human health criteria in this document. EPA is considering updating these BCFs or using Bioaccumulation Factors (BAFs). Until EPA formally updates these BCFs, these numbers shuld be used to calculate the human health criteria.

APPENDIX F

Water Quality Criteria Documents

The U.S. Environmental Protection Agency has published water quality criteria for toxic pollutant(s) categories. Copies of water quality criteria documents are available from the National Technical Information Service (NTIS), 5285 Front Royal Road, Springfield, VA 22161, (703) 487-4650. Prices of individual documents may be obtained by contacting NTIS. Order numbers are listed below. Where indicated, documents may be obtained from the Water Resource Center, 401 M St., S.W. RC-4100, Washington, DC 20460, (202) 260-7786.

Chemical	NTIS Order No.	EPA Document No.
Acenaphthene	PB 81-117269	EPA 440/5-80-015
Acrolein	PB 81-117277	EPA 440/5-80-016
Acrylonitrile	PB 81-117285	EPA 440/5-80-017
Aesthetics	PB 263943	EPA 440/9-76-023
Aldrin/Dieldrin	PB 81-117301	EPA 440/5-80-019
Alkalinity	PB 263943	EPA 440/9-76-023
Aluminum	PB 88-245998	EPA 440/5-86-008
Ammonia	PB 85-227114	EPA 440/5-85-001
Ammonia (saltwater)	PB 89-195242	EPA 440/5-88-004
Antimony	PB 81-117319	EPA 440/5-80-020
Antimony (III) — aquatic (draft)	resource center	
Arsenic — 1980	PB 81-117327	EPA 440/5-80-021
— 1984	PB 85-227445	EPA 440/5-84-033
Asbestos	PB 81-117335	EPA 440/5-80-022
Bacteria — 1976	PB 263943	EPA 440/9-76-023
— 1984	PB 86-158045	EPA 440/5-84-002
Barium	PB 263943	EPA 440/9-76-023
Benzene	PB 81-117293	EPA 440/5-80-018
Benzidine	PB 81-117343	EPA 440/5-80-023
Beryllium	PB 81-117350	EPA 440/5-80-024
Boron	PB 263943	EPA 440/9-76-023
Cadmium — 1980	PB 81-117368	EPA 440/5-80-025
— 1984	PB 85-224031	EPA 440/5-84-032
Carbon Tetrachloride	PB 81-117376	EPA 440/5-80-026
Chlordane	PB 81-117384	EPA 440/5-80-027
Chloride	PB 88-175047	EPA 440/5-88-001
Chlorinated Benzenes	PB 81-117392	EPA 440/5-80-028
Chlorinated Ethanes	PB 81-117400	EPA 440/5-80-029
Chlorinated Naphthalene	PB 81-117426	EPA 440/5-80-031
Chlorinated Phenols	PB 81-117434	EPA 440/5-80-032

Chemical	NTIS Order No.	EPA Document No.
Chlorine	PB 85-227429	EPA 440/5-84-030
Chloroalkyl Ethers	PB 81-117418	EPA 440/5-80-030
Chloroform	PB 81-117442	EPA 440/5-80-033
2-Chlorophenol	PB 81-117459	EPA 440/5-80-034
Chlorophenoxy Herbicides	PB 263943	EPA 440/9-76-023
Chlorpyrifos	PB 87-105359	EPA 440/5-86-005
Chromium — 1980	PB 81-117467	EPA 440/5-80-035
—1984	PB 85-227478	EPA 440/5-84-029
Color	PB 263943	EPA 440/9-76-023
Copper — 1980	PB 81-117475	EPA 440/5-80-036
— 1984	PB 85-227023	EPA 440/5-84-031
Cyanide	PB 85-227460	EPA 440/5-84-028
Cyanides	PB 81-117483	EPA 440/5-80-037
DDT and Metabolites	PB 81-117491	EPA 440/5-80-038
Demeton	PB 263943	EPA 440/9-76-023
Dichlorobenzenes	PB 81-117509	EPA 440/5-80-039
Dichlorobenzidine	PB 81-117517	EPA 440/5-80-040
Dichloroethylenes	PB 81-117525	EPA 440/5-80-041
2,4-Dichlorophenol	PB 81-117533	EPA 440/5-80-042
Dichloropropane/ Dichloropropene	PB 81-117541	EPA 440/5-80-043
2,4-Dimethylphenol	PB 81-117558	EPA 440/5-80-044
Dinitrotoluene	PB 81-117566	EPA 440/5-80-045
Diphenylhydrazine	PB 81-117731	EPA 440/5-80-062
Di-2-Ethylhexyl Phthalate — aquatic (draft)	resource center	
Dissolved Oxygen	PB 86-208253	EPA 440/5-86-003
Endosulfan	PB 81-117574	EPA 440/5-80-046
Endrin	PB 81-117582	EPA 440/5-80-047
Ethylbenzene	PB 81-117590	EPA 440/5-80-048
Fluoranthene	PB 81-117608	EPA 440/5-80-049
Gasses, Total Dissolved	PB 263943	EPA 440/9-76-023
Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses	PB 85-227049	
Guthion	PB 263943	EPA 440/9-76-023
Haloethers	PB 81-117616	EPA 440/5-80-050
Halomethanes	PB 81-117624	EPA 440/5-80-051
Hardness	PB 263943	EPA 440/9-76-023
Heptachlor	PB 81-117632	EPA 440/5-80-052
Hexachlorobenzene — aquatic (draft)	resource center	
Hexachlorobutadiene	PB 81-117640	EPA 440/5-80-053
Hexachlorocyclohexane	PB 81-117657	EPA 440/5-80-054

Chemical	NTIS Order No.	EPA Document No.
Hexachlorocyclopentadiene	PB 81-117665	EPA 440/5-80-055
Iron	PB 263943	EPA 440/9-76-023
Isophorone	PB 81-117673	EPA 440/5-80-056
Lead — 1980	PB 81-117681	EPA 440/5-80-057
— 1984	PB 85-227437	EPA 440/5-84-027
Malathion	PB 263943	EPA 440/9-76-023
Manganese	PB 263943	EPA 440/9-76-023
Mercury — 1980	PB 81-117699	EPA 440/5-80-058
— 1984	PB 85-227452	EPA 440/5-84-026
Methoxychlor	PB 263943	EPA 440/9-76-023
Mirex	PB 263943	EPA 440/9-76-023
Naphthalene	PB 81-117707	EPA 440/5-80-059
Nickel — 1980	PB 81-117715	EPA 440/5-80-060
— 1986	PB 87-105359	EPA 440/5-86-004
Nitrates/Nitrites	PB 263943	EPA 440/9-76-023
Nitrobenzene	PB 81-117723	EPA 440/5-80-061
Nitrophenols	PB 81-117749	EPA 440/5-80-063
Nitrosamines	PB 81-117756	EPA 440/5-80-064
Oil and Grease	PB 263943	EPA 440/9-76-023
Parathion	PB 87-105383	EPA 440/5-86-007
Pentachlorophenol — 1980	PB 81-117764	EPA 440/5-80-065
— 1986	PB 87-105391	EPA 440/5-85-009
pH	PB 263943	EPA 440/9-76-023
Phenanthrene — aquatic (draft)	resource center	
Phenol	PB 81-117772	EPA 440/5-80-066
Phosphorus	PB 263943	EPA 440/9-76-023
Phthalate Esters	PB 81-117780	EPA 440/5-80-067
Polychlorinated Biphenyls	PB 81-117798	EPA 440/5-80-068
Polynuclear Aromatic Hydrocarbons	PB 81-117806	EPA 440/5-80-069
Selenium — 1980	PB 81-117814	EPA 440/5-80-070
— 1987	PB 88-142239	EPA 440/5-87-008
Silver	PB 81-117822	EPA 440/5-80-071
Silver — aquatic (draft)	resource center	
Solids (dissolved) and Salinity	PB 263943	EPA 440/9-76-023
Solids (suspended) and Turbidity	PB 263943	EPA 440/9-76-023
Sulfides/Hydrogen Sulfide	PB 263943	EPA 440/9-76-023
Tainting Substances	PB 263943	EPA 440/9-76-023
Temperature	PB 263943	EPA 440/9-76-023
2,3,7,8-Tetrachlorodibenzo-P-Dioxin	PB 89-169825	EPA 440/5-84-007
Tetrachloroethylene	PB 81-117830	EPA 440/5-80-073
Thallium	PB 81-117848	EPA 440/5-80-074
Toluene	PB 81-117863	EPA 440/5-80-075

Chemical	NTIS Order No.	EPA Document No.
Toxaphene — 1980	PB 81-117863	EPA 440/5-80-076
— 1986	PB 87-105375	EPA 440/5-86-006
Tributyltin — aquatic (draft)	resource center	
Trichloroethylene	PB 81-117871	EPA 440/5-80-077
2,4,5-Trichlorophenol — aquatic (draft)	resource center	
Vinyl Chloride	PB 81-117889	EPA 440/5-80-078
Zinc — 1980	PB 81-117897	EPA 440/5-80-079
— 1987	PB 87-143581	EPA 440/5-87-003