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COMPARATIVE TOXICOLOGY OF MARINE FISHES AND CRUSTACEANS

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Comparative Toxicology of Marine Fishes and Crustaceans*

Glenn W. Suter, II, and Aaron E. Rosen

ABSTRACT. Data collected on the effects of chemicals on marine fishes and crustaceans was evaluated for its predictive power of assessing risks to marine resources. The data sets consisted of acute median lethal concentrations (LC_{50}s) and chronic maximum acceptable toxicant concentrations (MATCs) and were analyzed using errors-in-variables regression models and simple comparisons. The major conclusions were (1) the variability found in the marine data is comparable to that found in freshwater data; (2) the variance appears to be primarily due to variation in the response of organisms to chemicals rather than to variation among test laboratories; (3) the standard marine test fish, sheepshead minnow (Cyprinodon variegatus), appears to be representative of marine fishes; (4) the responses of marine crustaceans are so highly diverse that the concept of a representative crustacean is questionable; (5) mysid and penaeid shrimp appear to be particularly sensitive to toxic chemicals; (6) chronic responses of marine species are predictable from their acute responses with reasonable precision; and (7) given the taxonomic distances involved, chronic responses of standard freshwater test species are good predictors of chronic responses of standard marine test species.

1. INTRODUCTION

Fishes and crustaceans inhabiting coastal marine waters are subject to the effects of a variety of pollutants, habitat loss, harvesting, entrainment in water intakes, and natural stresses. Changes in the abundance of these organisms are apparent but difficult to explain. The goal of this project was to collect data on the effects of chemicals on marine fishes and crustaceans and evaluate the predictive power of the data using environmental risk assessment methods developed for the Environmental Protection Agency (EPA) (Barnthouse and Suter, 1986). The results would be a tool for determining locations where pollutants may be affecting coastal stocks of fishes and crustaceans.

The specific objectives were (1) to evaluate the utility of existing marine toxicity data for developing the types of taxonomic and acute-chronic extrapolation formulas that have been used for risk analysis of toxic effects on freshwater organisms (Suter et al., 1983, 1986), (2) to examine the representativeness of the standard marine test species, (3) to compare the relative sensitivities to toxicants of different marine species, and (4) to evaluate the feasibility of extrapolating from freshwater to marine species.

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2. METHODS

2.1 Data Sets

In this study, we used four data sets pertaining to persistence of toxicity in freshwater and marine species over various states of their life cycles. The first is a marine chronic toxicity data set consisting of data from studies reporting acceptable life cycle, partial life cycle, or early life stage maximum acceptable toxicant concentrations (MATCs) for marine or estuarine fishes or crustaceans. The MATC is the geometric mean of the lowest concentration producing a statistically significant effect and the highest concentration producing no such effect on survival, growth, or fecundity in any life stage in a life cycle, partial life cycle, or early life stage test. It is used as a threshold for toxic effects in exposures of indefinite duration but does not correspond to any particular level or type of effect on any particular life stage. The MATCs and associated 96-h median lethal concentrations (LC_{50}s) for 114 species-chemical pairs are listed in Table 1. The second is an equivalent set of chronic data for freshwater fishes, containing the results of 177 chronic tests (Barnthouse and Suter, 1986). The third is a set of chronic data (MATCs and associated acute LC_{50}s) for Daphnia, the standard freshwater crustacean test species. The Daphnia data were taken from the EPA's national water quality criteria support documents (EPA, 1980, 1985). The last data set consists of the 2580 LC_{50} values for saltwater fishes and crustaceans in the EPA aquatic toxicity data base AQUIRE.

2.2. Regression Analysis

To extrapolate data between taxa and between toxicological benchmarks (numeric expressions of the results of toxicity tests, e.g., LC_{50}s and MATCs), we used an errors-in-variables regression model (Suter et al., 1986). The choice of extrapolation model for this method was based on the following characteristics of toxicity data: (1) the observed values of both the independent variable (X) and dependent variable (Y) are subject to error of measurement and to inherent variability; (2) X is not a controlled variable (like settings on a thermostat); (3) values assumed by X and Y are open-ended and nonnormally distributed (Ricker, 1973).

These characteristics suggest that an ordinary least squares regression model would be inappropriate and an errors-in-variables model should be used (Ricker, 1973). Since we can estimate the value of λ, the ratio of the point variances of Y to X, a functional model, provides maximum likelihood estimators of the regression parameters. For further discussion of errors-in-variables regression models, see Wald (1940); Ricker (1973); and Mandel (1984).

The estimators of the slope (b) and intercept (a) are

$$b = \{\Sigma y^2 - \lambda\Sigma x^2 + [(\Sigma y^2 - \lambda\Sigma x^2)^2 + 4\lambda(\Sigma xy)^2]^{1/2}\}/2\Sigma xy, \text{ and}$$

$$a = \bar{y} - b\bar{x},$$

where $x=x_i-\bar{X}$ and $y=Y_i-\bar{Y}$ for $i=1...n$.

Table 1. Marine chronic data set (all values are mg/L)

Chemical	Species[a]	LC_{50}	MATC	MATC type[b]	Source
AC 222, 705	CV	1.1	0.04	ELS	Hansen et al., 1983
Acenaphthene	CV	3,100	710	ELS	Ward et al., 1981c
Acephate	MB	7,300	900	LC	EPA, 1981
Silver nitrate	MB	249	19	LC	Lussier et al., in press
	MB	141	15	LC	McKenney, no date
	MB	--	59	LC	McKenney, no date
	MB	300	53	LC	McKenney, no date
Aldicarb	MB	16	1.2	LC	EPA, 1981
Ammonium jarosite	CV	500,000	180,000	ELS	Ward et al., 1982
Aroclor 1016	CV	--	7.1	ELS	Hansen et al., 1975
Aroclor 1254	CV	--	0.098	ELS	Schimmel et al., 1974a
Arsenic	MB	1,740	893	LC	Lussier et al., in press
Atrazine	MB	1,000	123	LC	Ward and Ballentine, 1985
	CV	16,000	2,542	ELA	Ward and Ballentine, 1985
Bis(tributyltin) oxide	CV	0.96	3.7	LC	Ward et al., 1981a
Bromoform	CV	7,100	6,400	ELS	Ward et al., 1981c
Carbofuran	CV	386	19	LC	Parrish et al., 1977
Carbophenothion	PP	2.9	0.28	LC	EPA, 1981
	MB	3.0	0.76	LC	EPA, 1981
	CV	2.8	1.9	ELS	EPA, 1981
Cadmium	MB	15.5	5.5	LC	Nimmo et al., 1977, 1978
	MB	110	7.1	LC	Gentile et al., 1982b
Chlordane	CV	12.5	0.6	LC	Parrish et al., 1978
	CV	24.5	11	ELS	Parrish et al., 1976
Oxidants	MP	0.054	0.072	ELS	Goodman et al., 1983
Chlorpyrifos	MM	1.7	0.37	ELS	Goodman et al., 1985
	LT	1.3	1.2	ELS	Goodman et al., 1985
	MY	4.2	0.54	ELS	Goodman et al., 1985
	MP	1.3	46	ELS	Goodman et al., 1985
	CV	--	2.3	ELS	Cripe et al., in press
	OB	560	<3.7	ELS	Hansen et al., in press
Chlorine	MP	54	46	ELS	EPA, 1985
Cyanide	MB	113	70	LC	Lussier et al., in press
	CV	300	36	ELS	EPA, 1985
Chromium	MB	2030	132	LC	Lussier et al., in press
Copper	MB	181	54	LC	Lussier et al., in press
DEF (S,S,S-tributyl-phosphorothrithioate)	MB	4.55	<0.34	LC	Nimmo et al., 1981
Diazinon	MB	4.82	1.94	LC	Nimmo et al., 1981
Diazinon	CV	1.470	<0.47	LC	Goodman et al., 1979
Dieldrin	EA	23	1.4	LC	Daniels and Allan, 1981
Dieldrin	MB	4.5	0.73	LC	EPA. 1980
Dimilin	MB	2.06	<0.075	LC	Nimmo et al., 1980

Table 1. (continued)

Chemical	Species[a]	LC_{50}	MATC	MATC type[b]	Source
Endosulfan	MB	1.37	0.48	LC	EPA, 1980
	MB	1.30	0.48	LC	McKenney, no date
	MB	1.05	0.21	LC	McKenney, no date
	MB	5.00	0.41	LC	McKenney, no date
	MB	4.60	0.25	LC	McKenney, no date
	CV	0.95	0.40	ELS	EPA, 1980
	CV	--	0.68	ELS	Hansen and Cripe, 1984
	CV	--	0.41	ELS	Hansen and Cripe, 1984
	CV	--	0.41	ELS	Hansen and Cripe, 1984
	CV	--	0.37	ELS	Hansen and Cripe, 1984
	CV	--	1.77	ELS	Hansen and Cripe, 1984
	CV	--	0.29	ELS	Hansen and Cripe, 1984
	CV	--	0.86	ELS	Hansen and Cripe, 1984
	CV	--	0.83	ELS	Hansen and Cripe, 1984
	CV	--	0.27	ELS	Hansen and Cripe, 1984
	CV	--	0.45	ELS	Hansen and Cripe, 1984
Endrin	PP	0.35	0.04	LC	Tyler-Schroeder, 1979
	MB	0.185	0.01	LC	McKenny, 1982
	CV	0.34	0.19	LC	Hansen et al., 1977a
	CV	0.38	0.31	ELS	Schimmel et al., 1974b
EPN (o-Ethyl o-p-nitrophenyl-phenylthiophosphonate)	MB	3.01	1.35	LC	Nimmo et al., 1981
	CV	--	5.7	LC	Cripe et al., 1984
Ethoprop	MB	7.5	0.47	LC	EPA, 1981
	CV	180	16	ELS	EPA, 1981
Fenvalerate	CV	5	1.1	ELS	Hansen et al., 1983
Fluoranthene	MB	40	16	LC	EPA, 1980
Guthion	CV	--	0.35	LC	Cripe et al., 1984
Heptachlor	CV	3.68	1.58	ELS	Goodman et al., 1976
	CV	10.5	1.4	LC	Hansen and Parrish, 1977
Mercury	MB	3.5	1.1	LC	Gentile et al., 1982a and 1983
Isophorone	CV	140,000	110,000	ELS	Ward et al., 1981c
Kepone	EA	40	7.1	LC	Allan and Daniels, 1980
	MB	10.1	0.043	LC	Nimmo et al., 1977
	CV	69.5	0.094	LC	Goodman et al., 1982.
	CV	69.5	0.08	LC	Hansen et al., 1977b
Leptophos	MB	3.31	1.06	LC	Nimmo et al., 1981
Malathion	CV	51	6	LC	Parrish et al., 1977
Methoxychlor	CV	49	17	LC	Parrish et al., 1977
Methyl Parathion	MB	0.77	0.13	LC	Nimmo et al., 1981
Nickel	MB	508	93	LC	Gentile et al., 1982a
Lead	MB	3,130	25	LC	Lussier et al., in press

Table 1. (continued)

Chemical	Species[a]	LC_{50}	MATC	MATC type[b]	Source
Pentachlorobenzene	CV	--	26	ELS	Hansen and Cripe, 1984
	CV	--	29	ELS	Hansen and Cripe, 1984
	CV	--	66	ELS	Hansen and Cripe, 1984
	CV	--	87	ELS	Hansen and Cripe, 1984
	CV	--	145	ELS	Hansen and Cripe, 1984
	CV	--	73	ELS	Hansen and Cripe, 1984
	CV	--	155	ELS	Hansen and Cripe, 1984
	CV	--	88	ELS	Hansen and Cripe, 1984
	CV	--	74	ELS	Hansen and Cripe, 1984
Pentachloroethane	MB	5,060	580	LC	EPA, 1980
Pentachlorophenol	CV	420	64	LC	Parrish et al., 1978
Permethrin	CV	7.8	1.06	ELS	Hansen et al., 1983
Phorate	MB	0.33	0.11	LC	Nimmo et al., 1981
	CV	1.3	0.31	ELS	EPA, 1981
Selenium	MB	1,500	212	LC	Ward et al., 1981b
	CV	7,400	675	ELS	Ward et al., 1981b
Tetrachloroethylene	MB	10,200	450	LC	EPA, 1980
Thiobencarb	MB	330	28	LC	McKenney, no date
Thallium	CV	20,900	6,000	ELS	EPA, 1980
Toluene	CV	13,000	5,000	ELS	Ward et al., 1981c
Toxaphene	MB	2.69	0.10	LC	Nimmo et al., 1981
	CV	1.1	1.7	ELS	Goodman et al., 1976
Trifluralin	CV	190	2.5	LC	Parrish et al., 1978
Zinc	MB	499	166	LC	Lussier et al., in press
1-Chloronaphthalene	CV	690	555	ELS	Ward et al., 1981c
1,2,4-Trichlorobenzene	CV	21,400	222	ELS	EPA, 1980
1,2,4,5-Tetrachloro-benzene	CV	330	129	ELS	Ward et al., 1981c
1,3-Dichloropropane	MB	10,300	3,040	LC	EPA, 1980
2,4-Dicholoro-6-methyl-phenol	CV	3,700	360	ELS	Ward et al., 1981c
2,4-Dinitrophenol	CV	29,400	7900	ELS	EPA, 1980.
4-Nitrophenol	CV	32,000	12,650	ELS	Ward et al., 1981c

[a]CV = *Cyprinodon variegatus* , EA = *Eurytemora affinis*, LT = *Leuresthes tenuis*, MB = *Mysidopsis bahia*, MM = *Media menidia*, MP = *Menidia peninsulae*, MY = *Menidia beryllina*, OB = *Opsanus beta*, PP = *Palaemonetes pugio*.

[b]ELS = early life stage, LC = Life cycle or partial life cycle.

The variance of a single predicted Y-value for a given X-value ($X = X_0$) is given in Mandel (1984) as

$$var(Y|X_0) = s_e^2 \{1+1/n+[1+(b^2/\lambda)]^2[(X_0-\bar{X})^2/\Sigma u^2]\},$$

where

$$s_e^2 = (b^2\Sigma x^2-2b\Sigma xy+\Sigma y^2)/(n-2), \text{ and}$$

$$\Sigma u^2 = \Sigma x^2+2b/\lambda\Sigma xy+(b^2/\lambda)\Sigma y^2.$$

This variance is the appropriate value to use in calculating confidence intervals and risk estimates because the interest in this case is the certainty concerning an individual future observation of Y, such as a toxic threshold for an untested species-chemical combination. This variance is larger than the variance on the mean of a Y X_0, which in turn is larger than the variance of the regression coefficient, which is the number provided by most programmable calculators. Confidence intervals calculated from this variance are larger than those that are conventionally reported and are referred to as prediction intervals (Wonnacott and Wonnacott, 1985).

For ease in using of this method, we reduce the variance formula to

$$var(X|Y_0) = F_1 + F_2(X_0 - \bar{X})^2.$$

and where the inverse regression is useful, the variance of a predicted X-value is

$$var(X|Y_0) = G_1 + G_2(Y_0 - \bar{Y})^2.$$

We provide values for F_1, F_2, G_1, and G_2 in Tables 2, 4, 6, and 7. These variances are used to calculate 95% prediction intervals and can be used in calculating risks of toxic effects (Suter et al., 1986).

This model requires that λ, the ratio of the point variances of Y to X, be estimated. When extrapolating between common benchmarks for organisms aggregated at the same taxonomic level, λ was set to 1. Otherwise was set to the ratios of the n-weighted means of the variances of benchmarks from replicate tests. When extrapolating from LC_{50}s to MATCs for fishes, was set to the ratio of the mean of variances of all sets of replicate fish MATCs to the mean of variances of all sets of replicate fish LC_{50}s.

2.3 Relative Sensitivity

The possibility that certain species might be particularly sensitive or insensitive to toxic chemicals was examined by ranking and by using the ratio LC_{50}(sp.x)/LC_{50}(sp.r), where sp.r is any species in the AQUIRE data set for which there are at least eight 96-h LC_{50} values and sp.x is any species that has a 96-h LC_{50} for any chemical in common with sp.r. MATC values from the acute-chronic data set were assessed analogously. The geometric mean, standard error, and range of the quotients of these ratios were determined for each sp.r.

2.4 General

All the data used in the regressions are log transformed, and the reported results are for the transformed values. Log transformations were used to induce homogeneity of variance. Test results expressed as greater than or less than values were not used except for ranking. Where replicate data existed for a single combination of test type, species, and chemical, one replicate was chosen at random for each analysis (using the mean of replicates would have artificially reduced the variance).

3. RESULTS

3.1 Taxonomic Extrapolations

Taxonomic extrapolations of marine acute LC_{50}s from the AQUIRE data set are presented in Table 2. The extrapolations are performed between taxa having the next higher taxonomic level in common rather than simply matching all possible species combinations. This approach permits extrapolation to species that have rarely or never been tested by assuming that they are represented by those members of a taxon to which they belong that have been tested. It is based on the concept that taxonomic similarity implies toxicological similarity (Suter et al., 1983; LeBlanc, 1984; Suter et al., 1986). Only those regressions for which there were at least five data pairs and significant correlations were included.

We use the 95% prediction intervals (PIs) on Y at mean X as indicators of the quality of the extrapolations because we are interested in the ability of the model to predict future observations. Comparison of correlation coefficients (r values), as is commonly done, would be appropriate only if we were interested in comparing the ability of different models to explain the variance in existing data. The average PIs for each taxonomic level of marine fishes and crustaceans are presented in Table 3 and compared to earlier results for freshwater taxa (Suter et al., 1986). The extrapolation uncertainty, represented by the average PIs, increases by almost a factor of 2 as we move up the taxonomic hierarchy from congeneric species to orders within Osteichthyes and Crustacea. The mean PIs are quite similar to those for freshwater fishes and arthropods at the same taxonomic levels even though the freshwater data, which came from only one laboratory, would be expected to have less extraneous variance than the AQUIRE data, which came from numerous laboratories. This result suggests that the uncertainty in these extrapolations is primarily due to variance in the response of the organisms rather than to the test methods and conditions.

3.2 Representativeness of Standard Test Species

Recently, the atheriniform fish *Cyprinodon variegatus* (sheepshead minnow) and the mysid shrimp *Mysidopsis bahia* have attained the status of standard marine test species. Eighty-eight percent of the marine MATCs were for one of these species (Table 1), and they are becoming predominant in acute toxicity studies. Table 4 contains the results of regressions to predict the response of higher taxa of fish and crustaceans from test results for these two species. As expected, *C. variegatus* LC_{50}s are reasonably representative of LC_{50}s for

Table 2. Taxonomic extrapolations of log LC_{50} values from the AQUIRE data set

Taxonomic level	Taxon X	Taxon Y	n[a]	PI[b]	Slope	Inter-cept	F_1	F_2	G_1	G_2	$\bar{X}$	$\bar{Y}$
					Osteichthyes							
Species	*Fundulus heteroclitus*	*Fundulus majalis*	12	0.75	1.12	-0.32	0.15	0.01	0.12	0.01	1.67	1.56
Genus	*Cyprinodon*	*Fundulus*	9	0.82	0.96	0.21	0.17	0.12	0.19	0.01	1.33	1.49
Family	Atherinidae	Cyprinodontidae	32	0.83	0.90	0.50	0.18	0.003	0.22	0.004	1.42	1.77
	Mugilidae	Labridae	12	1.74	0.82	0.70	0.79	0.04	1.17	0.09	1.33	1.79
	Cyprinodontidae	Poeciliidae	12	0.53	0.75	0.19	0.07	0.02	0.13	0.05	0.15	0.31
Order	Anguiliformes	Tetraodontiformes	12	1.08	0.89	1.09	0.30	0.02	0.38	0.02	1.40	2.34
	Anguiliformes	Perciformes	34	1.40	0.96	0.21	0.51	0.01	0.55	0.01	1.11	1.28
	Anguiliformes	Gasterosteiformes	8	1.22	1.04	0.52	0.39	0.10	0.36	0.08	0.85	1.40
	Anguiliformes	Atheriniformes	46	0.94	1.05	0.06	0.23	0.003	0.21	0.002	1.31	1.43
	Atheriniformes	Cypriniformes	7	2.69	0.82	1.93	1.89	0.03	2.77	0.08	2.26	3.79
	Atheriniformes	Tetraodontiformes	46	1.06	0.88	1.00	0.29	0.004	0.38	0.01	1.43	2.26
	Atheriniformes	Perciformes	148	1.41	0.92	0.10	0.51	0.001	0.61	0.002	1.27	1.27
	Atheriniformes	Gasterosteiformes	36	0.97	0.94	0.49	0.24	0.01	0.28	0.01	0.92	1.35
	Gasterosteiformes	Tetraodontiformes	8	1.27	1.12	0.31	0.42	0.10	0.33	0.06	1.40	1.88
	Gasterosteiformes	Perciformes	33	1.55	1.15	-0.67	0.62	0.04	0.47	0.02	1.47	1.03
	Perciformes	Tetraodontiformes	34	1.40	0.91	0.93	0.52	0.01	0.62	0.02	1.28	2.10
	Cypriniformes	Perciformes	6	0.84	0.46	-0.35	0.18	0.01	0.86	0.18	3.13	1.10
	Perciformes	Salmoniformes	7	0.15	1.19	-0.76	0.01	0.00	0.01	0.00	2.80	2.80
					Crustacea							
Family	Cacridae	Crangonidae	5	0.94	1.33	-2.25	0.23	0.24	0.13	0.08	4.29	3.46
	Crangonidae	Palaemonidae	21	1.09	0.98	0.48	0.31	0.01	0.32	0.01	1.29	1.74
	Crangonidae	Pandalidae	10[c]	0.61	0.81	0.69	0.10	0.03	0.15	0.06	4.26	4.13
	Crangonidae	Portunidae	9[c]	0.58	1.18	-0.33	0.09	0.03	0.06	0.01	4.00	4.41
	Palaemonidae	Penaeidae	11	1.18	1.14	-1.19	0.37	0.02	0.18	0.004	1.32	0.68
	Palaemonidae	Portunidae	5	1.79	0.71	1.33	0.84	0.03	1.67	0.14	3.68	3.94
	Pandalidae	Portunidae	7[c]	0.48	0.97	0.59	0.06	0.03	0.06	0.04	4.04	4.51
Order	Amphipoda	Decapoda	50	2.22	1.23	-0.69	1.29	0.02	0.85	0.01	1.93	1.68
	Calanoidea	Decapoda	14	2.83	3.05	-3.12	2.08	1.27	0.22	0.01	1.82	2.42
	Decapoda	Harpacticoidea	23	3.16	1.18	1.06	2.59	0.09	1.84	0.05	3.38	3.88
	Decapoda	Mysidae	10	0.75	0.35	0.16	0.15	0.01	1.18	0.49	1.38	0.65

[a] n = number of points, each consisting of LC_{50}s for species from the X and Y taxa tested for the same chemical and duration.

[b] Prediction interval ($\alpha = 0.05$) at the mean ($\bar{X}$) is $\bar{Y} \pm PI$.

[c] Derived from 48-h tests; all others are 96-h.

Table 3. Summary of freshwater and marine taxonomic extrapolations for LC_{50}s

Taxonomic level	n[a]	n-weighted mean of 95% prediction intervals
Species		
Marine fish	1	0.75
Freshwater fish	8	0.76
Freshwater arthropods	2	1.10
Genera		
Marine fish	1	0.82
Freshwater fish	8	0.74
Freshwater arthropods	2	0.78
Families		
Marine fish	3	1.00
Freshwater fish	4	0.97
Marine crustaceans	7	0.94
Freshwater arthropods	3	1.37
Orders		
Marine fish	13	1.27
Freshwater fish	10	1.35
Marine crustaceans	4	2.38
Freshwater arthropods	10	2.06

[a] n = the number of pairs of taxa at that taxonomic level.

Table 4. Extrapolations from the standard test species *Cyprinodon variegatus* and *Mysidopsis bahia*

Species X	Taxon Y	Benchmark	n[a]	PI[b]	Slope	Intercept	F_1	F_2	$\bar{X}$	$\bar{Y}$
C. variegatus	Osteichthyes	LC_{50}	51	1.49	0.97	0.03	0.58	0.01	1.25	1.24
	Atheriniformes	LC_{50}	17	1.00	1.02	0.04	0.26	0.01	1.22	1.28
	Perciformes	LC_{50}	20	1.56	0.82	-0.12	0.63	0.02	1.36	0.99
	Gasterosteiformes	LC_{50}	5	1.10	1.08	0.43	0.31	0.07	0.81	1.30
	Crustacea	LC_{50}	34	2.15	1.14	-0.41	1.21	0.11	1.78	1.62
	Crustacea	MATC	15	1.80	0.93	-0.54	0.84	0.03	0.53	-0.05
M. bahia	Crustacea	LC_{50}	14	2.46	2.89	-0.55	1.57	0.53	0.71	1.50
	Osteichthyes	LC_{50}	29	1.85	1.08	0.35	0.88	0.01	2.42	2.96
	Crustacea	MATC	5	2.22	0.80	0.43	1.28	0.27	-0.57	-1.28
	Osteichthyes[c]	MATC	12	1.40	1.02	0.75	0.51	0.28	0.03	0.78

[a] n = number of points, each consisting of a toxicological benchmark value for a standard test species and an equivalent value for a species belonging to the Y taxon, for the same chemical.

[b] Prediction interval ($\alpha = 0.05$) at the mean ($\bar{X}$) is $\bar{Y} \pm PI$.

[c] All are *C. variegatus*.

other members of the order Atheriniformes (i.e., the slope and intercept are near 1 and 0, and the PI is ± an order of magnitude which is comparable to the family-level extrapolations in Table 3). There is somewhat greater uncertainty in predicting LC_{50}s for all Osteichthyes from C. variegatus (PI = 1.49) and considerably greater uncertainty associated with the prediction of LC_{50}s for Crustacea (PI = 2.15; see Fig. 1a) and MATCs for Crustacea (PI = 1.80; see Fig. 1b). Based on the regression statistics, C. variegatus appears to be less sensitive than Perciformes and Crustacea, more sensitive than Gasterosteiformes, and fairly typical of Osteichthyes in general.

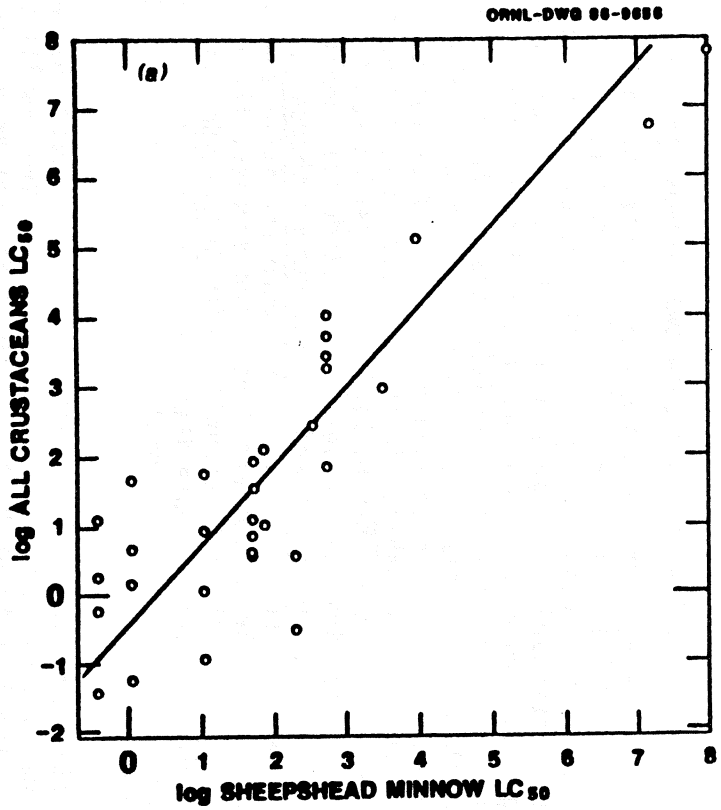
There are not enough MATCs for marine fishes, other than C. variegatus, to examine the representativeness of C. variegatus chronic responses for all Osteichthyes. However, analogy to the extrapolation of C. variegatus LC_{50}s (Table 4) and the extrapolation of Pimephales promelas MATCs to all freshwater Osteichthyes (PI = 1.12 with n = 51; Suter et al., 1986) suggest that the PI for MATCs for marine Osteichthyes would fall within ±1-1.5 orders of magnitude of C. variegatus MATCs.

Extrapolations from M. bahia to higher taxa are more uncertain (i.e., the PIs are larger) than those from C. variegatus. While the differences in PIs for the same Y taxon are not large, it is surprising that the responses of M. bahia are no more accurate than those of a fish in predicting the acute or chronic response of crustaceans as a group (Fig. 2). The crustaceans are highly diverse, and perhaps no single member can serve any better than a fish as a representative of the class.

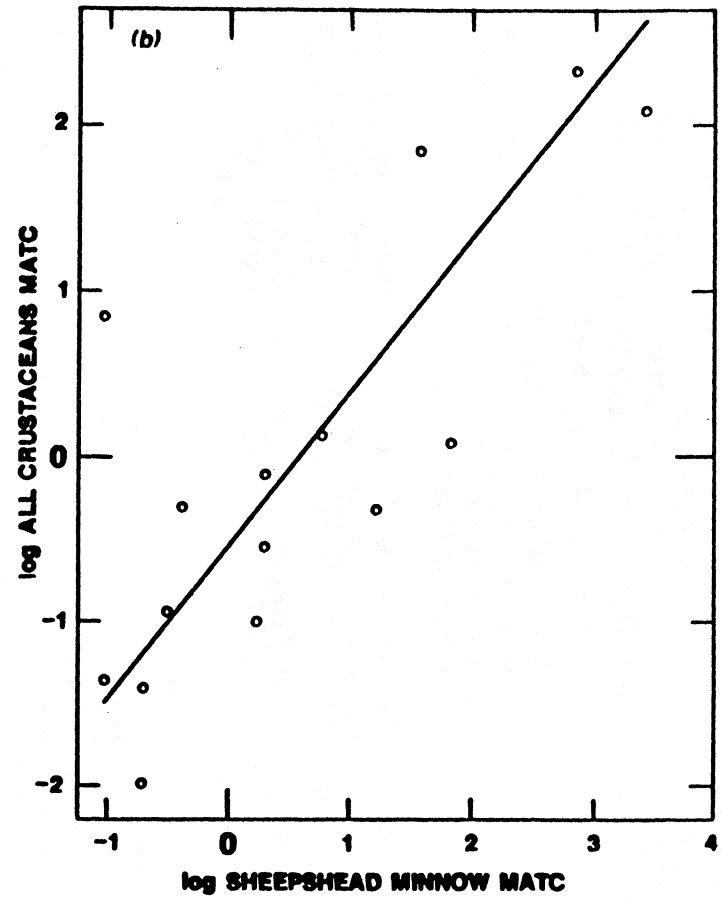
3.3 Relative Sensitivity

The geometric means of ratios of LC_{50}s show two shrimps, Penaeus duorarum and M. bahia, to be, on average, the most sensitive species of marine fishes or crustaceans in the AQUIRE data set (Table 5). While they were, on average, 13 and 42 times as sensitive, respectively, as other species, neither mean ratio was more than 1 standard error greater than 1 (i.e., equal average toxicity). The high standard errors were due to the 18 cases where a species was more than 3 orders of magnitude less sensitive than either M. bahia or P. duorarum, all of which involved fishes and all but one of which involved a pesticide. Other species were more sensitive in only 15% of cases for P. duorarum and 17% of cases for M. bahia. In the one case where a species was 2 orders of magnitude more sensitive than M. bahia, that species was P. duorarum. In the two cases where a species was 2 orders of magnitude more sensitive than P. duorarum, the species were M. bahia and Marinogammarus obtusatus. Thus, these species are generally, but not invariably, more sensitive than other species with which we are able to compare them. Two species, the cyprinid minnow Alburnus alburnus and the tetraodontid puffer Sphoeroides maculatus, have geometric mean ratios that are more than 2 standard errors less than 1, suggesting that they are relatively insensitive to chemicals. The other 17 species considered have ratios near 1.

It is considerably more difficult to draw conclusions about the relative sensitivity of marine species in chronic tests. However, the results tend to confirm the relative sensitivity of M. bahia. M. bahia was more sensitive than other crustaceans in three out of four cases. It was more sensitive than

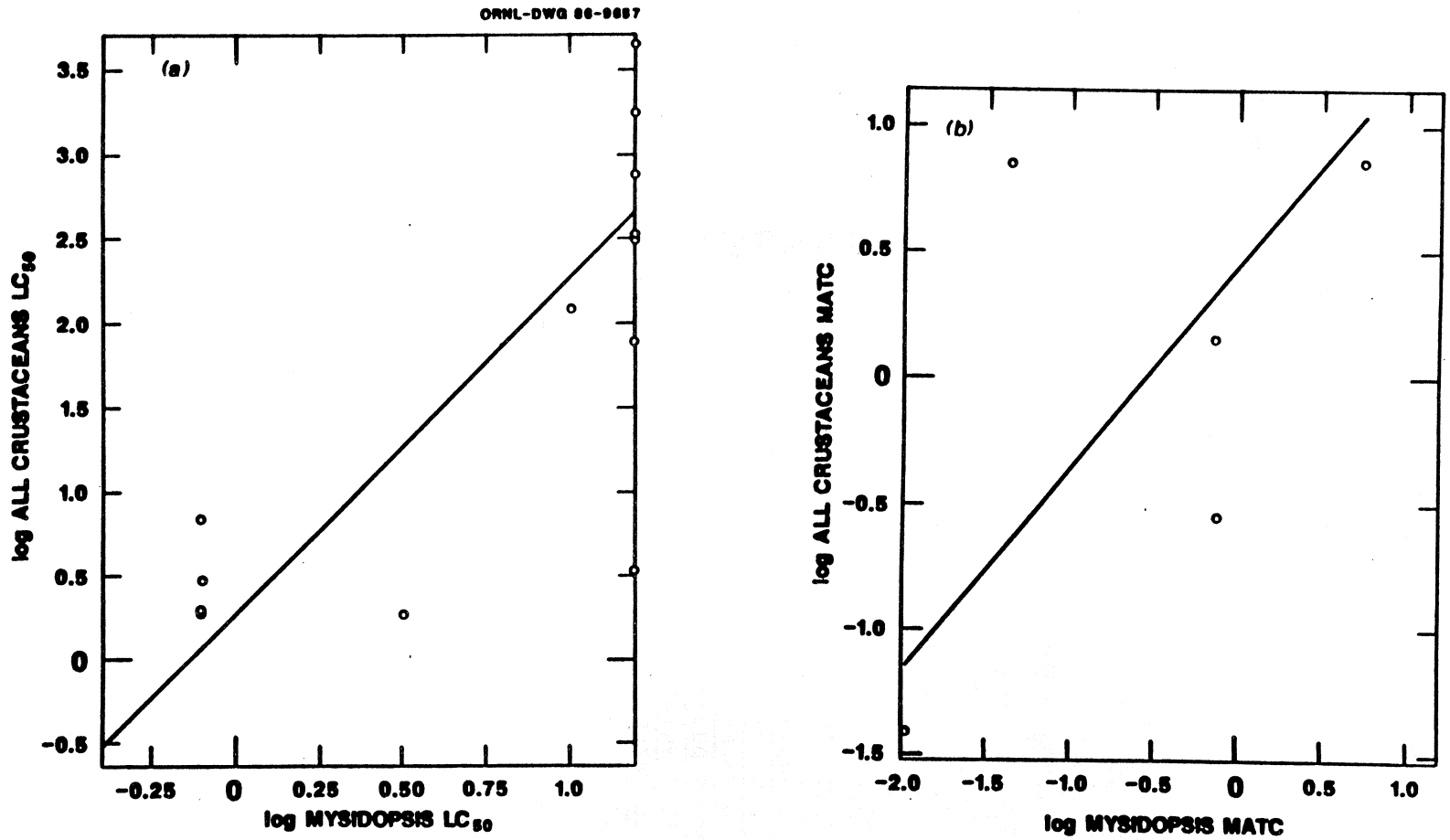


(a) Regression of LC_{50} for all crustacea against those for C. variegatus. Units are log (μg/L). n = 34; PI = 2.15.



(b) Regression of MATC values for all crustacea against those for M. variegatus. Units are log (μg/L). n = 15; PI = 1.80

Figure 1.



(a) Regression of LC_{50} values for all crustacea against those for M. bahia. Units are log (μg/L); n = 14; PI = 2.46.

(b) Regression of MATCs for all crustacea against those for M. bahia. Units are log (μg/L). n = 5; PI = 2.22.

Figure 2.

Table 5. Ratios of (LC_{50}s) of all marine fish and crustaceans to potential reference species

Reference species	n[a]	Geometric mean ratio	Standard error
Alburnus alburnus	53	0.07	0.10
Anguilla rostrata	149	1.06	0.71
Cancer magister	36	0.99	14.76
Crangon crangon	13	0.97	12.08
Crangon septemspinosa	162	6.37	247.32
Cyprinodon variegatus	85	0.86	1.07
Fundulus heteroclitus	167	0.39	0.38
Fundulus majalis	149	0.76	0.91
Gasterosteus aculeatus	139	0.42	17.40
Menidia beryllina	11	0.68	4.00
Menidia menidia	174	2.15	90.81
Morone saxatilis	23	1.09	0.79
Mugil cephalus	161	0.47	1.15
Mysidopsis bahia	30	41.97	3321.4
Nitocra spinipes	97	0.09	0.88
Pagurus longicar	154	1.01	74.25
Palaemonetes pugio	92	0.68	45.36
Palaemonetes vulgaris	162	1.83	164.75
Penaeus duorarum	81	13.51	516.35
Sphoeroides maculatus	149	0.10	0.06
Thalassoma bifas	149	1.35	2.83

[a] n = number of ratios.

C. variegatus in 10 out of 13 cases, less sensitive once (cyanide), and the comparison was inconclusive twice. C. variegatus was never more sensitive than other fishes; however, all but one of the matches with four other fish species were for a single chemical, chlorpyrifos. The geometric mean ratio of the MATCs for all other marine species to the MATCS for M. bahia is 7, but this value is less than 1 standard error greater than 1.

3.4 Acute-Chronic Extrapolations

MATCs for fishes and crustaceans were regressed against LC_{50} values for the same species determined in the same study (Fig. 3). The data are given in Table 1, and the results are presented in Table 6. The PI for all marine fishes and all chemicals (1.27) is better than the analogous regression for freshwater fishes (PI = 1.53; Suter et al., 1986), and the PI for all marine crustaceans and all chemicals (0.90, which is based on 90% M. bahia data) is better than the analogous regression for Daphnia (PI = 1.35; Suter et al., 1986).

3.5 Freshwater-Marine Extrapolations

Because of the relatively small number of marine MATCs available, it would be useful to extrapolate from freshwater to saltwater MATCs. The results of such extrapolations for the standard marine and freshwater fishes and crustaceans are presented in Table 7 and in Fig. 4. Since Pimephales promelas and C. variegatus are in different orders of Osteichthyes and Daphnia and M. bahia are in different orders of Crustacea, these extrapolations are between distantly related organisms as well as between major habitat types. The PI for the P. promelas to C. variegatus extrapolation (1.58) is a little worse than the PIs for extrapolating LC_{50}s between orders of fish within water types (Table 3). The PI for the Daphnia to M. bahia extrapolation (1.70) falls between the mean PI for extrapolating LC_{50}s between orders of marine crustaceans and those for extrapolating between orders of freshwater arthropods (Table 3). The slopes and intercepts for these two regressions are very close to 1 and 0, indicating that the members of both species pairs have very similar average sensitivities.

These results are somewhat surprising, since salinity affects the toxicity of some chemicals, particularly metals (Engel and Fowler, 1979). However, there is no sign that metals are particularly difficult to extrapolate between freshwater and saltwater. While there are no MATCs for common metals for the two fishes, more than half of the MATC pairs for crustaceans are for metals, and they cluster around the slope 1, intercept 0 line (Fig. 4b). The results are consistent with the finding of Klapow and Lewis (1979) that LC_{50}s for freshwater and saltwater species have indistinguishable distributions for nine metals and four other chemicals. They are also consistent with the finding of LeBlanc (1984) that the LC_{50}s for C. variegatus and M. bahia are significantly correlated with those for P. promelas and D. magna, respectively, and with the finding of Zaroogian et al. (1985) that structure-activity models developed for P. promelas and Poecilia reticulata (guppy) can be used to predict LC_{50} values for C. variegatus and M. bahia.

Table 6. Extrapolation from 96-h LC_{50}s to MATCs for marine fishes and crustaceans

Taxon	n[a]	PI[b]	λ	Slope	Intercept	F_1	F_2	$\bar{X}$	$\bar{Y}$
Osteichthyes	41	1.27	1.4	0.98	-0.60	0.42	0.004	1.80	1.16
Crustacea	43	0.90	1.4	1.00	-0.88	0.21	0.003	1.58	0.70

[a] n = number of points, each consisting of an LC_{50} and an MATC for the same species and chemical in the same laboratory.

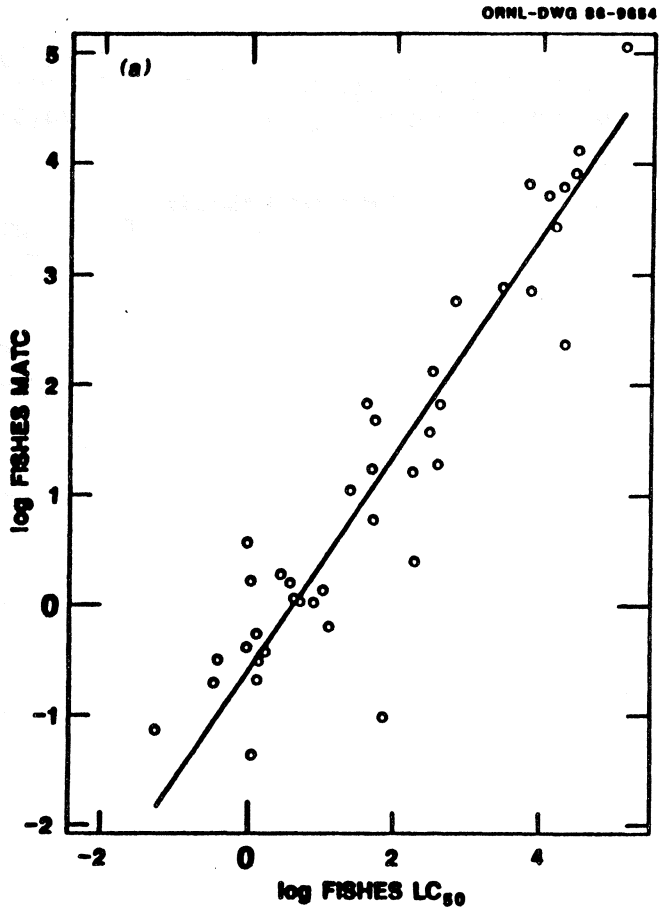
[b] Prediction interval (α = 0.05) at the mean ($\bar{X}$) is $\bar{Y} \pm$ PI.

Table 7. Freshwater to marine extrapolations

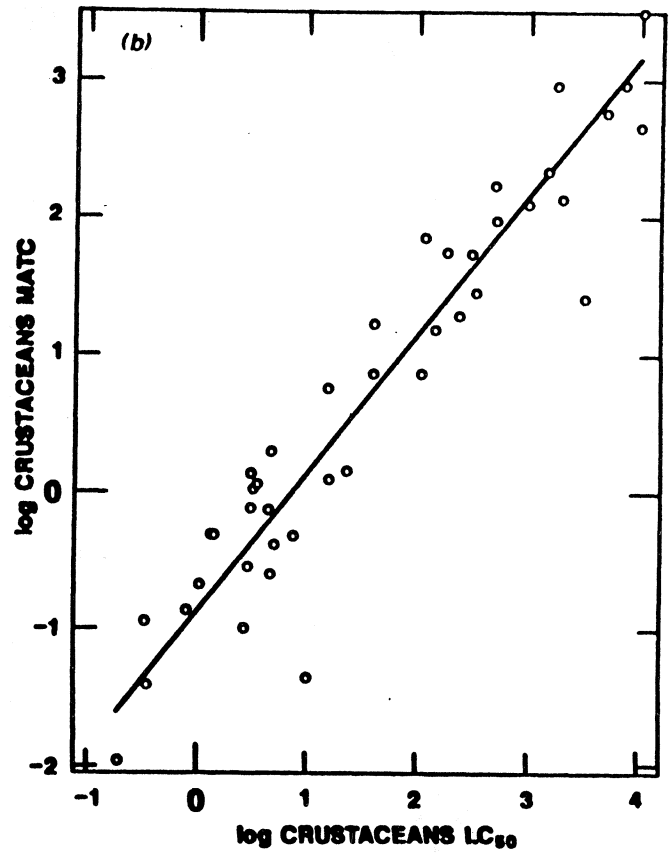
Species X	Taxon Y	Benchmark	n[a]	PI[b]	Slope	$Intercept_1$	F_2	F_2	$\bar{X}$	$\bar{Y}$
P. promelas	*C. variegatus*	MATC	16	1.58	0.99	-0.02	0.65	0.01	1.10	1.07
D. magna	*M. bahia*	MATC	15	1.70	1.01	-0.06	0.76	0.04	1.38	1.33
	Marine crustaceans	MATC	17	1.90	0.95	0.00	0.94	0.04	1.31	1.24

[a] n = the number of points, each consisting of an MATC for species X and an MATC for species Y for the same chemical.

[b] Prediction interval (α = 0.05) at the mean ($\bar{X}$) is $\bar{Y} \pm$ PI.

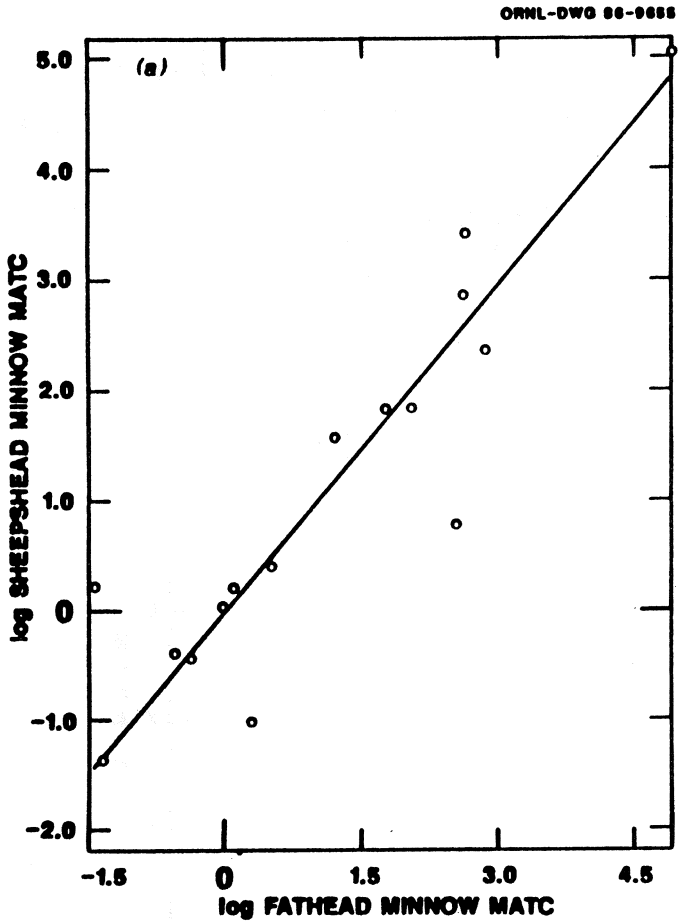


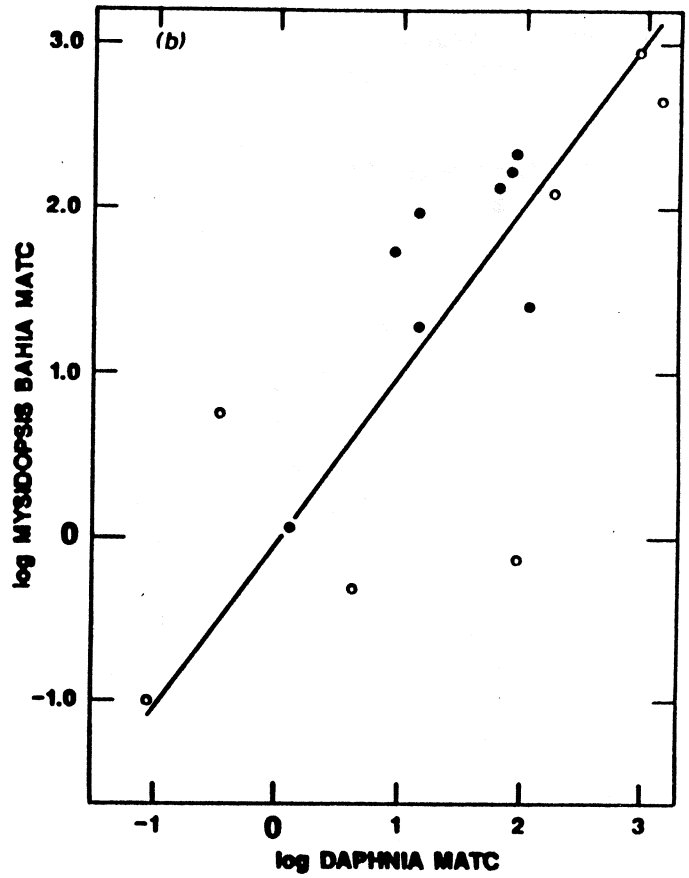
(a) Regression of MATC values against LC_{50} values for fish. Units are log (μg/L); n = 41; PI = 1.27.



(b) Regression of MATC values against LC_{50} values for marine crustaceans. Units are log (μg/L); n = 43; PI = 0.90.

Figure 3.





(a) Regression of MATCs for C. variegatus against those for P. promelas. Units are log (μg/L); n = 16; PI = 1.58.

(b) Regression of MATCs for M. bahia against those for Daphnia for the same chemical. Points marked x are metals; those marked o are nonmetals. Units are log (μg/L); n = 15; PI = 1.70.

Figure 4.

4. DISCUSSION

While the statistics of the regression lines presented above can be used to indicate relative sensitivity, they should not be used simply to generate point predictors. Since there is no such organism as the mean fish or mean shrimp, the variance about the line cannot be treated as measurement error about a true value. Rather the variance about the line should be used to determine the probability that the LC_{50} or MATC for an untested species from the specified taxon will be less than a particular ambient concentration, given a test result for a standard test species.

For example, if a chemical has an LC_{50} of 17.8 µg/L (1.25 log µg/L) for C. variegatus, then from Table 4 we can see that, given a two-tailed 95% PI of 1.24 ± 1.49 at X = 1.25, 97.5% of marine Osteichthyes would be expected to have an LC_{50} greater than 0.56 µg/L (-0.25 log µg/L). In other words, the risk that the median lethal concentration for any particular fish species would be less than 0.56 is 0.025. Calculations of probabilities for other predicted test results or for particular ambient concentrations are performed analogously; for detailed instructions, see Suter et al. (1986).

The analyses of relative sensitivity confirm prior results for freshwater species, indicating that no species is consistently the most sensitive (Kenaga, 1978). Since there is no most sensitive species, some correction must always be made when the results of a single species test is used to determine a safe or acceptable concentration for a taxon or biota. These corrections can be based on the risks of not protecting the species of interest or of affecting more than x% of members of a taxon calculated from the variances on these regressions.

Alternatively, the results of the analyses of relative sensitivity can be used as a guide for planning monitoring studies or interpreting their results. While no species can be counted on to respond first to pollution, monitoring efforts would be best concentrated on the species with the greatest average sensitivity. Of the taxa considered, penaeid shrimp and mysid shrimp appear to have the highest average sensitivity, and crustaceans appear to be more sensitive than fish.

Ideally the array of species tested would have been randomly chosen from the higher taxa in which they occur. This is clearly not the case: test species are chosen for their ability to be maintained in the laboratory for at least the duration of the test, for their availability to the investigators, and in some cases for their relevance to a particular pollution problem, but not by some random process. However, they have not, in general, been chosen for their sensitivity because relative sensitivities are only now becoming apparent. Therefore, we do not believe for example, that, the nine atherinid fish in the AQUIRE data set are a biased sample of the Atherinidae. What is problematical is the ability of the tested species to represent marine fishes and crustaceans in general. Many important marine taxa such as the Clupeiformes, Gadiformes, and Pleuronectiformes are clearly underrepresented, and some, such as the Scombridae, will never lend themselves to testing. Therefore, it is possible, and even likely, that highly sensitive marine species exist that do not appear in this analysis.

The other aspect of representativeness of the data sets is the assumption that the tested chemicals are representative of those that will be important in the field. This assumption is difficult to verify, and we know that the mode of action of a chemical can affect the extrapolations (Suter et al., 1986). This assumption can be minimized by partitioning the data sets by chemical class, but in most cases the data sets are not large enough to tolerate partitioning. That being the case, we must assume that investigators had a reasonable grasp of the relative importance of pollutants when they chose their test chemicals or that the chemicals chosen for testing are at least a reasonable assortment.

5. CONCLUSIONS

The major conclusion of this study is that data extrapolations are required to assess the effects of almost all chemicals on any marine fish or crustacean because of the large number of species and chemicals to be assessed. These extrapolations must often be made at very high taxonomic levels, and even when they can be made between closely related species, nontrivial uncertainties are involved. In general, toxic effects can be predicted with $\pm$1-2 order of magnitude precision. These uncertainties are comparable to those associated with extrapolations of freshwater data (Suter et al., 1986). The magnitude of extrapolation uncertainties has often been unappreciated because of the use of inappropriate statistics, small data sets, or uninformative measures such as correlation coefficients. The practical significance of these uncertainties depends on the individual assessment. Given the extremely wide range of concentrations at which chemicals occur in the environment, most environmental concentrations can be expected to fall outside the prediction intervals for most predicted-effects concentrations. When environmental concentrations do fall within the uncertainty bounds of a predicted-effects concentration, that uncertainty can be considered to constitute a risk, that is, a probability of an undesired effect (Suter et al., 1983). The uncertainty and associated risk can be decreased by additional testing.

Another conclusion that may be drawn from this study is that additional, more systematic testing needs to be done to determine the relative sensitivity of marine organisms to pollutant chemicals. The existing data, particularly that for chronic toxicity, cannot be said to be representative of the marine biota. Particular emphasis must be placed on invertebrates because of their great diversity. A testing program that examines the responses of a taxonomic array of species to a set of chemicals with diverse modes of action could greatly improve the confidence with which data from standard test species can be used in assessments.

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