



Ambient Aquatic Life Water Quality Criteria for Tributyltin (TBT) - Final



AMBIENT AQUATIC LIFE WATER QUALITY CRITERIA FOR
TRIBUTYLTIN
CAS Registry Number (See Text)

December 2003

U.S. ENVIRONMENTAL PROTECTION AGENCY
OFFICE OF WATER
OFFICE OF SCIENCE AND TECHNOLOGY
HEALTH AND ECOLOGICAL CRITERIA DIVISION
WASHINGTON D.C.

NOTICES

This document has been reviewed by the Health and Ecological Criteria Division, Office of Science and Technology, U.S. Environmental Protection Agency, and is approved for publication.

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

This document is available to the public through the National Technical Information Service (NTIS), 5285 Port Royal Road, Springfield, VA 22161. It is also available on EPA's web site: <http://www.epa.gov/waterscience/criteria/tributyltin>.

FOREWORD

Section 304(a)(1) of the Clean Water Act of 1977 (P.L. 95-217) requires the Administrator of the Environmental Protection Agency to publish water quality criteria that accurately reflect the latest scientific knowledge on the kind and extent of all identifiable effects on health and welfare that might be expected from the presence of pollutants in any body of water, including ground water. This final document is a revision of proposed criteria based upon consideration of scientific input received from U.S. EPA staff, the public and independent peer reviewers. Criteria contained in this document replace any previously published EPA aquatic life criteria for tributyltin (TBT).

The term "water quality criteria" is used in two sections of the Clean Water Act, section 304(a)(1) and section 303(c)(2). The term has a different program impact in each section. In section 304, the term represents a non-regulatory, scientific assessment of ecological effects. Criteria presented in this document are such scientific assessments. If water quality criteria associated with specific stream uses are adopted by a state as water quality standards under section 303, they become enforceable maximum acceptable pollutant concentrations in ambient waters within that state. Water quality criteria adopted in state water quality standards could have the same numerical values as criteria developed under section 304. However, in many situations states might want to adjust water quality criteria developed under section 304 to reflect local environmental conditions and human exposure patterns. Alternatively, states may use different data and assumptions than EPA in deriving numeric criteria that are scientifically defensible and protective of designated uses. It is not until their adoption as part of state water quality standards that criteria become regulatory. Guidelines to assist the states and Indian tribes in modifying the criteria presented in this document are contained in the Water Quality Standards Handbook (U.S. EPA, 1994). This handbook and additional guidance on the development of water quality standards and other water-related programs of this Agency have been developed by the Office of Water.

This final document is guidance only. It does not establish or affect legal rights or obligations. It does not establish a binding norm and cannot be finally determinative of the issues addressed. Agency decisions in any particular situation will be made by applying the Clean Water Act and EPA regulations on the basis of specific facts presented and scientific information then available.

Geoffrey H. Grubbs
Director
Office of Science and Technology

EXECUTIVE SUMMARY

BACKGROUND:

Tributyltin (TBT) is a highly toxic biocide that has been used extensively to protect the hulls of large ships. It is a problem in the aquatic environment because it is extremely toxic to non-target organisms, is linked to imposex and immuno-suppression in snails and bivalves, and can be persistent. EPA is developing ambient water quality criteria for TBT through its authority under Section 304(a) of the Clean Water Act (CWA). These water quality criteria may be used by States and authorized Tribes to establish water quality standards for TBT.

CRITERIA:

Freshwater:

For TBT, the criterion to protect freshwater aquatic life from chronic toxic effects is 0.072 µg/L. This criterion is implemented as a four-day average, not to be exceeded more than once every three years on the average. The criterion to protect freshwater aquatic life from acute toxic effects is 0.46 µg/L. This criterion is implemented as a one-hour average, not to be exceeded more than once every three years on the average.

Saltwater:

For TBT, the criterion to protect saltwater aquatic life from chronic toxic effects is 0.0074 µg/L. This criterion is implemented as a four-day average, not to be exceeded more than once every three years on the average. The criterion to protect saltwater aquatic life from acute toxic effects is 0.42 µg/L. This criterion is implemented as a one-hour average, not to be exceeded more than once every three years on the average.

The saltwater chronic criterion for TBT differs from the criterion that was originally proposed for public review (0.010 µg/L). The development of the saltwater chronic criterion for TBT considers four lines of evidence:

- (1) the traditional endpoints of adverse effects on survival, growth, and reproduction as demonstrated in numerous laboratory studies;
- (2) the endocrine disrupting capability of TBT as observed in the production of imposex in field studies;
- (3) that TBT bioaccumulates in commercially and recreationally important freshwater and saltwater species; and
- (4) that an important commercial organism already known to be vulnerable to a prevalent pathogen was made even more vulnerable by prior exposure to TBT.

For these reasons, the criterion to protect saltwater aquatic life from chronic toxic effects is set at 0.0074 µg/L.

This document provides guidance to States and Tribes authorized to establish water quality standards under the Clean Water Act (CWA) to protect aquatic life from acute and chronic effects of TBT. Under the CWA, States and Tribes are to establish water quality criteria to protect designated uses. While this document constitutes U.S. EPA's scientific recommendations regarding ambient concentrations of TBT, this document does not substitute for the CWA or U.S. EPA's regulations; nor is it a regulation itself. Thus, it cannot impose legally binding requirements on U.S. EPA, States, Tribes, or the regulated community, and it might not apply to a particular situation based upon the circumstances. State and Tribal decision-makers retain the discretion to adopt approaches on a case-by-case basis that differ from this guidance when appropriate. U.S. EPA may change this guidance in the future.

ACKNOWLEDGMENTS

Larry T. Brooke (freshwater)
University of Wisconsin-Superior
Superior, Wisconsin and Great
Lakes Environmental Center,
Traverse City, Michigan

David J. Hansen (saltwater)
U.S. EPA, Atlantic Ecology Division
Narragansett, Rhode Island and
Great Lakes Environmental Center,
Traverse City, Michigan

Gregory J. Smith
Great Lakes Environmental Center
Columbus, Ohio

Frank Gostomski
(document coordinator)
U.S. EPA
Health and Ecological Criteria Division
Office of Water
Washington, D.C.

TECHNICAL ASSISTANCE AND PEER REVIEW

Herbert E. Allen
University of Delaware
Newark, Delaware

Rick D. Cardwell
Parametrix, Inc.,
Redmond, Washington

Peter M. Chapman
EVS Environmental Consultants
North Vancouver, British Columbia

Lenwood W. Hall
University of Maryland
Queenstown, Maryland

Michael H. Salazar
Applied Biomonitoring
Kirkland, Washington

Peter F. Seligman
U.S. Navy SSC SD
San Diego, California

Robert L. Spehar
U.S. EPA
Mid-Continent Ecology Division
Duluth, Minnesota

Glen Thursby
U.S. EPA
Atlantic Ecology Division
Narragansett, Rhode Island

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INTRODUCTION¹

Organotins are compounds consisting of one to four organic components attached to a tin atom via carbon-tin covalent bonds. When there are fewer than four carbon-tin bonds, the organotin cation can combine with an anion such as acetate, carbonate, chloride, fluoride, hydroxide, oxide, or sulfide. Thus a species such as tributyltin (TBT) is a cation whose formula is $(C_4H_9)_3Sn^+$. In sea water, TBT exists mainly as a mixture of the chloride, the hydroxide, the aquo complex, and the carbonate complex (Laughlin et al. 1986a).

The principal use of organotins is as a stabilizer in the manufacturing of plastic products, for example, as an anti-yellowing agent in clear plastics and as a catalyst in poly(vinyl chloride) products (Piver 1973). Another and less extensive use of organotins is as a biocide (fungicide, bactericide, insecticide) and as a preservative for wood, textiles, paper, leather and electrical equipment. Total world-wide production of organotin compounds is estimated at 50,000 tons per year with between 15 and 20% of the production used in the biologically active triorganotins (Bennett 1996).

A large market exists for organotins in antifouling paint for the wet bottom of ship hulls. The most common organometallics used in these paints are TBT oxide and TBT methacrylate. Protection from fouling with these paints lasts more than two years and is superior to copper- and mercury-based paints. These paints have an additional advantage over other antifouling paints, such as copper sulfate based paint, by not promoting bimetallic corrosion. The earliest paints containing TBT were “free association” paints that contained a free suspension of TBT and caused high concentrations of TBT to be leached to the aquatic environment when the paint application was new. A later refinement was the “ablative” paint that shed the outer layer when in contact with water but at a slower rate than the free association paint. Further development of organometallic antifouling paints have been in the production of paints containing copolymers that control the release of the organotins and result in longer useful life of the paint as an antifoulant (Bennett 1996; Champ and Seligman 1996; Kirk-Othmer 1981). The U.S. Navy (1984) proposed application of some paints containing TBT to hulls of naval ships. Such paint formulations have been shown to be an effective and relatively long-lived deterrent to adhesion of barnacles and other fouling organisms. Encrustations on ships' hulls by these organisms reduce maximum speed and increase fuel consumption. According to the U.S. Navy (1984), use of

¹A comprehension of the “Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses” (Stephen et al. 1985), hereafter referred to as the Guidelines, is necessary to understand the following text, tables and calculations.

TBT paints, relative to other antifouling paints, would not only reduce fuel consumption by 15% but would also increase time between repainting from less than 5 years to 5 to 7 years. Interaction between the toxicities of TBT and other ingredients in the paint apparently is negligible, but needs further study (Davidson et al. 1986a). The use of TBT in antifouling paints on ships, boats, nets, crab pots, docks, and water cooling towers probably contributes most to direct release of organotins into the aquatic environment (Clark et al. 1988; Hall and Pinkney 1985; Kinnetic Laboratory 1984).

A non-toxic alternative to TBT is the non-stick paint system available from several of the major paint manufacturers. When the paint is applied properly, the ship's hull becomes too smooth for algae, barnacles and other marine organisms to attach themselves to the surface (Greenpeace 1999)

The solubility of TBT compounds in water is influenced by such factors as the oxidation-reduction potential, pH, temperature, ionic strength, and concentration and composition of the dissolved organic matter (Clark et al. 1988; Corbin 1976). The solubility of tributyltin oxide in water was reported to be 750 $\mu\text{g/L}$ at pH of 6.6, 31,000 $\mu\text{g/L}$ at pH of 8.1, and 30,000 $\mu\text{g/L}$ at pH 2.6 (Maguire et al. 1983). The carbon-tin covalent bond does not hydrolyze in water (Maguire et al. 1983,1984), and the half-life for photolysis due to sunlight is greater than 89 days (Maguire et al. 1985; Seligman et al. 1986). Biodegradation is the major breakdown pathway for TBT in water and sediments with half-lives of several days to weeks in water, and from several days to months or more than a year in sediments (Clark et al. 1988; de Mora et al. 1989; Lee et al. 1987; Maguire 2000; Maguire and Tkacz 1985; Seligman et al. 1986, 1988, 1989; Stang and Seligman 1986; Stang et al. 1992). Breakdown products include dibutyltins (DBT), monobutyltins (MBT) and tin with some methyltins detected when sulfate reducing conditions were present (Yonezawa et al. 1994). Porous sediments with aerobic conditions decrease degradation time (Watanabe et al. 1995).

Several review papers have been written which cover the production, use, chemistry, toxicity, fate and hazards of TBT in the aquatic environment (Alzieu 1996; Batley 1996; Clark et al. 1988; Eisler 1989; Gibbs and Bryan 1996b; Hall and Bushong 1996; Laughlin 1996; Laughlin et al. 1996; Maguire 1996; Waldock et al. 1996; WHO 1990). The toxicities of organotin compounds are related to the number of organic components bonded to the tin atom and to the number of carbon atoms in the organic components. Toxicity to aquatic organisms generally increases as the number of organic components increases from one to three and decreases with the incorporation of a fourth, making triorganotins more toxic than other forms. Within the triorganotins, toxicity increases as the number of carbon atoms in the organic moiety increases from one to four, then decreases. Thus the organotin most toxic to aquatic life is TBT (Hall and Pinkney 1985; Laughlin and Linden 1985; Laughlin et al. 1985). TBTs inhibit Na^+ and K^+ ATPases and are ionophores controlling exchange of Cl^- , Br^- , F^- and

other ions across cell membranes (Selwyn 1976).

Metabolism of TBT has been studied in several species. Some species of algae, bacteria, and fungi have been shown to degrade TBT by sequential dealkylation, resulting in dibutyltin, then monobutyltin, and finally inorganic tin (Barug 1981; Maguire et al. 1984). Barug (1981) observed the biodegradation of TBT to di- and monobutyltin by bacteria and fungi only under aerobic conditions and only when a secondary carbon source was supplied. Inorganic tin can be methylated and demethylated by estuarine microorganisms (Jackson et al. 1982). Maguire et al. (1984) reported that a 28-day culture of the green alga, *Ankistrodesmus falcatus*, with TBT resulted in 7% inorganic tin. Maguire (1986) reported that the half-life of TBT exposed to microbial degradation was five months under aerobic conditions and 1.5 months under anaerobic conditions. TBT is also accumulated and metabolized by an eel grass, *Zostera marina* (Francios et al. 1989). Chiles et al. (1989) found that much of the TBT accumulated on the surface of saltwater algae and bacteria as well as within the cell. The major metabolite of TBT in saltwater crabs, fish, and shrimp was dibutyltin (Lee 1985, 1986). A review of the metabolism of TBT by marine aquatic organisms has been provided by Lee (1996).

TBT is an endocrine-disrupting chemical (Matthiessen and Gibbs 1998). The chemical causes masculinization of certain female gastropods. It is likely the best studied example of endocrine-disrupting effect. The metabolic mechanism is thought to be due to elevating testosterone titers in the animals and over-riding the effects of estrogen. There are several theories of how TBT accomplishes the buildup of testosterone. Evidence suggests that competitive inhibition of cytochrome P450-dependent aromatase is probably occurring in TBT exposed gastropods (Matthiessen and Gibbs 1998). TBT may also interfere with sulfur conjugation of testosterone and its phase I metabolites and their excretion resulting in a build-up of pharmacologically active androgens in some animal tissues (Ronis and Mason 1996).

TBT has been measured in the water column and found highly (70-90%) associated with the dissolved phase (Johnson et al. 1987; Maguire 1986; Valkirs et al. 1986a), but TBT readily adsorbs to sediments and suspended solids where it may persist (Cardarelli and Evans 1980; Harris et al. 1996; Seligman et al. 1996). TBT accumulates in sediments with adsorption coefficients which range from 1.1×10^2 to 8.2×10^3 L/Kg; desorption appears to be a two step process (Unger et al. 1987, 1988). Langston and Pope (1995) found that at environmentally realistic concentrations of 10 ng/L, TBT partitioning coefficients were closer to 2.5×10^4 .

In a modeling and risk assessment study of TBT in a freshwater lake, Traas et al. (1996) predicted that TBT concentrations in the water and suspended matter would decrease rapidly and TBT concentrations in sediment and benthic organisms would decrease at a much slower rate. Dowson et al.

(1996) measured the half-life of TBT in the top 5 cm of aerated freshwater and estuarine sediments. They measured half-lives of 360 days at 450 ng/g TBT to 775 days at 1300 ng/g TBT. No degradation of TBT occurred during 330 days in anaerobic sediments.

A more rapid biotic and abiotic degradation of TBT was reported by Stang et al. (1992) in fine-grained sediments at the sediment-water interface. Sediments in these studies were observed to degrade 50 percent or more of the TBT to DBT and MBT in both sterile and unsterilized sediments in a period of days, suggesting both chemical and biological degradation processes. The authors speculated that it is likely that much of the residual TBT measured in harbor and estuarine sediments is either in particulate form from paint chips, making it less available, or has been mixed into the sediment in anoxic layers with reduced degradation.

The water surface microlayer contains a much higher concentration of TBT than the water column (Cleary and Stebbing 1987; Hall et al. 1986; Maguire 1986; Valkirs et al. 1986a). Gucinski (1986) suggested that this enrichment of the surface microlayer could increase the bioavailability of TBT to organisms in contact with this layer.

Elevated TBT concentrations in fresh and salt waters, sediments, and biota are primarily associated with harbors and marinas (Cleary and Stebbing 1985; Espourteille et al. 1993; Gibbs and Bryan 1996a; Grovhoug et al. 1996; Hall 1988; Hall et al. 1986; Langston et al. 1987; Maguire 1984,1986; Maguire and Tkacz 1985; Maguire et al. 1982; Minchin and Minchin 1997; Peven et al. 1996; Prouse and Ellis 1997; Quevauviller et al. 1989; Salazar and Salazar 1985b; Seligman et al. 1986,1989; Short and Sharp 1989; Stallard et al. 1986; Stang and Seligman 1986; Unger et al. 1986; Valkirs et al. 1986b; Waite et al. 1996; Waldock and Miller 1983; Waldock et al. 1987). Several studies have been conducted in harbors to measure the effects of TBT on biota. Lenihan et al. (1990) hypothesized that changes in faunal composition in hard bottom communities in San Diego Bay were related to boat mooring and TBT. Salazar and Salazar (1988) found an apparent relationship between concentrations of TBT in waters of San Diego Bay and reduced growth of mussels. No organotins were detected in the muscle tissue of feral chinook salmon, *Oncorhynchus tshawytscha*, caught near Auke Bay, Alaska, but concentrations as high as 900 µg/kg were reported in muscle tissue of chinook salmon held in shallow-water pens treated with TBT (Short 1987; Short and Thrower 1986a). Organotin concentrations in European coastal waters in the low part per trillion range have been associated with oyster shell malformations (Alzieu et al. 1989; Minchin et al. 1987).

Reevaluation of harbors in the United Kingdom revealed that since the 1987 restrictions which banned the retail sale and use of TBT paints for small boats or mariculture purposes, oyster culture has returned in the harbor areas where boat traffic is low and water exchange is good (Dyrynda 1992;

Evans et al. 1996; Minchin et al. 1996,1997; Page and Widdows 1991; Waite et al. 1991). Tissue concentrations of TBT in oysters have decreased in most of the sites sampled in the Gulf of Mexico since the introduction of restrictions (1988-1989) on its use (Wade et al. 1991). Canada restricted the use of TBT-containing boat-hull paints in 1989, and there has been a reduction in female snail reproductive deformities (imposex) in many Canadian west coast sampling sites since the action (Tester et al. 1996). In a four-year (1987-1990) monitoring study for butyltins in mussel tissue on the two U.S. coasts, a general decrease in tissue concentrations was measured on the west coast, and east coast sites showed mixed responses (Uhler et al. 1989,1993). Some small ports in France have not seen a decline in imposex since the ban on TBT in boat hull paints (Huet et al. 1996). Suspicions are that the legislation banning the paints is being ignored. Several freshwater ecosystems were studied since the ban on antifouling paints in Switzerland in 1988. By 1993 TBT concentrations were decreasing in the water, but declines were not seen in the sediment or in the zebra mussel, *Dreissena polymorpha* (Becker-van Slooten and Tarradellas 1995; Fent and Hunn 1995).

Because of the assumption that certain anions do not contribute to TBT toxicity, only data generated in toxicity and bioconcentration tests on TBTCI (tributyltin chloride; CAS 1461-22-9), TBTF (tributyltin fluoride; CAS 1983-10-4), TBTO [bis(tributyltin) oxide; CAS 56-35-9], commonly called "tributyltin oxide" and TBTS [bis(tributyltin) sulfide; CAS 4808-30-4], commonly called "tributyltin sulfide" were used in the derivation of the water quality criteria concentrations for aquatic life presented herein. All concentrations from such tests are expressed as TBT, not as tin and not as the chemical tested. The conversion factors are 0.8911 for TBTCI, 0.9385 for TBTF, 0.9477 for TBTO, 0.9005 for TBTS, and 2.444 for Tin (Sn). Therefore, many concentrations listed herein are not those in the reference cited but are concentrations adjusted to TBT. A comprehension of the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" (Stephan et al. 1985), here after referred to as the Guidelines, and the response to public comment (U.S. EPA 1985a) is necessary to understand the following text, tables, and calculations. Results of such intermediate calculations as recalculated LC50s and Species Mean Acute Values are given to four significant figures to prevent roundoff error in subsequent calculations, not to reflect the precision of the value. The Guidelines require that all available pertinent laboratory and field information be used to derive a criterion consistent with sound scientific evidence. The saltwater criterion for TBT follows this requirement by using data from chronic exposures of copepods and molluscs rather than Final Acute Values and Acute-Chronic Ratios to derive the Final Chronic Value. The Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA) data base of information from the pesticide industry was searched and some useful information was located for deriving the criteria. The

latest comprehensive literature search for information for this document was conducted in January 1997 for fresh- and saltwater organisms. Some more recent data have been included in the document.

ACUTE TOXICITY TO AQUATIC ANIMALS

Data that may be used, according to the Guidelines, in the derivation of Final Acute Values for TBT are presented in Table 1. Acute values are available for thirteen freshwater species representing twelve genera. For freshwater Species Mean Acute Values, 23% were <2.0 µg/L, 38% were <4.0 µg/L, 69% were <8.0 µg/L, and 92% were <12.73 µg/L. A freshwater clam, *Elliptio complanatus*, had an LC50 of 24,600 µg/L. The relatively low sensitivity of the freshwater clam to TBT is surprising due to the molluscicidal qualities of TBT. The organism likely closes itself to the environment, minimizing chemical intake, and is able to temporarily tolerate high concentrations of TBT.

The most sensitive freshwater organisms tested are from the phylum Coelenterata (Table 3). Three species of hydras were tested and have Species Mean Acute Values (SMAVs) ranging from 1.14 to 1.80 µg/L. Other invertebrate species tested in flow-through measured tests include an amphipod, *Gammarus pseudolimnaeus*, and an annelid, *Lumbriculus variegatus*, and in a static measured test, larvae of a mosquito, *Culex* sp (Brooke et al. 1986). The 96-hr LC50s and SMAVs are 3.7, 5.4 and 10.2 µg/L, respectively. Six tests were conducted with the cladoceran, *Daphnia magna*. The 48-hr EC50 value of 66.3 µg/L (Foster 1981) was considerably less sensitive than those from the other tests which ranged from 1.58 µg/L (LeBlanc 1976) to 18 µg/L (Crisinel et al. 1994). The SMAV for *D. magna* is 4.3 µg/L because, according to the Guidelines, when test results are available from flow-through and concentration measured tests, these have precedence over other types of acute tests.

All the vertebrate species tested are fish. The most sensitive species is the fathead minnow, *Pimephales promelas*, which has a SMAV of 2.6 µg/L from a single 96-hr flow-through measured test (Brooke et al. 1986). Rainbow trout, *Oncorhynchus mykiss*, were tested by four groups with good agreement between them. The 96-hr LC50s ranged from 3.45 to 7.1 µg/L with a SMAV of 4.571 µg/L for the three tests (Brooke et al. 1986; Martin et al. 1989; ABC Laboratories, Inc. 1990a), which were conducted using flow-through conditions and measured concentrations. Juvenile catfish, *Ictalurus punctatus*, were exposed to TBT in a flow-through and measured concentration test and resulted in a 96-hr LC50 of 5.5 µg/L which is in good agreement with the other tested freshwater fish species (Brooke et al. 1986). Bluegill, *Lepomis macrochirus*, were tested by three groups. The value of 227.4 µg/L (Foster 1981) appears high compared to those of 7.2 µg/L (Buccafusco 1976b) and 8.3 µg/L

(ABC Laboratories, Inc. 1990b). Only the flow-through measured test (ABC Laboratories, Inc. 1990b) can be used, according to the Guidelines, to calculate the SMAV of 8.3 µg/L.

Freshwater Genus Mean Acute Values (GMAVs) are available for twelve genera which vary by more than 21,000 times from the least sensitive to the most sensitive. However, eleven of the twelve genera differ from one another by a maximum factor of 11.2 times (Figure 1). Based upon the twelve available GMAVs the Final Acute Value (FAV) for freshwater organisms is 0.9177 µg/L. The FAV is lower than the lowest freshwater SMAV of 1.14 µg/L. The freshwater Criterion Maximum Concentration is 0.4589 µg/L which is calculated by dividing the FAV by two.

Tests of the acute toxicity of TBT to resident North American saltwater species that are useful for deriving water quality criteria concentrations have been performed with 26 species of invertebrates and seven species of fish (Table 1). The range of acute toxicity to saltwater animals is a factor of about 1,176. Acute values range from 0.24 µg/L for juveniles of the copepod, *Acartia tonsa* (Kusk and Petersen 1997) to 282.2 µg/L for adult Pacific oysters, *Crassostrea gigas* (Thain 1983). The 96-hr LC50s for seven saltwater fish species range from 1.460 µg/L for juvenile chinook salmon, *Oncorhynchus tshawytscha* (Short and Thrower 1986b, 1987) to 25.9 µg/L for subadult sheepshead minnows, *Cyprinodon variegatus* (Bushong et al. 1988).

Larval bivalve molluscs and juvenile crustaceans appear to be much more sensitive than adults during acute exposures. The 96-hr LC50 for larval Pacific oysters, *Crassostrea gigas*, was 1.557 µg/L, whereas the value for adults was 282.2 µg/L (Thain 1983). The 96-hr LC50s for larval and adult blue mussels, *Mytilus edulis*, were 2.238 and 36.98 µg/L, respectively (Thain 1983). The 96-hr LC50 of 0.01466 µg/L reported by Becerra-Huencho (1984) for post larvae of the hard clam, *M. mercenaria*, was not used because results of other studies with embryos, larvae, and post larvae of the hard clam where acutely lethal concentrations range from 0.6 to 4.0 µg/L (Tables 1 and 6) cast doubt on this LC50 value. Juveniles of the crustacean *Acanthomysis sculpta* were slightly more sensitive to TBT than adults (Davidson et al. 1986a, 1986b; Valkirs et al. 1985).

Genus Mean Acute Values for 30 saltwater genera range from 0.61 µg/L for *Acanthomysis* to 204.4 µg/L for *Ostrea* (Table 3). Genus Mean Acute Values for the 10 most sensitive genera differ by a factor of less than four (Figure 2). Included within these genera are four species of molluscs, six species of crustaceans, and one species of fish. The saltwater Final Acute Value (FAV) for TBT was calculated to be 0.8350 µg/L (Table 3), which is greater than the lowest saltwater Species Mean Acute Value of 0.61 µg/L. The saltwater Criterion Maximum Concentration is 0.4175 µg/L and is calculated by dividing the FAV by two.

CHRONIC TOXICITY TO AQUATIC ANIMALS

The available data that are usable, according to the Guidelines, concerning the chronic toxicity of TBT are presented in Table 2. Brooke et al. (1986) conducted a 21-day life-cycle test with a freshwater cladoceran and reported that the survival of adult *D. magna* was 40% at a TBT concentration of 0.5 µg/L, and 100% at 0.2 µg/L. The mean number of young per adult per reproductive day was reduced 30% by 0.2 µg/L, and was reduced only 6% by 0.1 µg/L. The chronic limits are 0.1 and 0.2 µg/L based upon reproductive effects on adult daphnids. The chronic value for *D. magna* is 0.1414 µg/L (geometric mean of the chronic limits), and the acute-chronic ratio of 30.41 is calculated using the acute value of 4.3 µg/L from the same study.

D. magna were exposed in a second 21-day life-cycle test to TBT (ABC Laboratories, Inc. 1990d). Exposure concentrations ranged from 0.12 to 1.27 µg/L as TBT. Survival of adults was significantly reduced (45%) from the controls at 0.34 µg/L but not at 0.19 µg/L. Mean number of young per adult per reproductive day was significantly reduced at the same concentrations affecting survival. The chronic limits are 0.19 µg/L where no effects were seen and 0.34 µg/L where survival and reproduction were reduced. The Chronic Value is 0.2542 µg/L and the Acute-Chronic Ratio is 44.06 when calculated from the acute value of 11.2 µg/L from the same test. The Acute-Chronic Ratio for *D. magna* is 36.60 which is the geometric mean of the two available Acute-Chronic ratios (30.41 and 44.06) for this species.

In an early life-stage test (32-day duration) with the fathead minnow, *Pimephales promelas*, all fish exposed to the highest exposure concentration of 2.20 µg/L died during the test (Brooke et al. 1986). Survival was not reduced at 0.92 µg/L or any of the lower TBT concentrations. The mean weight of the surviving fish was reduced 4% at 0.08 µg/L, 9% at 0.15 µg/L, 26% at 0.45 µg/L, and 48% at 0.92 µg/L when compared to the control fish. Mean length of fry at the end of the test was significantly ($p \leq 0.05$) reduced at concentrations ≥ 0.45 µg/L. The mean biomass at the end of the test was higher at the two lowest TBT concentrations (0.08 and 0.15 µg/L) than in the controls, but was reduced by 13 and 52% at TBT concentrations of 0.45 and 0.92 µg/L, respectively. Because the reductions in weight of individual fish were small at the two lowest concentrations (0.08 and 0.15 µg/L) and the mean biomass increased at these same concentrations, the chronic limits are 0.15 and 0.45 µg/L based upon growth (length and weight). Thus the chronic value is 0.2598 µg/L and the acute-chronic ratio is 10.01 calculated using the acute value of 2.6 µg/L from the same study.

A partial life cycle test (began with egg capsules and ended before egg capsules were produced by the F₁ generation) was conducted with the saltwater stenoglossan snail *Nucella lapillus* (Harding et al. 1996). The study was conducted for one year with observations of egg capsule production, survival, and growth. The study by Harding et al. (1996) was a continuation of a study by Bailey et al. (1991) during which they exposed eggs and juvenile snails for one year to TBT. The study by Harding et al. (1996) began with egg capsules produced by adults at the end of the study by Bailey et al. (1991). Negative effects due to TBT were only observed in egg capsule production from the adults of the previous study. Females that had not been exposed for one year to TBT produced 14.42 egg capsules per female. Females that had been exposed to <0.0027, 0.0074, 0.0278, and 0.1077 µg TBT/L for one year produced 135.6, 104.6, 44.8, and 23.4% as many egg capsules as the controls for the respective TBT concentrations. The chronic value is based upon reproductive effects and is the geometric mean of the lowest observed effect concentration (LOEC) of 0.0278 µg/L and the no observed effect concentration (NOEC) of 0.0074 µg/L which is 0.0143 µg/L. Survival and growth were not affected at any TBT concentration tested. An acute-chronic ratio of 5,084 can be calculated using the acute value from this test of 72.7 µg/L. The acute-chronic ratio for *N. lapillus* is about 139 times higher than the next lower acute-chronic ratio for *D. magna* (36.60). It is not used to calculate a final acute-chronic ratio because it is more than ten times higher than any other ratio.

Two partial life-cycle toxicity tests were conducted using the marine copepod, *Eurytemora affinis* (Hall et al. 1987, 1988a). Tests began with egg-carrying females and lasted 13 days. In the first test, mean brood size was reduced from 15.2 neonates/female in the control to 0.2 neonates/female in 0.479 µg/L after three days. Percentage survival of neonates was 79% less than control survival in the lowest tested TBT concentration (0.088 µg/L), and 0% in 0.479 µg/L. The chronic value is <0.088 µg/L in this test.

In the second copepod test, percentage survival of neonates was significantly reduced (73% less than control survival) in 0.224 µg/L; brood size was unaffected in any tested concentration (0.018-0.224 µg/L). No statistically significant effects were detected in concentrations #0.094 µg/L. The chronic value in this test is 0.145 µg/L. It is calculated as the geometric mean of the NOEC (0.094 µg/L) and the LOEC (0.224 µg/L). The acute-chronic ratio is 15.17 when the acute value of 2.2 µg/L from this test is used.

Life-cycle toxicity tests were conducted with the saltwater mysid, *Acanthomysis sculpta* (Davidson et al. 1986a, 1986b). The effects of TBT on survival, growth, and reproduction of *A. sculpta* were determined in five separate tests lasting from 28 to 63 days. The tests separately examined effects of TBT on survival (1 test), growth (3 tests) and reproduction (1 test) instead of the approach of

examining all endpoints in one life-cycle test. All tests began with newly released juveniles and lasted through maturation and spawning; therefore, they are treated as one life-cycle test. The number of juveniles released per female at a TBT concentration of 0.19 µg/L was 50% of the number released in the control treatment, whereas the number released at the next lower TBT concentration (0.09 µg/L) was not significantly different from the control treatment. Reductions in the number of juveniles released resulted from deaths of embryos within brood pouches of individual females and not from reduced fecundity. Numbers of females releasing viable juveniles was reduced in 0.19 and 0.33 µg/L. At concentrations of 0.38 µg/L and above, survival and weight of female mysids were reduced; all mysids in 0.48 µg/L died. The chronic value (0.1308 µg/L) is the geometric mean of 0.09 µg/L and 0.19 µg/L and is based upon reproductive effects. The acute-chronic ratio is 4.664 when an acute value of 0.61 µg/L reported by Valkirs et al. (1985) is used (Table 2). The acute and chronic tests were conducted in the same laboratory.

The Final Acute-Chronic Ratio of 12.69 was calculated as the geometric mean of the acute-chronic ratios of 36.60 for *D. magna*, 10.01 for *P. promelas*, 4.664 for *A. sculpta* and 15.17 for *E. affinis*. Division of the freshwater and saltwater Final Acute Values by 12.69 results in Final Chronic Values for freshwater of 0.0723 µg/L and for saltwater of 0.0658 µg/L (Table 3). Both of these Chronic Values are below the experimentally determined chronic values from life-cycle or early life-stage tests (0.1414 µg/L for *D. magna* and 0.1308 µg/L for *A. sculpta*). The close agreement between the saltwater Final Chronic Value and the freshwater Final Chronic Value suggests that salinity has little if any affect on the toxicity of TBT.

TOXICITY TO AQUATIC PLANTS

The various plant species tested are highly variable in sensitivity to TBT. Twenty species of algae and diatoms were tested in fresh and salt water. The saltwater species appear to be more sensitive to TBT than the freshwater species for which data are available. No explanation is apparent.

Blanck et al. (1984) and Blanck (1986) reported the concentrations of TBT that prevented growth of thirteen freshwater algal species (Table 4). These concentrations ranged from 56.1 to 1,782 µg/L, but most were between 100 and 250 µg/L. Fargasova and Kizlink (1996), Huang et al. (1993), and Miana et al. (1993) measured severe reduction in growth of several green alga species at TBT concentrations ranging from 1 to 12.4 µg/L. Several green alga species appear to be as sensitive to TBT as many animal species (compare Table 4 with Table 1).

Toxicity tests on TBT have been conducted with five species of saltwater phytoplankton including the diatoms, *Skeletonema costatum* and *Nitzschia* sp., and the flagellate green algae, *Dunaliella tertiolecta*, *D. salina* and *D. viridis*. The 14-day EC50's (reduction in growth) for *S. costatum* of >0.12 but <0.24 µg/L in one test and 0.06 µg/L in a second test (EG&G Bionomics 1981c) were the lowest values reported for algal species. Thain (1983) reported that measured concentrations from 0.97 to 17 µg/L were algistatic to the same species in five-day exposures. The results for algal toxicity tests with the same species varied between laboratories by more than an order of magnitude. A diatom, *Nitzschia* sp., and two flagellate green alga of the genus *Dunaliella* sp. were less sensitive to TBT than *S. costatum*, but they were more sensitive than most species of freshwater algae. No data are available on the effects of TBT on fresh or saltwater vascular plants.

A Final Plant Value, as defined in the Guidelines, cannot be obtained because no test in which the concentrations of TBT were measured and the endpoint was biologically important has been conducted with an important aquatic plant species. The available data do indicate that freshwater and saltwater plants will be protected by TBT concentrations that adequately protect freshwater and saltwater animals.

ENDOCRINE DISRUPTION EFFECTS DATA

TBT has been shown to produce the superimposition of male sexual characteristics on female neogastropod (stenoglossan) snails (Gibbs and Bryan 1987; Smith 1981b) and one freshwater gastropod (Prosobranchia), *Marisa cornuarietis* (Schulte-Oehlmann et al. 1995). This phenomenon, termed "imposex," can result in females with a penis, a duct leading to the vas deferens, and a convolution of the normally straight oviduct (Smith 1981a). Other anatomical changes associated with imposex are detailed in Gibbs et al. (1988) and Gibbs and Bryan (1987). Severity of imposex is quantified using relative penis size (RPSI; ratio of female to male penis volume³ x 100) and the six developmental stages of the vas deferens sequence index (VDSI) (Bryan et al. 1986; Gibbs et al. 1987). TBT has been shown to impact populations of the Atlantic dogwhelk (dogwhelk), *Nucella lapillus*, which has direct development. In neoglossian snails with indirect development (planktonic larval stages), the impacts of TBT are less certain because recruitment from distant stocks of organisms can occur. Natural pseudohemaphroditism in neoglossans occurs (Salazar and Champ 1988) and may be caused by other organotin compounds (Bryan et al. 1988). However, increased global incidence and severity of imposex has been associated with areas of high boating activity and low to moderate

concentrations (low parts per trillion) of TBT in water, sediment or snails and other biota (Alvarez and Ellis 1990; Bailey and Davies 1988a, 1988b; Bryan et al. 1986, 1987a; Davies et al. 1987, Durchon 1982; Ellis and Pottisina 1990; Gibbs and Bryan 1986, 1987; Gibbs et al. 1987; Langston et al. 1990; Short et al. 1989; Smith 1981a, 1981b; Spence et al. 1990a). Imposex has been observed (12% of the females) in common whelk, *Buccinum undatum*, in the North Sea as far as 110 nautical miles from land (Ide et al. 1997). The sample from this site averaged 1.4 µg TBT/kg of wet weight soft tissues. Other samples of organisms collected nearer to shore in various places in the North Sea generally had higher TBT concentrations.

Although imposex has been observed in 45 species of snails worldwide (Ellis and Pottisina 1990, Jenner 1979), definitive laboratory and field studies implicating TBT as the cause have focused on seven North American or cosmopolitan species; the Atlantic dogwhinkle (*N. lapillus*), file dogwhinkle (*N. lima*), eastern mud snail [*Ilyanassa (Nassarius) obsoleta*], a snail (*Hinia reticulata*), whelks (*Thais orbita* and *T. clavigera*), and the European sting winkle (*Ocenebra erinacea*). Imposex has been associated with reduced reproductive potential and altered density and population structure in field populations of *N. lapillus* (Harding et al. 1997; Spence et al. 1990a). This is related to blockage of the oviduct by the vas deferens, hence, prevention of release of egg capsules, sterilization of the female or change into an apparently functional male (Bryan et al. 1986; Gibbs and Bryan 1986, 1987; Gibbs et al. 1987, 1988). TBT may reduce populations of *N. lima* because snails were absent from marinas in Auke Bay, AK. At intermediate distances from marinas, about 25 were caught per hour of sampling and 250 per hour were caught at sites distant from marinas (Short et al. 1989). Snails from intermediate sites had blocked oviducts. Reduced proportions of female *I. obsoleta* in Sarah Creek, VA also suggests population impacts (Bryan et al. 1989a). However, other causes may explain this as oviducts which were not blocked and indirect development (planktonic larvae) facilitating recruitment from other areas which may limit impacts.

Several field studies have used transplantations of snails between sites or snails painted with TBT paints to investigate the role of TBT or proximity to marinas in the development of imposex without defining actual exposure concentrations of TBT. Short et al. (1989) painted *N. lima* with TBT-based paint, copper paints or unpainted controls. For 21 females painted with TBT paint, seven developed penises within one month, whereas, penises were absent from 35 females from other treatments. Smith (1981a) transplanted *I. obsoleta* between marinas and "clean" locations and found that incidence of imposex was unchanged after 19 weeks in snails kept at clean locations or marinas, increased in snails transplanted from clean sites to marinas and decreased somewhat in transplants from marinas to clean sites. Snails exposed in the laboratory to TBT-based paints in two separate

experiments developed imposex within one month with maximum impact within 6 to 12 months (Smith 1981a). Snails painted with non-TBT paints were unaffected.

Concentration-response data demonstrate a similarity in the response of snails to TBT in controlled laboratory and field studies (Text Table A). Eastern mud snails, *I. obsoleta*, collected from the York River, VA near Sarah Creek had no incidence of imposex (Bryan et al. 1989a) and contained no detectable TBT (<0.020 µg/g dry weight). The average TBT concentrations of York River water was 0.0016 µg/L. In contrast, the average TBT concentrations from four locations in Sarah Creek, VA were from 0.010 to 0.023 µg/L, snails contained about 0.1 to 0.73 µg/g TBT and there was a 40 to 100% incidence of imposex. Short et al. (1989) collected file dogwhinkle snails, *N. lima*, from Auke Bay, AK and did not detect imposex or TBT in snails from sites far from marinas. Snails from locations near marinas all exhibited imposex and contained 0.03 to 0.16 µg/g TBT dry wt. tissue.

Text Table A. Summary of available laboratory and field studies relating the extent of imposex of female snails, measured by relative penis size (RPSI = ratio of female to male penis volume \times 100) and the vas deferens sequence index (VDSI), as a function of tributyltin concentration in water and dry tissue.

Species	Experimental Design	TBT Concentration		Imposex			Reference
		Water $\mu\text{g/L}$	Snail Tissue $\mu\text{g/g dry}$	RPSI	VDSI	Comments	
Eastern mud snail, <i>Ilyanassa obsoleta</i>	Field-York River, UK	0.0016	<0.02	-	-	No imposex	Bryan et al. 1989a
	-Sarah Creek	0.01-0.023	- 0.1-0.73	-	-	40-100% incidence	
File dogwhinkle, <i>Nucella lima</i>	Field-Auke Bay, AK	-	ND(<0.01)	0.0	0.0	0% incidence	Short et al. 1989
	-Auke Bay, AK	-	0.03-0.16	14-34	2.2-4.3	100% incidence, reduced abundance	
Atlantic dogwhinkle, (adults), <i>Nucella lapillus</i>	Crooklets Beach, UK Laboratory: 2 year exposure	<0.0012*	0.14-0.25*	2-65	2.9	-	Bryan et al. 1987a
		0.0036*	0.41*	10/14.2	3.7/3.7	-	
		0.0083*	0.74*	43.8	3.9	-	
		0.046*	4.5*	56.4	4.0	-	
Atlantic dogwhinkle, <i>Nucella lapillus</i>	Laboratory, spires painted, 8 mo.	-	- 5.1*	10-50	-	-	Bryan et al. 1987b
		0.26*	8.5*	63.3	4.1	Some sterilization	
Atlantic dogwhinkle, (egg capsule to adult), <i>Nucella lapillus</i>	Crooklets Beach, UK Laboratory; 2 year exposure	<0.0012	0.19	3.7	3.2	Normal females	Gibbs et al. 1988
		0.0036	0.58	48.4	4.4	1/3 sterile, 160 capsules	
		0.0093	1.4	96.6	5.1	All sterile, 2 capsules	
		0.049	4.1	109	5.0	All sterile, 0 capsules	
Atlantic dogwhinkle, <i>Nucella lapillus</i>	Transplants, Crooklets Beach to Dart Estuary, UK	0.022-0.046	9.7	96.3	5.0	All sterile	Gibbs et al. 1988
		-	0.11*	0.0	-	0% aborted egg capsules	
Atlantic dogwhinkle, <i>Nucella lapillus</i>	Port Joke, UK Crooklets Beach Meadfoot Renny Rocks Batten Bay	-	0.21*	2.0	-	0% aborted egg capsules	Gibbs and Bryan 1986; Gibbs et al. 1987
		-	0.32*	30.6	-	15% aborted egg capsules	
		-	0.43*	38.9	-	38% aborted egg capsules	
		-	1.54*	22.9	-	79% aborted egg capsules	
Atlantic dogwhinkle, <i>Nucella lapillus</i>	Laboratory, flow-through, one year	<0.0015	<0.10*	0.10	1.06	Control, 37.1% imposex	Bailey et al. 1991
		<0.0015	<0.10*	0.04	0.70	Solvent control, 24.3% imposex	
		<0.0027	0.35*	5.33	3.15	5.3% reduced growth, 92.3% imposex	
		0.0077	1.10*	20.84	3.97	11.0% reduced growth, 100% imposex	
		0.0334	3.05*	42.08	4.33	17.1% reduced growth, 100% imposex	
Atlantic dogwhinkle, <i>Nucella lapillus</i>	Laboratory, flow-through, one year	0.1246	4.85*	63.40	4.25	18.9% reduced growth, 100% imposex	Harding et al. 1996
		<0.0015	<0.10	0.07	1.28	Control, 42.2% imposex	
		<0.0015	<0.10	0.04	1.14	Solvent control, 37.5% imposex	
		0.0026	<0.10	64.04	3.98	98.9% imposex	
		0.0074	0.38	88.57	4.96	98.8% imposex	
0.0278	1.12	90.96	5.00	100% imposex			
0.1077	3.32	117.70	4.99	98.7% imposex			

Text Table A. Continued

Species	Experimental Design	TBT Concentration		Imposex			Reference
		Water $\mu\text{g/L}$	Snail Tissue $\mu\text{g/g dry}$	RPSI	VDSI	Comments	
Snail, <i>Hinia reticulata</i>	Field-32 sites N and NW France	-	<1.5	<10	<3.0	Low imposex incidence	Stroben et al. 1992a
		-	>1.5	>30	>3.0	High imposex incidence	
Whelk, <i>Thais orbita</i>	Field-Queenscliff, UK	-	0.365*	19.55	-	100% incidence	Foale 1993
	-Sandringham	-	0.224*	12.16	-	100% incidence	
	-Brighton	-	<0.002*	7.34	-	100% incidence	
	-Portarlinton	-	0.255*	3.67	-	92.3% incidence	
	-Morningsong	-	0.045*	2.55	-	100% incidence	
	-Williamstown	-	<0.002*	1.25	-	100% incidence	
	-Martha Point	-	0.031*	0.03	-	25% incidence	
	-Kirk Point	-	0.011*	0.02	-	35.7% incidence	
	-Cape Schanck	-	0.108*	0	-	0% incidence	
	-Cape Schanck	-	0.095*	0	-	0% incidence	
	-Barwon Heads	-	ND	0	-	0% incidence	
	-Barwon Heads	-	0.071*	0	-	0% incidence	
Whelk, <i>Thais clavigera</i>	Field-32 sites Japan	-	0.005	<10	-	100% incidence	Horiguchi et al. 1995
		-	0.010	1-42	-	100% incidence	
		-	0.020	30-75	-	100% incidence	
		-	0.030	30	-	100% incidence	
		-	0.040	75	-	100% incidence	
		-	0.050	80	-	100% incidence	
		-	0.060	80	-	100% incidence	
		-	0.070	40	-	100% incidence	
Common whelk, (juvenile), <i>Buccinum undatum</i>	Laboratory; 10 month exposure with Wadden Sea water	Control	-	-	-	17% some imposex	Mensink et al. 1996
		0.01	-	-	-	22% some imposex	
		0.10	-	-	-	80% some imposex	
		1.0	\$40.1	-	-	98% severe imposex; 88% developed vas deferens	
European sting winkle, <i>Ocenebra erinacea</i>	Field-19 sites SW UK	-	0.185	0	-	No imposex	Gibbs et al. 1990
		-	<0.024	0	-	No imposex	
		-	0.187	16.3	-	Females somewhat deformed	
		-	0.773	66.9	-	Females highly deformed	
		-	2.313	88.2	-	Females highly deformed	
		-	0.976	71.1	-	Females highly deformed	
		-	1.057	53.4	-	Females highly deformed	
		-	1.200	84.2	-	Females highly deformed	
		-	0.303	7.4	-	Females highly deformed	
		-	0.122	7.0	-	Females somewhat deformed	
		-	0.703	36.0	-	Females somewhat deformed	
		-	0.764	52.7	-	Females somewhat deformed	
		-	0.527	46.5	-	Females somewhat deformed	
		-	0.488	42.3	-	Females highly deformed	
		-	0.366	0.04	-	Females highly deformed	
		-	0.253	33.9	-	Females highly deformed	
-	0.832	58.0	-	Females highly deformed			
-	1.010	79.3	-	Females somewhat deformed			
-	0.510	59.7	-	Females highly deformed			
-	-	-	-	Females highly deformed			
-	-	-	-	Females highly deformed			
-	-	-	-	Females highly deformed			
-	-	-	-	Females highly deformed			

* Concentrations changed from $\mu\text{g Sn/L}$ or $\mu\text{g Sn/g}$ wet tissue to $\mu\text{g TBT/L}$ or $\mu\text{g TBT/g}$ dry weight tissue. Dry weight estimated as 20% of wet weight (or 80% water content).

Gibbs et al. (1988) conducted a laboratory study with the Atlantic dogwhelk, *Nucella lapillus*, exposed to a dilution water control and four TBT exposures. Since the dilution water control collected nearby had low levels of TBT, organisms were also collected from a nearby uncontaminated site. Concentrations of TBT in females were 0.19 $\mu\text{g/g}$ dry wt. in the field, 0.58 $\mu\text{g/g}$ dry wt. in the 0.0036 $\mu\text{g/L}$ laboratory water treatment and from 1.4 to 7.7 $\mu\text{g/g}$ dry weight in >0.0093 $\mu\text{g/L}$ laboratory water treatments. Similar concentrations of TBT (9.7 $\mu\text{g/g}$ dry wt.) were found in snails which became sterile after they were placed in the Dart Estuary, UK where TBT concentrations range from 0.022 to 0.046 $\mu\text{g/L}$ (Gibbs et al. 1988). Gibbs and Bryan (1986) and Gibbs et al. (1987) report imposex and reproductive failures at other marine sites where TBT concentrations in female snails range from 0.32 to 1.54 $\mu\text{g/g}$ dry wt. tissue. In two studies conducted concurrently with *H. lapillus* for one year each, imposex was observed. In the first study (Bailey et al. 1991), imposex (stage 2) was observed in 92.3% of the females exposed to TBT at near 0.0027 $\mu\text{g/L}$ or greater (up to 0.1246 $\mu\text{g/L}$) at the end of the study. Harding et al. (1996) exposed the offspring from parents exposed in the study by Bailey et al. (1991) for one additional year to similar TBT concentrations. In the second generation of TBT-treated snails, body burdens of TBT were lower in the second generation at similar treatment concentrations used in the first generation, but the RPSI and VDSI values were higher. Harding et al. (1996) found 98.7% imposex in females at TBT concentrations 0.0026 $\mu\text{g/L}$.

Four species of snails (*Hinia reticulata*, *Thais orbita*, *T. clavigera* and *Ocenebra erinacea*) not resident to North America also demonstrated imposex effects when exposed to TBT in field studies (Text Table A). The snail *H. reticulata* is less sensitive to TBT than other snails having higher body burdens (>1.5 $\mu\text{g/g}$ dry wt.) before showing effects of imposex. *Thais* sp. showed high imposex incidence at tissue concentrations as low as 0.005 $\mu\text{g/g}$ dry wt. and no imposex at other locations with tissue concentrations of 0.108 $\mu\text{g/g}$ dry wt. *Ocenebra erinacea* did not show imposex in a field study at body burdens as high as 0.185 $\mu\text{g/g}$ dry wt., but females were deformed at all higher concentrations.

In summary, in both field and laboratory studies, concentrations of TBT in water of about 0.0015 $\mu\text{g/L}$ or less and in tissues of about 0.2 $\mu\text{g/g}$ dry wt. or less do not appear to cause imposex in *N. lapillus*. Imposex begins to occur at about 0.003 $\mu\text{g TBT/L}$, and some reproductive failure at concentrations greater than 0.005 $\mu\text{g/L}$, with complete sterility occurring after chronic exposure of sensitive early life-stages at 0.009 $\mu\text{g/L}$ (for less sensitive stages, imposex does not occur until about 0.02 $\mu\text{g/L}$ in some studies and greater than 0.2 $\mu\text{g/L}$ in others).

BIOACCUMULATION

Bioaccumulation of TBT has been measured in one species of freshwater mollusc and four species of freshwater fish (Table 5). Adults of the zebra mussel, *Dreissena polymorpha*, were placed in cages at a freshwater marina and at an uncontaminated site in a lake for 105 days (Becker-van Slooten and Tarradellas 1994). Subsamples of the organisms were periodically monitored for TBT tissue concentrations. They reached steady-state concentrations after 35 days. The BCF/BAF was 17,483 when adjusted for wet weight and lipid normalized to 1% for TBT at an average water exposure concentration of 0.0703 µg/L. Growth of the TBT-exposed organisms may have been slightly reduced. Martin et al. (1989) determined the whole body bioconcentration factor (BCF) for rainbow trout, *Oncorhynchus mykiss* to be 406 after a 64-day exposure to 0.513 µg TBT/L. Equilibrium of the TBT concentration was achieved in the fish in 24 to 48 hrs. In a separate exposure to 1.026 µg TBT/L, rainbow trout organs were assayed for TBT content after a 15-day exposure. The BCFs ranged from 312 for muscle to 5,419 for peritoneal fat. TBT was more highly concentrated than the metabolites of di- and monobutyltin or tin. Carp, *Cyprinus carpio*, and guppy, *Poecilia reticulatus*, demonstrated plateau BCF's in 14 days. BCFs were 501.2 based on carp muscle tissue, and from approximately 1,190 to 2,250 based on whole body tissue (Tsuda et al. 1990a). BCFs based on whole body guppy tissue were somewhat lower ranging from 240 to 460 (Tsuda et al. 1990b). Goldfish, *Carassius auratus*, reached a BCF in the whole body after 28 days of 1,976.

The extent to which TBT is accumulated by saltwater animals from the field or from laboratory tests lasting 28 days or more has been investigated with four species of bivalve molluscs, two species of snails (Table 5). Thain and Waldock (1985) reported a BCF of 6,833 for the soft parts of blue mussel spat exposed to 0.24 µg/L for 45 days. In other laboratory exposures of blue mussels, Salazar et al. (1987) observed BCFs of 10,400 to 37,500 after 56 days of exposure. BAFs from field deployments of mussels were similar to the BCFs from laboratory studies: 11,000 to 25,000 (Salazar and Salazar 1990a) and 5,000 to 60,000 (Salazar and Salazar 1991). In a study by Bryan et al. (1987a), laboratory BCFs for the snail *Nucella lapillus* (11,000 to 38,000) also were similar to field BAFs (17,000). Year-long laboratory studies by Bailey et al. (1991) and Harding et al. (1996) produced similar BAFs in *N. lapillus* ranging from 6,172 to 21,964. In these tests, TBT concentrations ranged from 0.00257 to 0.125 µg/L, but there was no increase in BAFs with increased water concentration of TBT.

The soft parts of the Pacific oyster, *Crassostrea gigas*, exposed to TBT for 56 days contained 11,400 times the exposure concentration of 0.146 µg/L (Waldock and Thain 1983). A BCF of 6,047

was observed for the soft parts of the Pacific oyster exposed to 0.1460 µg/L for 21 days (Waldock et al. 1983). The lowest steady-state BCF reported for a bivalve was 192.3 for the soft parts of the European flat oyster, *Ostrea edulis*, exposed to a TBT concentration of 2.62 µg/L for 45 days (Thain and Waldock 1985; Thain 1986). Other tests with the same species (Table 5) resulted in BCFs ranging from 397 to 1,167.

In a field study conducted in the Icelandic harbor of Reykjavik with the blue mussel, *M. edulis*, and the Atlantic dogwhelk, *N. lapillus*, seasonal fluctuations were seen in body burdens of TBT and DBT (Skarphedinsdottir et al. 1996). They did not report the water concentrations for TBT, and speculated that because shipping did not vary seasonally, the fluctuations in body burdens were due to seasonal feeding and resting activities. They demonstrated that body burdens of TBT and DBT were highest at high latitudes during late summer or early autumn.

No U.S. FDA action level or other maximum acceptable concentration in tissue, as defined in the Guidelines, is available for TBT, and, therefore, no Final Residue Value can be calculated.

OTHER DATA

Many tests with TBT and various freshwater or saltwater organisms have been conducted either for a different duration or by different protocols than those specified in the Guidelines for inclusion in Tables 1, 2, 4 and 5. These data, presented in Table 6, are potentially useful and sometimes support data in other tables. For example, plant tests were included in Table 6 rather than Table 4 if the test duration was less than 4 days or the exposure concentrations were not measured. Tests with animals were included in Table 6 for a number of reasons, including considerations of test duration, type of test, and test endpoints other than those of toxicity or bioaccumulation. The data in this section are used to lower the saltwater CCC value.

Several studies report the effects of TBT on natural groups of organisms in laboratory microcosms. In most of these studies, the effects of TBT administered to the water were rapid. Two microcosm studies were conducted with freshwater organisms (Delupis and Miniero, 1989; Miniero and Delupis, 1991) in which single dose effects were measured on natural assemblages of organisms. In both studies, the effects were immediate. *D. magna* disappeared soon after an 80 µg/L dose of TBT, ostracods increased, and algal species increased immediately then gradually disappeared during the 55-day study. In the second study (Miniero and Delupis, 1991), metabolism was monitored by measuring oxygen consumption and again the effects were rapid. Doses of TBT (4.7 and 14.9 µg/L) were

administered once and metabolism was reduced at 2.5 days and returned to normal in 14.1 days in the lower exposure. In the higher exposure, metabolism was reduced in one day and returned to normal in 16 days. Kelly et al. (1990a) observed a similar response in a seagrass bed at 22.21 µg/L of TBT. The primary herbivore, *Cymadusa compta*, declined and the sea grass increased in biomass. Saltwater microbial populations were exposed for one hour to TBT concentrations of 4.454 and 89.07 µg/L then incubated for 10 days (Jonas et al. 1984). At the lower concentration, metabolism of nutrient substrates was reduced and at the higher concentration, 50 percent mortality of microbes occurred.

Several fresh and saltwater algal species were exposed to TBT for various time intervals and several endpoints determined. Toxicity (EC50) in freshwater species ranged from 5 µg/L for a natural assemblage to 20 µg/L for the green alga *Ankistrodesmus falcatus* (Wong et al. 1982). Several salt water alga, a green alga, *Dunaliella tertiolecta*; the diatoms, *Minutocellus polymorphus*, *Nitzshia* sp., *Phaeodactylum tricorutum*, *Skeletonema costatum*, and *Thalassiosira pseudonana*; the dinoflagellate, *Gymnodinium splendens*, the microalga, *Pavlova lutheri* and the macroalga, *Fucus vesiculosus* were tested for growth endpoints. The 72-hr EC50s based on population growth ranged from approximately 0.3 to <1.5 µg/L (Table 6). Lethal concentrations were generally more than an order of magnitude greater than EC50s and ranged from 10.24 to 13.82 µg/L. Identical tests conducted with tributyltin acetate, tributyltin chloride, tributyltin fluoride, and tributyltin oxide exposures with *S. costatum* resulted in EC50s from 0.2346 to 0.4693 µg/L and LC50s from 10.24 to 13.82 µg/L (Walsh et al. 1985, 1987).

The freshwater invertebrates, a rotifer (*Brachionus calyciflorus*) and a coelenterate (*Hydra* sp.), showed widely differing sensitivities to TBT. *Hydra* sp. were affected at 0.5 µg/L resulting in deformed tentacles, but the rotifer did not show an effect on hatching success until the exposure concentration reached 72 µg/L. The cladoceran, *D. magna*, has 24-hr EC50s ranging from 3 (Polster and Halacha 1972) to 13.6 µg/L (Vighi and Calamari 1985). When the endpoint of altered phototaxis was examined in a longer-term exposure of 8 days, a much lower effect concentration of 0.45 µg/L was measured (Meador 1986).

Saltwater invertebrates (exclusive of molluscs) had reduced survival at concentrations as low as 0.500 µg/L for the polychaete worm, *Neanthes arenaceodentata* in a 10 week exposure to TBT (Moore et al. 1991) and 0.003 µg/L in a copepod, *Acartia tonsa*, in an eight-day exposure (Kusk and Peterson 1997). Other invertebrates were more hardy including an amphipod, *Orchestia traskiana*, that had an LC80 and an LC90 of 9.7 µg/L for nine day exposures to TBTO and TBTF, respectively (Laughlin et al. 1982). Larvae of the mud crab, *Rhithropanopeus harrisi*, tolerated high concentrations of TBT with one test resulting in an LC50 of 33.6 µg/L for a 40 day exposure (Laughlin and French 1989).

A number of studies showed that TBT exposure resulted in developmental problems for non-

mollusc invertebrates. For example, the copepod, *A. tonsa*, had slower rate of development from nauplii to copepodid stage at 0.003 µg/L (Kusk and Petersen 1997); the grass shrimp, *Palaemonetes pugio*, had retarded telson regeneration at 0.1 µg/L (Khan et al. 1993); the mud crab, *R. harrisi*, had reduced developmental rate at 14.60 µg/L (Laughlin et al. 1983); retarded limb regeneration in the fiddler crab, *Uca pugilator*, at 0.5 µg/L (Weis et al. 1987a); and retarded arm regeneration in the brittle star, *Brevoortia tyrannus*, at 0.1 µg/L (Walsh et al. 1986a).

Vertebrates are as sensitive to TBT as invertebrates when the exposures are of sufficient duration. Rainbow trout, *O. mykiss*, exposed in short-term exposures of 24 to 48 hr have LC50 and EC50 values from 18.9 to 30.8 µg/L (Table 6). When the exposure is increased to 110 days (Seinen et al. 1981), the LC100 decreased to 4.46 µg/L and a 20% reduction in growth was seen at 0.18 µg/L. de Vries et al. (1991) measured a similar response in rainbow trout growth in another 110 day exposure. They demonstrated decreased survival and growth at 0.200 µg/L but not at 0.040 µg/L. Triebkorn et al. (1994) found reduced growth and behavior changes in the fish at 21 days when exposed to 0.5 µg/L. The frog, *Rana temporaria*, has a LC50 of 28.2 µg/L for a 5-day exposure to TBT (Laughlin and Linden 1982).

An attempt was made to measure the bioconcentration of TBT with the green alga, *Ankistrodesmus falcatus* (Maguire et al. 1984). The algae are able to degrade TBT to its di- and monobutyl forms. As a result, the concentrations of TBT steadily declined during the 28-day study. During the first seven days of exposure, the concentrations declined from 20 to 5.2 µg/L and the calculated BCF was 300 (Table 6). After 28 days of exposure, the TBT concentration had declined to 1.5 µg/L and the calculated BCF was 467. Several studies reported BCFs for fish but failed to demonstrate plateau concentrations in the organism. In these studies, rainbow trout BCFs ranged from 540 (Triebkorn et al. 1994) to 3,833 (Schwaiger et al. 1992). Goldfish achieved a BCF of 1,230 (Tsuda et al. 1988b) in a 14-day exposure and carp achieved a BCF of 295 in the muscle tissue in 7 days (Tsuda et al. 1987).

Reproductive abnormalities have also been observed in the European flat oyster, *Ostrea edulis* (Thain 1986). After exposure for 75 days to a TBT concentration of 0.24 µg/L, a retardation in the sex change from male to female was observed and larval production was completely inhibited. A TBT concentration of 2.6 µg/L prevented development of gonads (Table 6). Salazar et al. (1987) found no negative effects in the same species at 0.157 µg/L, but Thain and Waldock (1985) and Thain (1986) measured reduced growth at 0.2392 µg/L and reduced survival (30%) at 2.6 µg/L (Table 6).

Survival and growth of several commercially important saltwater bivalve molluscs have been studied during acute and long-term exposures to TBT. Mortality of larval blue mussels, *Mytilus edulis*,

exposed to 0.0973 µg/L for 15 days was 51%; survivors were moribund and stunted (Beaumont and Budd 1984). Similarly, Dixon and Prosser (1986) observed 79% mortality of mussel larva after 4 days exposure to 0.1 µg/L. Lapota et al. (1993) reported reduced shell growth in the blue mussel, *Mytilus edulis*, at 0.050 µg/L and no reduction of shell development at 0.006 µg/L in a 33-d study. The test had exposure solutions renewed every third or fourth day during which time TBT concentrations declined 33 to 90%. Growth of juvenile blue mussels was significantly reduced after 7 to 66 days at 0.31 to 0.3893 µg/L (Stromgren and Bongard 1987; Valkirs et al. 1985). Growth rates of mussels transplanted into San Diego Harbor were impacted at sites where TBT concentrations exceeded 0.2 µg/L (Salazar and Salazar 1990b). At locations where concentrations were less than 0.1 µg/L, the presence of optimum environmental conditions for growth appear to limit or mask the effects of TBT. Less than optimum conditions for growth may permit the effect of TBT on growth to be expressed. Salazar et al. (1987) observed that 0.157 µg/L reduced growth of mussels after 56 days exposure in the laboratory; a concentration within less than a factor of two of that reducing growth in the field. Similarly, Salazar and Salazar (1987) observed reduced growth of mussels exposed to 0.070 µg/L for 196 days in the laboratory. The 66-day LC50 for 2.5 to 4.1 cm blue mussels was 0.97 µg/L (Valkirs et al. 1985,1987). Alzieu et al. (1980) reported 30% mortality and abnormal shell thickening among Pacific oyster larvae exposed to 0.2 µg/L for 113 days. Abnormal development was also observed in exposures of embryos for 24 hrs or less to TBT concentrations \$0.8604 µg/L (Robert and His 1981). Waldoock and Thain (1983) observed reduced growth and thickening of the upper shell valve of Pacific oyster spat exposed to 0.1460 µg/L for 56 days. Abnormal shell development was observed in an exposure to 0.77 µg/L that began with embryos of the eastern oyster, *Crassostrea virginica*, and lasted for 48 hours (Roberts 1987). Adult eastern oysters were also sensitive to TBT with reductions in condition index after exposure for 57 days to \$0.1 µg/L (Henderson 1986). Salazar et al. (1987) found no effect on growth after 56 days exposure to 0.157 µg/L to the oysters *C. gigas*, *Ostrea edulis* and *O. lurida*. Condition of adult clams, *Macoma nasuta*, and scallops, *Hinnites multirugosus* were not affected after 110 days exposure to 0.204 µg/L (Salazar et al. 1987).

Long-term exposures have been conducted with a number of saltwater crustacean species. Johansen and Mohlenberg (1987) exposed adult *A. tonsa* for five days to TBT and observed impaired (25% reduction) egg production on days 3, 4 and 5 in 0.1 µg/L. Impaired egg production to a lesser amount was observed on day 5 in 0.01 and 0.05 µg/L. Davidson et al. (1986a,1986b), Laughlin et al. (1983,1984b), and Salazar and Salazar (1985a) reported that TBT acts slowly on crustaceans and that behavior might be affected several days before mortality occurs. Survival of larval amphipods, *Gammarus oceanicus*, was significantly reduced after eight weeks of exposure to TBT concentrations

\$0.2816 µg/L (Laughlin et al. 1984b). Hall et al. (1988b) observed no effect of 0.579 µg/L on *Gammarus* sp. after 24 days. Developmental rates and growth of larval mud crabs, *Rhithropanopeus harrisi*, were reduced by a 15-day exposure to \$14.60 µg/L. *R. harrisi* might accumulate more TBT via ingested food than directly from water (Evans and Laughlin 1984). TBTF, TBTO, and TBTS were about equally toxic to amphipods and crabs (Laughlin et al. 1982,1983,1984a). Laughlin and French (1989) observed LC50 values for larval developmental stages of 13 µg/L for crabs (*R. harrisi*) from California vs 33.6 µg/L for crabs from Florida. Limb malformations and reduced burrowing were observed in fiddler crabs exposed to 0.5 µg/L (Weis and Kim 1988; Weis and Perlmutter 1987; Weis et al. 1987a). Exposure to \$0.1 µg/L during settlement of fouling organisms reduced number of species and species diversity of communities (Henderson 1986). The hierarchy of sensitivities of phyla in this test was similar to that of single species tests.

Exposure of embryos of the California grunion, *Leuresthes tenuis*, for ten days to 74 µg/L caused a 50% reduction in hatching success (Newton et al. 1985). At TBT concentrations between 0.14 and 1.72 µg/L, growth, hatching success, and survival were significantly enhanced. In contrast, growth of inland silverside larvae was reduced after 28 days exposure to 0.093 µg/L (Hall et al. 1988b). Juvenile Atlantic menhaden, *Brevoortia tyrannus*, avoided a TBT concentration of 5.437 µg/L and juvenile striped bass, *Morone saxatilis*, avoided 24.9 µg/L (Hall et al. 1984). BCFs were 4,300 for liver, 1,300 for brain, and 200 for muscle tissue of chinook salmon, *Oncorhynchus tshawytscha*, exposed to 1.49 µg/L for 96 hours (Short and Thrower 1986a,1986c).

TBT concentrations less than the saltwater Final Chronic Value of 0.0658 µg/L from Table 3 have been shown to affect the growth of early life-stages of commercially important bivalve molluscs and survival of ecologically important copepods (Table 6; Text Table B). Laughlin et al. (1987, 1988) observed a significant decrease in growth of hard clam (*Mercenaria mercenaria*) larvae exposed for 14 days to \$0.01 µg/L (Text Table B). Growth rate (increase in valve length) was 75% of controls in 0.01 µg/L, 63% in 0.025 µg/L, 59% in 0.05 µg/L, 45% in 0.1 µg/L, 29% in 0.25 µg/L and 2.2% in 0.5 µg/L. A five-day exposure followed by nine days in TBT-free water produced similar responses and little evidence of recovery.

Pacific oyster (*Crassostrea gigas*) spat exhibited shell thickening in 0.01 and 0.05 µg/L and reduced valve lengths in \$0.02 µg/L (Lawler and Aldrich 1987; Text Table B). Increase in valve length was 101% of control lengths in 0.01 µg/L, 72% in 0.02 µg/L, 17% in 0.05 µg/L, 35% in 0.1 µg/L and 0% in 0.2 µg/L. Shell thickening was also observed in this species exposed to \$0.02 µg/L for 49 days (Thain et al. 1987). They predicted from these data that approximately 0.008 µg/L would be the maximum TBT concentration permitting culture of commercially acceptable adults. Their field studies

agreed with laboratory results showing "acceptable" shell thickness where TBT concentrations averaged 0.011 and 0.015 $\mu\text{g/L}$, but not at higher concentrations. Decreased weights of oyster meats were associated with locations where there was shell thickening.

Growth of spat of the European oyster (*Ostrea edulis*) was reduced at 0.02 $\mu\text{g/L}$ (Thain and Waldock 1985; Text Table B). Spat exposed to TBT in static-renewal tests were 76-81% of control lengths and 75% of control weights; extent of impact increased with increased exposure. In these static-renewal and flow-through tests at exposures at about 0.02 $\mu\text{g/L}$, weight gain was identical; i.e., 35% of controls. Growth of larger spat was marginally reduced by 0.2392 $\mu\text{g/L}$ (Thain 1986; Thain and Waldock 1985). Axiak et al. (1995a) observed a 12% decrease in the height of *O. edulis* digestive cells exposed to 0.01 $\mu\text{g/L}$ TBT.

Text Table B. Summary of laboratory and field data on the effects of tributyltin on saltwater organisms at concentrations less than the Final Chronic Value of 0.0658 $\mu\text{g/L}$.

<u>Species</u>	<u>Experimental Design^a</u>	<u>Concentration</u> <u>($\mu\text{g/L}$)</u>	<u>Response</u>	<u>Reference</u>
		<u>Nominal</u>		
Hard clam (4 hr larvae - metamorphosis), <i>Mercenaria mercenaria</i>	R, M, 14-day duration, <150 larvae/50 ml replicate, three replicates. Measured = 80-100% nominal at t = 0.4 hr; 20-30% at t = 24 hr	control 0.01-0.5	100% Growth (Valve length) - 75%-22% Growth (Valve length) ^b	Laughlin et al. 1987,1988
		<u>Nominal</u>		
Pacific oyster (spat), <i>Crassostrea gigas</i>	R, N, 48-day duration, 20 spat/treatment	control 0.01-0.05 control 0.01-0.2 0.02-0.2	Shell thickening 100% Growth (Valve length) 101% Growth (Valve length) 0-72% Growth (Valve length) ^b	Lawler and Aldrich 1987

Pacific oyster (spat), <i>Crassostrea gigas</i>	R, N, 49-day duration, 0.7 to 0.9 g/spat	<u>Nominal</u> control 0.002 0.02-2.0	No shell thickening Shell thickening proportional to concentration increase	Thain et al. 1987
	Field	<u>Measured</u> 0.011-0.015 - 0.018-0.060	No shell thickening Shell thickening and decreased meat weight	
European oyster (spat), <i>Ostrea edulis</i>	R, N, 20-day duration, 50 spat/treatment	<u>Nominal</u> control 0.02-2.0	100% length 76-81% length ^b	Thain and Waldock 1985
		control 0.02-2.0	202% weight gain 151-50% weight gain	
European oyster (adult), <i>Ostrea edulis</i>	R, N, 96-hr duration	<u>Nominal</u> 0.010	12% decrease of height of digestive cells	Axiak et al. 1995a

^a R = renewal; F = flow-through, N = nominal, M = measured.

^b Significantly different from controls.

TBT concentrations less than the saltwater Final Chronic Value of 0.0658 µg/L from Table 3 have also been shown to cause imposex (the superimposition of male anatomical characteristics on females) in seven ecologically important North American species, especially the dogwhelk, *Nucella lapillus*. A summary of three definitive laboratory studies conducted with *N. lapillus* are presented in Text Table C below.

Text Table C. Effect and no-effect tributyltin concentrations in laboratory studies with the Atlantic dogwhelk (*Nucella lapillus*).

Duration of Study (Life-stage)	NOEC (µgTBT/L)	Comment	LOEC (µgTBT/L)	Comment	Reference
Two year (Egg capsule to adult)	0.0036	RPSI = 48.4 VDSI = 4.4 1/3 Sterile	0.0093	RPSI = 96.6 VDSI = 5.1 All sterile	Gibbs et al. 1988

One year (Adults)	0.0077	RPSI = 20.8 VDSI = 3.97 11.0% reduced growth; 100% imposex	0.0334	RPSI = 42.1 VDSI = 4.3 17.1% reduced growth; 100% imposex	Bailey et al. 1991
One year (Egg capsule to adult)	0.0074	RPSI = 64.0 VDSI = 4.0 Reproduction 104.6% of control	0.0278	RPSI = 88.6 VDSI = 5.0 Reduced reproduction 55.2%	Harding et al. 1996

NOEC = No-Observed-Effect-Concentration
 LOEC = Lowest-Observed-Effect-Concentration
 RPSI = Relative penis size index
 VDSI = Vas deferens sequence index

Of the three studies listed above, two were initiated with egg capsules and one was conducted with adults. As specified by the Guidelines, the one study conducted with adults (Bailey et al. 1991) will not be considered for criteria derivation since studies conducted with a more sensitive life stage take precedence over studies with less sensitive life stages. The laboratory study conducted by Gibbs et al. (1988) did document adverse effects of TBT exposure on egg capsule production of the female dogwhelk, but a valid laboratory water control was not run with the study (the dilution water control collected nearby had low levels of TBT). Thus, this study also will not be used to derive the saltwater chronic criterion. The Harding et al. (1996) study was initiated with egg capsules and had a valid control. The NOEC and LOEC values presented are for reproductive effects, which is a direct measure of this species ability to survive long term.

The National Guidelines (Stephan et al. 1985; pp 18 and 54) requires that the criterion be lowered if sound scientific evidence indicates that adverse effects might be expected on important species. The above data demonstrate that the reductions in growth occur in commercially or ecologically important saltwater species at concentrations of TBT less than the Final Chronic Value of 0.0658 µg/L derived using Final Acute Values and Acute-Chronic Ratios from Table 3. Imposex begins to occur at about 0.003 µg TBT/L, and some reproductive failure begins at concentrations greater than 0.005 µg/L, with complete sterility occurring after chronic exposure of sensitive early life-stages at 0.009 µg/L. For less sensitive stages of these species imposex does not begin until 0.02 µg/L in some studies and greater than 0.2 µg/L in others. The long-term laboratory study by Harding et al. (1996) demonstrated a significant reproductive effect on *N. lapillus* at TBT concentrations >0.0074 µg/L. Therefore, EPA believes the Final Chronic Value should be lowered to 0.0074 µg/L to limit

unacceptable impacts on *Mercenaria mercenaria*, *Crassostrea gigas* and *Ostrea edulis* observed at 0.02 µg/L, and reproductive effects on *N. lapillus* at concentrations greater than 0.0074 µg/L. At this criterion concentration, imposex would be expected in *Ilyanassa obsoleta*, *N. lapillus* and similarly sensitive neogastropods and growth of *M. mercenaria* might be somewhat lowered.

UNUSED DATA

Data from some studies were not used in this document, as they did not meet the criteria for inclusion as specified in the Guidelines (Stephan et al. 1985). The reader is referred to the Guidelines for further information regarding these criteria.

Studies Were Conducted with Species That Are Not Resident in North America

Ali et al. (1990)	Jantataeme (1991)	Scadding (1990)
Allen et al. (1980)	Karande and Ganti (1994)	Scammell et al. (1991)
Axiak et al. (1995b)	Karande et al. (1993)	Seiffer and Schoof (1967)
Batley et al. (1989, 1992)	Kubo et al. (1984)	Shiff et al. (1975)
Burridge et al. (1995)	Langston and Burt (1991)	Shimizu and Kimura (1992)
Camey and Paulini (1964)	Lewis et al. (1995)	Smith et al. (1979)
Danil'chenko (1982)	Nagabhushanam et al. (1991)	Spence et al. (1990b)
Deschiens and Floch (1968)	Nagase et al. (1991)	Stebbing et al. (1990)
Deschiens et al. (1964, 1966a, 1966b)	Nias et al. (1993)	Sujatha et al. (1996)
de Sousa and Paulini (1970)	Nishuichi and Yoshida (1972)	Tsuda et al. (1986, 1991a)
Fent (1991, 1992)	Oehlmann et al. (1996)	Upatham (1975)
Fent and Hunn (1993)	Reddy et al. (1992)	Upatham et al. (1980a, 1980b)
Fent and Meier (1992)	Ringwood (1992)	Vitturi et al. (1992)
Frick and DeJimenez (1964)	Ritchie et al. (1964)	Webbe and Sturrock (1964)
Girard et al. (1996)	Ruiz et al. (1994a, 1994b, 1995a)	Yamada et al. (1994)
Helmstetter and Alden (1995)	1995b, 1995c, 1997)	Yla-Mononen (1989)
Hopf and Muller (1962)	Sarojini et al. (1991, 1992)	

Data Were Compiled From Other Sources

Alzieu (1986)	Cardwell and Vogue (1986)	Eisler (1989)
Cardarelli and Evans (1980)	Champ (1986)	Envirosphere Company (1986)
Cardwell and Sheldon (1986)	Chau (1986)	Evans and Leksono (1995)

Gibbs and Bryan (1987)	Kumpulainen and Koivistoinen (1977)	Seligman et al. (1986)
Gibbs et al. (1991a)	Lau (1991)	Slesinger and Dressler (1978)
Good et al. (1980)	Laughlin (1986)	Stebbing (1985)
Guard et al. (1982)	Laughlin and Linden (1985)	Thayer (1984)
Hall (1988, 1991)	Laughlin et al. (1984a)	Thompson et al. (1985)
Hall and Pinkney (1985)	McCullough et al. (1980)	U.S. EPA (1975, 1985b)
Hall et al. (1991)	Monaghan et al. (1980)	U.S. Navy (1984)
Hodge et al. (1979)	NC Dept. of Natural Resources and	von Rumker et al. (1974)
International Joint Commission (1976)	Community Dev. (1983,1985)	Walsh (1986)
Jensen (1977)	Rexrode (1987)	Zuckerman et al. (1978)
Kimbrough (1976)	Salazar (1989)	

The Test Procedures, Test Material or Results Were Not Adequately Described

Bruno and Ellis (1988)	EG&G Bionomics (1981b)	Mercier et al. (1994)
Cardwell and Stuart (1988)	Fileiko and Isakova (1980)	Nosov and Kolosova (1979)
Chau et al. (1983)	Holwerda and Herwig (1986)	Smith (1981c)
Danil'chenko and Buzinova (1982)	Kelly et al. (1990b)	Stroganov et al. (1972, 1977)
de la Court (1980)	Kolosova et al. (1980)	
Deschiens (1968)	Laughlin (1983)	

Studies by Gibbs et al. (1987) were not used because data were from the first year of a two-year experiment reported in Gibbs et al. (1988). Data from the life-cycle test with sheepshead minnows (Ward et al. 1981) were not used because ratios of measured and nominal concentrations were inconsistent within and between tests suggesting problems in delivering TBT, analytical chemistry or both. Results of some laboratory tests were not used because the tests were conducted in distilled or deionized water without addition of appropriate salts (e.g., Gras and Rioux 1965; Kumar-Das et al. 1984). The concentration of dissolved oxygen was too low in tests reported by EG&G Bionomics (1981a). Douglas et al. (1986) did not observe sufficient mortalities to calculate a useful LC50.

TBT Was a Component of a Formulation, Mixture, Paint or Sediment

Boike and Rathburn (1973)	Goss et al. (1979)	NC Dept. of Natural Resources and
Cardarelli (1978)	Henderson and Salazar (1996)	Community Dev. (1983)
Deschiens and Floch (1970)	Mattiessen and Thain (1989)	Pope (1981)

Quick and Cardarelli (1977)

Sherman (1983)

Walker (1977)

Salazar and Salazar (1985a, 1985b)

Sherman and Hoang (1981)

Weisfeld (1970)

Santos et al. (1977)

Sherman and Jackson (1981)

Data were not used when the organisms were exposed to TBT by injection or gavage (e.g., Fent and Stegeman 1991, 1993; Horiguchi et al. 1997; Rice et al. 1995; Rice and Weeks 1990; Rouleau et al. 1995). Caricchia et al. (1991), Salazar and Chadwick (1991), and Steinert and Pickwell (1993), did not identify the organism exposed to TBT. Some studies did not report toxic effects of TBT (e.g., Balls 1987; Gibbs 1993; Meador et al. 1984; Page 1995; Salazar 1986; Salazar and Champ 1988).

Results of tests in which enzymes, excised or homogenized tissue, or cell cultures were exposed to the test material were not used (e.g., Avery et al. 1993; Blair et al. 1982; Bruschiweiler et al. 1996; Falcioni et al. 1996; Fent and Bucheli 1994; Fent and Stegeman 1991; Fisher et al. 1990; Josephson et al. 1989; Joshi and Gupta 1990; Pickwell and Steinert 1988; Reader et al. 1994, 1996; Rice and Weeks 1991; Virkki and Nikinmaa 1993; Wishkovsky et al. 1989; Zucker et al. 1992).

Data were not used when the test organisms were infested with tapeworms (e.g., Hnath 1970). Mottley (1978) and Mottley and Griffiths (1977) conducted tests with a mutant form of an alga. Tests conducted with too few test organisms were not used (e.g., EG&G Bionomics 1976; Good et al. 1979). High control mortalities occurred in tests reported by Rhea et al. (1995), Salazar and Salazar (1989) and Valkirs et al. (1985). Some data were not used because of problems with the concentration of the test material (e.g., Springborn Bionomics 1984b; Stephenson et al. 1986; Ward et al. 1981) or low survival in the exposure organisms (Chagot et al. 1990; Fent and Looser 1995). BCFs were not used when the concentration of TBT in the test solution was not measured (Davies et al. 1986; Laughlin et al. 1986b; Paul and Davies 1986) or were highly variable (Becker et al. 1992; Laughlin and French 1988). Reports of the concentrations in wild aquatic animals were not used if concentrations in water were unavailable or excessively variable (e.g., Curtis and Barse 1990; Davies et al. 1987, 1988; Davies and McKie 1987; Gibbs et al. 1991b; Hall 1988; Han and Weber 1988; Kannan et al. 1996; Oehlmann et al. 1991; Stab et al. 1995; Thrower and Short 1991; Wade et al. 1988; Zuolian and Jensen 1989).

SUMMARY

Freshwater Acute Toxicity. The TBT acute toxicity values for twelve freshwater animal species range from 1.14 µg/L for a hydra, *Hydra oligactis*, to 12.73 µg/L for the lake trout, *Salvelinus namaycush*. A thirteenth species, a clam (*Elliptio complanatus*), had an exceptionally high toxicity

value of 24,600 µg/L. There was no apparent trend in sensitivities with taxonomy; fish were nearly as sensitive as the most sensitive invertebrates and more sensitive than others. When the much less sensitive clam was not considered, remaining species sensitivities varied by a maximum of 11.2 times. Plants were about as sensitive as animals to TBT.

Freshwater Chronic Toxicity. Three TBT chronic toxicity tests have been conducted with freshwater animals. Reproduction of *D. magna* was reduced by 0.2 µg/L, but not by 0.1 µg/L, and the Acute-Chronic Ratio is 30.41. In another test with *D. magna* reproduction and survival was reduced at 0.34 µg/L but not at 0.19, and the Acute-Chronic Ratio is 44.06. Weight of fathead minnows was reduced by 0.45 µg/L, but not by 0.15 µg/L, and the acute-chronic ratio for this species was 10.01. Bioconcentration of TBT was measured in zebra mussels, *Dreissena polymorpha*, at 17,483 times the water concentration for the soft parts and in rainbow trout, *Oncorhynchus mykiss*, at 406 times the water concentration for the whole body. Growth of thirteen species of freshwater algae was inhibited by concentrations ranging from 1 to 1,782 µg/L.

Saltwater Acute Toxicity. Acute values for 33 species of saltwater animals range from 0.61 µg/L for the mysid, *Acanthomysis sculpta*, to 204.4 µg/L for adult European flat oysters, *Ostrea edulis*. Acute values for the ten most sensitive genera, including molluscs, crustaceans, and fishes, differ by less than a factor of four. Larvae and juveniles appear to be more acutely sensitive to TBT than adults.

Saltwater Chronic Toxicity. A partial life-cycle test of one-year duration was conducted with the snail, *Nucella lapillus*. TBT reduced egg capsule production. The chronic value for this species was 0.0143 µg/L. A life-cycle test was conducted with the copepod, *Eurytemora affinis*. The chronic value is based upon neonate survival and is 0.145 µg/L and the acute/chronic ratio is 15.17. A life-cycle toxicity test was conducted with the saltwater mysid, *Acanthomysis sculpta*. The chronic value for *A. sculpta* was 0.1308 µg/L based on reduced reproduction and the acute-chronic ratio was 4.664.

Bioconcentration. Bioconcentration factors for three species of bivalve molluscs range from 192.3 for soft parts of the European flat oyster to 60,000 for soft parts of the juvenile blue mussel, *Mytilus edulis*.

Acute-Chronic Ratio. The Final Acute-Chronic Ratio of 12.69 was calculated as the geometric mean of the acute-chronic ratios of 36.60 for *D. magna*, 10.01 for *P. promelas* (the two freshwater species), and 4.664 for *A. sculpta* and 15.17 for *E. affinis* (the two saltwater species). Division of the freshwater and saltwater Final Acute Values by 12.69 results in Final Chronic Values for freshwater of 0.0723 µg/L and for saltwater of 0.0658 µg/L (Table 3). The Chronic Values are below the

experimentally determined chronic values from life-cycle or early life-stage tests with freshwater species (0.1414 µg/L for *D. magna*), but almost five times higher than the chronic value for *N. lapillus* of 0.0143 µg/L.

Tributyltin chronically affects certain saltwater copepods, gastropods, and pelecypods at concentrations less than those predicted from "standard" acute and chronic toxicity tests. The data demonstrate that reductions in growth occur in commercially or ecologically important saltwater species at concentrations of TBT less than the Final Chronic Value of 0.0658 µg/L derived using Final Acute Values and Acute-Chronic Ratios from Table 3. Survival of the copepod *A. tonsa* was reduced in \$0.023 µg/L. Growth of larvae or spat of two species of oysters, *Crassostrea gigas* and *Ostrea edulis* was reduced in about 0.02 µg/L; some *C. gigas* larvae died in 0.025 µg/L. Shell thickening and reduced meat weights were observed in *C. gigas* at 0.01 µg/L. Reproductive effects were observed with *N. lapillus* at TBT concentrations >0.0074 µg/L. Since these levels were ones at which an effect was seen, a protective level for these commercially important species is, therefore, below 0.01 µg/L.

Weight of Evidence Considerations. The National Guidelines (Stephan et al. 1985) require that the criterion be lowered if sound scientific evidence indicates that adverse effects might be expected on important species. The above data demonstrate that the reductions in growth occur in commercially or ecologically important saltwater species at concentrations of TBT less than the final chronic value of 0.0658 µg/L derived using Final Acute Values and Acute-Chronic ratios from Table 3. Consistent with the Guidelines directive to consider other relevant data when establishing criteria, EPA believes the Final Chronic Value should be lowered to 0.0074 µg/L.

Organometallics, particularly TBT and methyl mercury, have been shown to impair the environment in multiple ways. A major concern with TBT is its ability to cause imposex (the superimposition of male anatomical characteristics on females) in a variety of species. Imposex has been observed in 45 species of snails worldwide, with definitive laboratory and field studies implicating TBT as the cause in seven North American or cosmopolitan species. As listed in Table 6, adult dogwhelk, *Nucella lapillus*, exposed to 0.05 µg/L TBT for 120 days showed 41 percent of the organisms evidencing imposex. A six month study of the same species in 1992 with a concentration of 0.012 µg/L TBT also showed imposex in the organisms. Other studies showed more than 92 percent of the female *N. lapillus* exposed to TBT at 0.0027 µg/L exhibiting imposex; a follow up study of offspring showed almost 99 percent imposex in females at TBT concentrations of 0.0026 µg/L. The long-term laboratory study by Harding et al. (1996) demonstrated significant reproductive effects on *N. lapillus* at TBT concentrations >0.0074 µg/L. Thus, numerous studies show imposex effects at doses well below

the calculated saltwater Final Chronic Value of 0.0658 µg/L. Many of the studies did not produce a No Observed Effect Level because significant effects were observed at the lowest concentration tested. The imposex effect may partially explain the results of the studies in Tables 2 and 6 which show abnormal growth patterns seen in other studies, including reduced growth, shell thickening and deformities. Imposex has also been linked with population declines of snails in Canada (Tester et al. 1996) and oysters in the United Kingdom (Dyrynda 1992 and others); these declines were reversed after restrictions on TBT use went into effect.

Another factor causing increased concern is the very high bioaccumulation and bioconcentration factors associated with TBT. For some species, these factors reach into the thousands and tens of thousands. Data are summarized in Table 5. They show BCF/BAF factors in the thousands for rainbow trout, *Oncorhynchus mykiss*, where TBT concentrations were approximately 1.0 µg/L, and in goldfish, *Crassius auratus*, where TBT concentrations were approximately 0.1 µg/L. For saltwater species, field studies of blue mussels, *Mytilus edulis*, at TBT concentrations of <0.1 µg/L, showed BCF or BAF concentrations up to 60,000 (Salazar and Salazar 1990a, 1991); the eastern oyster, *Crassostrea virginica*, exhibited factors of 15,000 in TBT concentrations of <0.3 µg/L (Roberts et al. 1996); and the Pacific oyster, *Crassostrea gigas*, had factors in the thousands when exposed to TBT in concentrations from 0.24 to 1.5 µg/L.

Immunological effects have been observed in eastern oysters exposed to 0.03 µg/L TBT which resulted in increased infection intensity and mortality when later exposed to “Dermo”, a protozoan pathogen. TBT is widely assumed to enhance the impairment caused by Dermo. However, data are currently insufficient to determine which levels of Dermo and of TBT result in this heightened interaction. Because levels of both Dermo and TBT are known to fluctuate widely, it is prudent in the face of this uncertainty regarding impact on a commercially important species to be conservative when establishing acceptable levels.

Conclusion. The development of a chronic criterion for TBT in saltwater considers four lines of evidence. It considers the traditional endpoints of adverse effects on survival, growth and reproduction as demonstrated in numerous laboratory studies; recognizing that a number of these studies have unbounded LOAELs at or near 0.01 µg/L, and recognizing further that only one study included levels below 0.01 µg/L and that study (on *Acartia tonsa* at 0.003 µg/L) showed inhibition of development. It considers the production of imposex in field studies and the impact of imposex on commercially significant species population levels. It also considers that TBT accumulates and/or concentrates in commercially and recreationally important freshwater and saltwater species. Finally, it considers the

potential immunological effects of TBT, as well as the finding that an important commercial organism (eastern oyster) already known to be vulnerable to the prevalent pathogen Dermo was made even more vulnerable by prior exposure to TBT.

Considering the traditional endpoints of adverse effects on survival, growth and reproduction, the Final Chronic Value would be set at 0.066 µg/L. However, the Agency believes that this level would not be protective for the additional factors cited above. The Agency is faced with the uncertainty created by the lack of understanding of the relationship of these factors, the low level at which effects are occurring, the lack of data below these levels enabling a definitive calculation of an acceptable exposure level, and the commercial viability of some of the species. Therefore, in this situation, the Agency is making a decision to establish the Final Chronic Value at 0.0074 µg/L in the belief that this level more closely approximates an acceptable level of exposure.

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 tributyltin does not exceed 0.072 µg/L more than once every three years on the average and if the one-hour average concentration does not exceed 0.46 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 tributyltin does not exceed 0.0074 µg/L more than once every three years on the average and if the one-hour average concentration does not exceed 0.42 µg/L more than once every three years on the average.

IMPLEMENTATION

As discussed in the Water Quality Standards Regulation (U.S. EPA 1983) and the Foreword of this document, a water quality criterion for aquatic life has regulatory impact only if it has been adopted in a state water quality standard. Such a standard specifies a criterion for a pollutant that is consistent with a particular designated use. With the concurrence of the U.S. EPA, states designate one or more uses for each body of water or segment thereof and adopt criteria that are consistent with the use(s) (U.S. EPA 1987,1994). Water quality criteria adopted in state water quality standards could have the same numerical values as criteria developed under Section 304, of the Clean Water Act. However, in many situations states might want to adjust water quality criteria developed under Section 304 to reflect local environmental conditions and human exposure patterns. Alternatively, states may use different data and assumptions than EPA in deriving numeric criteria that are scientifically defensible and protective of designated uses. State water quality standards include both numeric and narrative criteria. A state may adopt a numeric criterion within its water quality standards and apply it either state-wide to all waters designated for the use the criterion is designed to protect or to a specific site. A state may use an indicator parameter or the national criterion, supplemented with other relevant information, to interpret its narrative criteria within its water quality standards when developing NPDES effluent limitations under 40 CFR 122.44(d)(1)(vi).2

Site-specific criteria may include not only site-specific criterion concentrations (U.S. EPA 1994), but also site-specific, and possibly pollutant-specific, durations of averaging periods and frequencies of allowed excursions (U.S. EPA 1991). The averaging periods of "one hour" and "four days" were selected by the U.S. EPA on the basis of data concerning the speed with which some aquatic species can react to increases in the concentrations of some aquatic pollutants, and "three years" is the Agency's best scientific judgment of the average amount of time aquatic ecosystems should be provided between excursions (Stephan et al. 1985; U.S. EPA 1991). However, various species and ecosystems react and recover at greatly differing rates. Therefore, if adequate justification is provided, site-specific and/or pollutant-specific concentrations, durations, and frequencies may be higher or lower than those given in national water quality criteria for aquatic life.

Use of criteria, which have been adopted in state water quality standards, for developing water quality-based permit limits and for designing waste treatment facilities requires selection of an appropriate wasteload allocation model. Although dynamic models are preferred for the application of these criteria (U.S. EPA 1991), limited data or other considerations might require the use of a steady-state model (U.S. EPA 1986).

Guidance on mixing zones and the design of monitoring programs is also available (U.S. EPA 1987, 1991).

Figure 1. Ranked Summary of Tributyltin GMAVs - Freshwater.

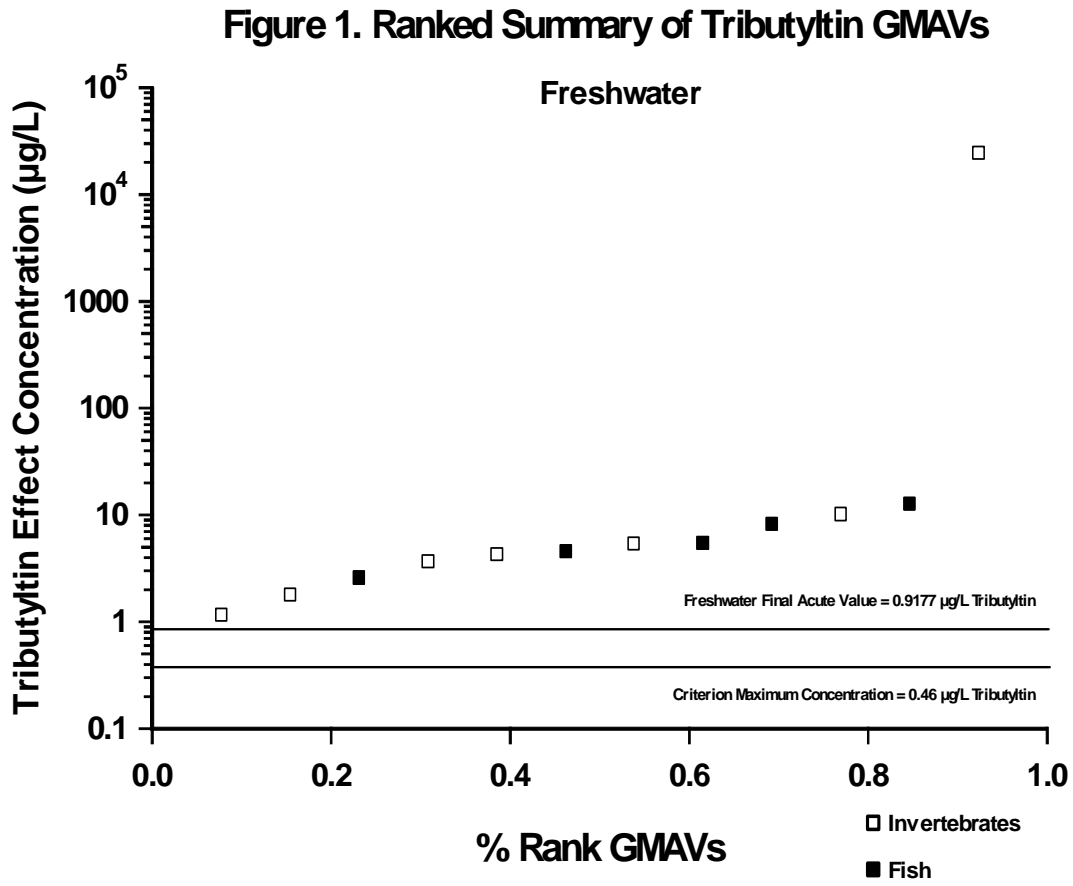


Figure 2. Ranked Summary of Tributyltin GMAVs - Saltwater.

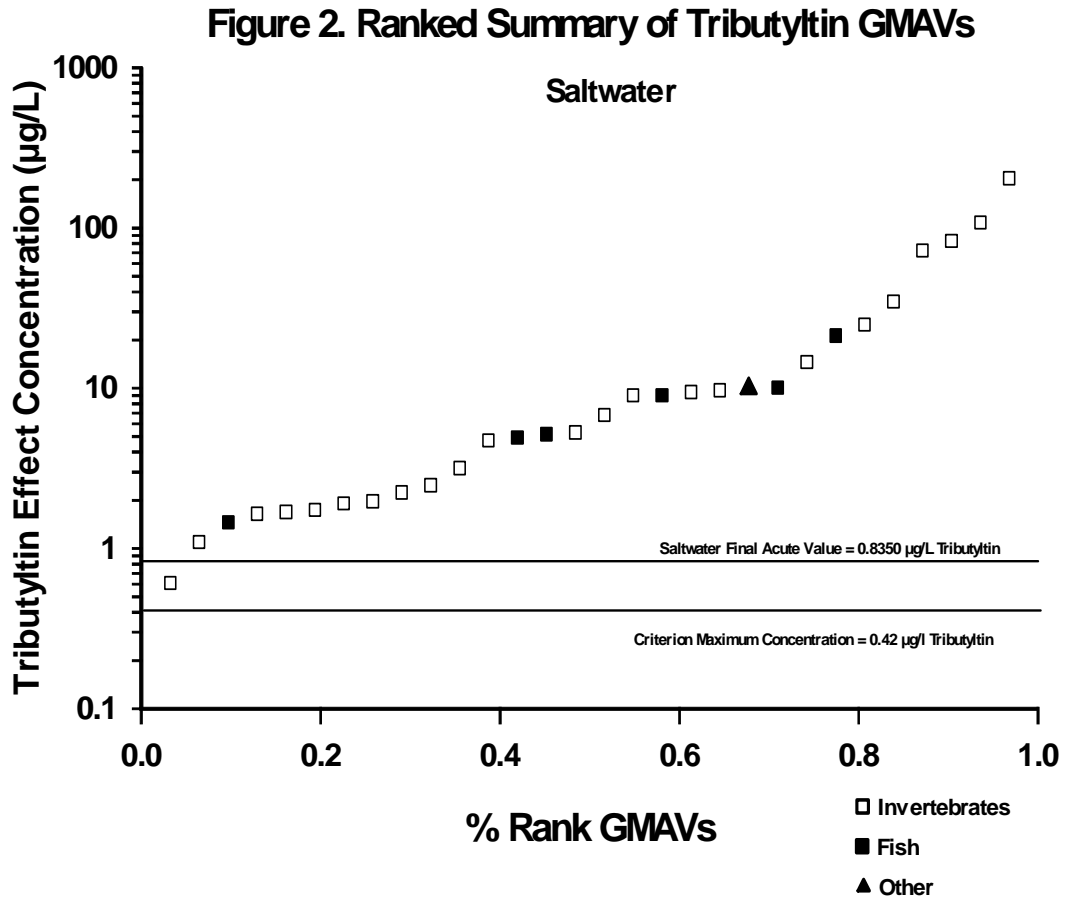


Figure 3. Chronic Toxicity of Tributyltin to Aquatic Animals.

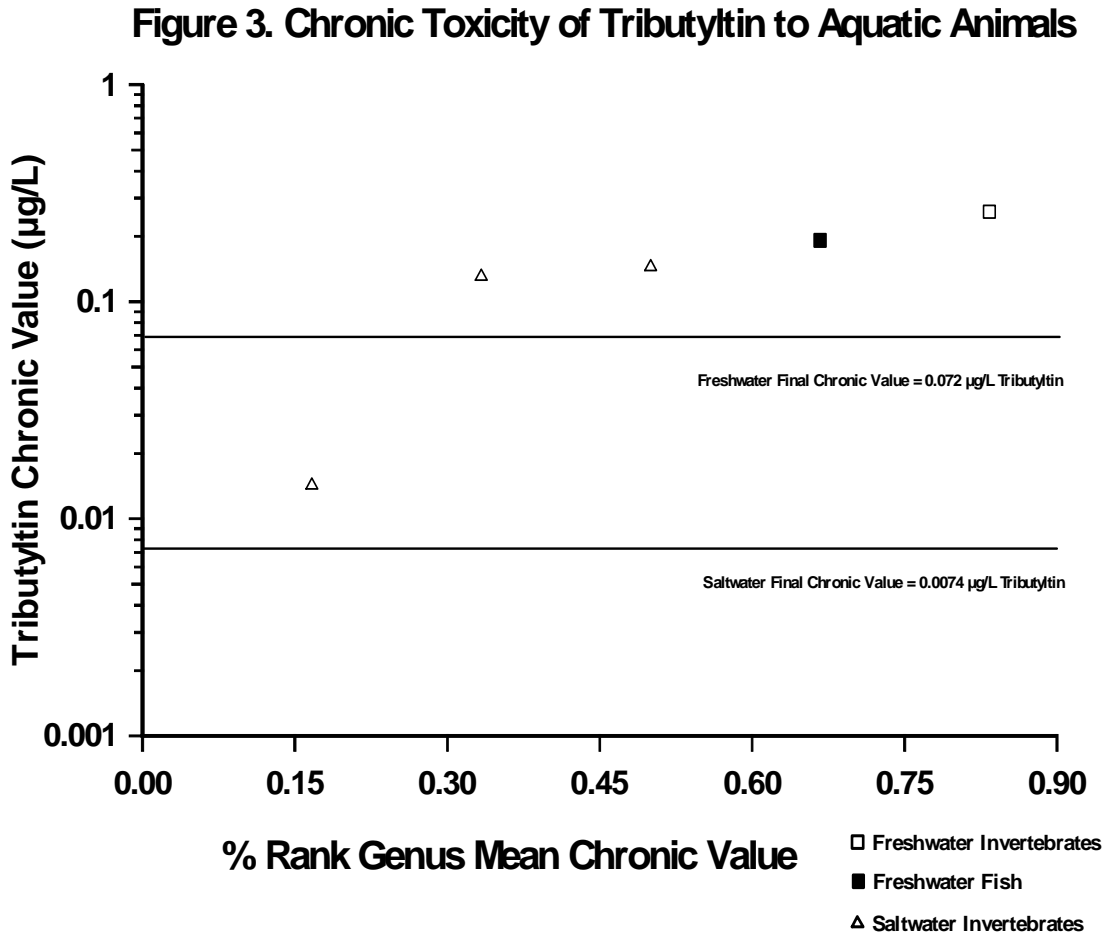


Table 1. Acute Toxicity of Tributyltin to Aquatic Animals

<u>Species</u>	<u>Method</u> ^a	<u>Chemical</u> ^b	<u>Hardness</u> (mg/L as CaCO ₃)	<u>LC50</u> or <u>EC50</u> (<u>µg/L</u>) ^c	<u>Species Mean</u> <u>Acute Value</u> (<u>µg/L</u>)	<u>References</u>
<u>FRESHWATER SPECIES</u>						
Hydra, <i>Hydra littoralis</i>	S,M	TBTO (97.5%)	100	<u>1.11</u>	-	TAI Environmental Sciences, Inc. 1989a
Hydra, <i>Hydra littoralis</i>	S,M	TBTO (97.5%)	120	<u>1.30</u>	1.201	TAI Environmental Sciences, Inc. 1989b
Hydra, <i>Hydra oligactis</i>	S,M	TBTO (97.5%)	100	<u>1.14</u>	1.14	TAI Environmental Sciences, Inc. 1989a
Hydra, <i>Chlorohydra</i> <i>viridissima</i>	S,M	TBTO (97.5%)	120	<u>1.80</u>	1.80	TAI Environmental Sciences, Inc. 1989b
Annelid (9 mg), <i>Lumbriculus</i> <i>variegatus</i>	F,M	TBTO (96%)	51.8	<u>5.4</u>	5.4	Brooke et al. 1986
Freshwater clam, (113 mm TL; 153 g) <i>Elliptio complanatus</i>	S,U	TBTO (95%)	-	<u>24,600</u>	24,600	Buccafusco 1976a
Cladoceran, <i>Daphnia magna</i>	S,U	TBTO	-	66.3	-	Foster 1981
Cladoceran (adult), <i>Daphnia magna</i>	S,U	TBTCl	-	5.26	-	Meador 1986
Cladoceran (<24 hr), <i>Daphnia magna</i>	S,U	TBTO (95%)	-	1.58	-	LeBlanc 1976
Cladoceran (<24 hr), <i>Daphnia magna</i>	R,M	TBTO (97.5%)	172	11.2	-	ABC Laboratories, Inc. 1990c
Cladoceran (<24 hr), <i>Daphnia magna</i>	S,U	TBTCl	250	18	-	Crisinel et al. 1994

Table 1. Acute Toxicity of Tributyltin to Aquatic Animals (continued)

<u>Species</u>	<u>Method^a</u>	<u>Chemical^b</u>	<u>Hardness (mg/L as CaCO₃)</u>	<u>LC50 or EC50 (µg/L)^c</u>	<u>Species Mean Acute Value (µg/L)</u>	<u>References</u>
<u>FRESHWATER SPECIES</u>						
Cladoceran (<24 hr), <i>Daphnia magna</i>	F,M	TBTO (96%)	51.5	<u>4.3</u>	4.3	Brooke et al. 1986
Amphipod, <i>Gammarus pseudolimnaeus</i>	F,M	TBTO (96%)	51.8	<u>3.7</u>	3.7	Brooke et al. 1986
Mosquito (larva), <i>Culex</i> sp.	S,M	TBTO (96%)	51.5	<u>10.2</u>	10.2	Brooke et al. 1986
Rainbow trout, (45 mm TL; 0.68g) <i>Oncorhynchus mykiss</i>	S,U	TBTO (95%)	-	6.5	-	Buccafusco et al. 1978
Rainbow trout (juvenile), <i>Oncorhynchus mykiss</i>	F,M	TBTO (96%)	50.6	<u>3.9</u>	-	Brooke et al. 1986
Rainbow trout (1.47 g), <i>Oncorhynchus mykiss</i>	F,M	TBTO (97%)	135	<u>3.45</u>	-	Martin et al. 1989
Rainbow trout (1.4g), <i>Oncorhynchus mykiss</i>	F,M	TBTO (97.5%)	44	<u>7.1</u>	4.571	ABC Laboratories, Inc. 1990a
Lake trout (5.94 g), <i>Salvelinus naymaycush</i>	F,M	TBTO (97%)	135	<u>12.73</u>	12.73	Martin et al. 1989
Fathead minnow (juvenile), <i>Pimephales promelas</i>	F,M	TBTO (96%)	51.5	<u>2.6</u>	2.6	Brooke et al. 1986
Channel catfish, (65 mm TL; 1.9 g) <i>Ictalurus punctatus</i>	S,U	TBTO (95%)	-	11.4	-	Buccafusco 1976a

FRESHWATER SPECIES

Table 1. Acute Toxicity of Tributyltin to Aquatic Animals (continued)

<u>Species</u>	<u>Method^a</u>	<u>Chemical^b</u>	<u>Hardness (mg/L as CaCO₃)</u>	<u>LC50 or EC50 (µg/L)^c</u>	<u>Species Mean Acute Value (µg/L)</u>	<u>References</u>
Channel catfish (juvenile), <i>Ictalurus punctatus</i>	F,M	TBTO (96%)	51.8	<u>5.5</u>	5.5	Brooke et al. 1986
Bluegill, <i>Lepomis macrochirus</i>	S,U	TBTO	-	227.4	-	Foster 1981
Bluegill, (36 mm TL: 0.67 g) <i>Lepomis macrochirus</i>	S,U	TBTO (95%)	-	7.2	-	Buccafusco 1976b
Bluegill (1.01 g), <i>Lepomis macrochirus</i>	F,M	TBTO (97.5%)	44	<u>8.3</u>	8.3	ABC Laboratories, Inc. 1990b
<u>Species</u>	<u>Method^a</u>	<u>Chemical^b</u>	<u>Salinity (g/kg)</u>	<u>LC50 or EC50 (µg/L)^c</u>	<u>Species Mean Acute Value (µg/L)</u>	<u>References</u>
<u>SALTWATER SPECIES</u>						
Lugworm (larva), <i>Arenicola cristata</i>	S,U	TBTO	28	<u>- 2-4</u>	-	Walsh et al. 1986b
Lugworm (larva), <i>Arenicola cristata</i>	S,U	TBTA	28	<u>- 5-10</u>	- 4.74	Walsh et al. 1986b
Polychaete (adult), <i>Neanthes arenaceodentata</i>	S,M	TBTO	33-34	21.41 ^d	-	Salazar and Salazar 1989
Polychaete (juvenile), <i>Neanthes arenaceodentata</i>	S,M	TBTO	33-34	<u>6.812</u>	6.812	Salazar and Salazar 1989
Polychaete (adult), <i>Armandia brevis</i>	R,M	TBTCl (96%)	28.5	<u>25</u>	25	Meador 1997
Blue mussel (adult), <i>Mytilus edulis</i>	R,-	TBTO	-	36.98 ^d	-	Thain 1983
Blue mussel (adult), <i>Mytilus edulis</i>	S,M	TBTO	33-34	34.06 ^d	-	Salazar and Salazar 1989

Table 1. Acute Toxicity of Tributyltin to Aquatic Animals (continued)

<u>Species</u>	<u>Method</u> ^a	<u>Chemical</u> ^b	<u>Salinity</u> <u>(g/kg)</u>	<u>LC50</u> <u>or EC50</u> <u>(µg/L)</u> ^c	<u>Species Mean</u> <u>Acute Value</u> <u>(µg/L)</u>	<u>References</u>
<u>SALTWATER SPECIES</u>						
Blue mussel (larva), <i>Mytilus edulis</i>	R,-	TATO	-	<u>2.238</u>	2.238	Thain 1983
Pacific oyster (adult), <i>Crassostrea gigas</i>	R,-	TBTO	-	282.2 ^d	-	Thain 1983
Pacific oyster (larva), <i>Crassostrea gigas</i>	R,-	TBTO	-	<u>1.557</u>	1.557	Thain 1983
Eastern oyster, <i>Crassostrea virginica</i>	R,U	TBTCl	18-22	<u>3.96</u>	3.96	Roberts 1987
European flat oyster (adult), <i>Ostrea edulis</i>	R,-	TBTO	-	<u>204.4</u>	204.4	Thain 1983
Atlantic dogwhelk (<24 hr-old), <i>Nucella lapillus</i>	R,M	TBTO	34-35	<u>72.7</u>	72.7	Harding et al. 1996
Hard clam (larva), <i>Mercenaria mercenaria</i>	R,U	TBTCl	18-22	<u>1.65</u>	1.65	Roberts 1987
Copepod (juvenile), <i>Eurytemora affinis</i>	F,M	TBTCl	10.6	<u>2.2</u>	-	Hall et al. 1988a
Copepod (subadult), <i>Eurytemora affinis</i>	F,M	TBT	10	<u>2.5</u>	-	Bushong et al. 1987;1988
Copepod (subadult), <i>Eurytemora affinis</i>	F,M	TBT	10	<u>1.4</u>	1.975	Bushong et al. 1987;1988
Copepod (adult), <i>Acartia tonsa</i>	R,U	TBTO (95%)	-	0.6326	-	U'ren 1983
Copepod (10-12-d-old), <i>Acartia tonsa</i>	S,U	TBTCl (99.3%)	18	0.47	-	Kusk and Petersen 1997
Copepod (10-12-d-old), <i>Acartia tonsa</i>	S,U	TBTCl (99.3%)	28	0.24	-	Kusk and Petersen 1997
Copepod (subadult), <i>Acartia tonsa</i>	F,M	TBT	10	<u>1.1</u>	1.1	Bushong et al. 1987;1988
Copepod (adult), <i>Nitocra spinipes</i>	S,U	TBTF	7	<u>1.877</u>	-	Linden et al. 1979

Table 1. Acute Toxicity of Tributyltin to Aquatic Animals (continued)

<u>Species</u>	<u>Method</u> ^a	<u>Chemical</u> ^b	<u>Salinity</u> <u>(g/kg)</u>	<u>LC50</u> <u>or EC50</u> <u>(µg/L)</u> ^c	<u>Species Mean</u> <u>Acute Value</u> <u>(µg/L)</u>	<u>References</u>
<u>SALTWATER SPECIES</u>						
Copepod (adult), <i>Nitocra spinipes</i>	S,U	TBTO	7	<u>1.946</u>	1.911	Linden et al. 1979
Mysid (juvenile), <i>Acanthomysis sculpta</i>	R,M	- ^e	-	0.42	-	Davidson et al. 1986a,1986b
Mysid (adult), <i>Acanthomysis sculpta</i>	F,M	- ^e	-	1.68 ^d	-	Valkirs et al. 1985
Mysid (juvenile), <i>Acanthomysis sculpta</i>	F,M	- ^e	-	<u>0.61</u>	0.61	Valkirs et al. 1985
Mysid (subadult), <i>Metamysidopsis</i> <i>elongata</i>	S,M	TBTO	33-34	<u>1.946</u>	-	Salazar and Salazar 1989
Mysid (adult), <i>Metamysidopsis</i> <i>elongata</i>	S,M	TBTO	33-34	<u>2.433</u>	-	Salazar and Salazar 1989
Mysid (adult), <i>Metamysidopsis</i> <i>elongata</i>	S,M	TBTO	33-34	<u>6.812</u>	3.183	Salazar and Salazar 1989
Mysid (<1 day), <i>Americamysis bahia</i>	F,M	TBTCl	19-22	<u>1.1</u>	-	Goodman et al. 1988
Mysid (5 day), <i>Americamysis bahia</i>	F,M	TBTCl	19-22	<u>2.0</u>	-	Goodman et al. 1988
Mysid (10 day), <i>Americamysis bahia</i>	F,M	TBTCl	19-22	<u>2.2</u>	1.692	Goodman et al. 1988
Amphipod (adult), <i>Gammarus</i> sp.	F,M	TBT	10	<u>5.3</u>	5.3	Bushong et al. 1988
Amphipod (adult), <i>Orchestia traskiana</i>	R,M	TBTO	30	<u>>14.60^f</u>	>14.60	Laughlin et al. 1982
Amphipod (adult), <i>Rhepoxynius</i> <i>abronius</i>	R,M	TBTCl (96%)	32.3	<u>108</u>	108	Meador 1997
Amphipod (3-5 mm; 2-5 mg), <i>Eohaustorius</i> <i>estuaris</i>	R,M	TBTCl (96%)	28.8-29.5	<u>10</u>	10	Meador 1993; Meador et al. 1993; Meador 1997

Table 1. Acute Toxicity of Tributyltin to Aquatic Animals (continued)

<u>Species</u>	<u>Method</u> ^a	<u>Chemical</u> ^b	<u>Salinity</u> <u>(g/kg)</u>	<u>LC50</u> <u>or EC50</u> <u>(µg/L)</u> ^c	<u>Species Mean</u> <u>Acute Value</u> <u>(µg/L)</u>	<u>References</u>
<u>SALTWATER SPECIES</u>						
Amphipod (adult), <i>Eohaustorius</i> <i>washingtonianus</i>	R,M	TBTCI (96%)	32.7	<u>9</u>	9	Meador 1997
Grass shrimp (adult), <i>Palaemonetes pugio</i>	F,U	TBTO	-	<u>20</u>	20	Clark et al. 1987
Grass shrimp (subadult), <i>Palaemonetes</i> sp.	F,M	TBT	10	>31 ^d	-	Bushong et al. 1988
Grass shrimp (adult), <i>Palaemonetes</i> sp.	R,U	TBTO	20	31.41 ^d	-	Kahn et al. 1993
Grass shrimp (larva), <i>Palaemonetes</i> sp.	R,U	TBTO	20	<u>4.07</u>	4.07	Kahn et al. 1993
American lobster (larva), <i>Homarus americanus</i>	R,U	TBTO	32	<u>1.745</u> ^f	1.745	Laughlin and French 1980
Shore crab (larva), <i>Carcinus maenas</i>	R,-	TBTO	-	<u>9.732</u>	9.732	Thain 1983
Mud crab (larva), <i>Rhithropanopeus</i> <i>harrisii</i>	R,U	TBTS	15	>24.3 ^f	-	Laughlin et al. 1983
Mud crab (larva), <i>Rhithropanopeus</i> <i>harrisii</i>	R,U	TBTO	15	<u>34.90</u> ^f	34.90	Laughlin et al. 1983
Shore crab (larva), <i>Hemigrapsus nudus</i>	R,U	TBTO	32	<u>83.28</u> ^f	83.28	Laughlin and French 1980
Amphioxus, <i>Branchiostoma</i> <i>caribaeum</i>	F,U	TBTO	-	<u><10</u>	<10	Clark et al. 1987
Chinook salmon (juvenile), <i>Oncorhynchus</i> <i>tshawytscha</i>	S,M	TBTO	28	<u>1.460</u>	1.460	Short and Thrower 1986b;1987
Atlantic menhaden (juvenile), <i>Brevoortia tyrannus</i>	F,M	TBT	10	<u>4.7</u>	-	Bushong et al. 1987;1988

Table 1. Acute Toxicity of Tributyltin to Aquatic Animals (continued)

<u>Species</u>	<u>Method</u> ^a	<u>Chemical</u> ^b	<u>Salinity</u> <u>(g/kg)</u>	<u>LC50</u> <u>or EC50</u> <u>(µg/L)</u> ^c	<u>Species Mean</u> <u>Acute Value</u> <u>(µg/L)</u>	<u>References</u>
<u>SALTWATER SPECIES</u>						
Atlantic menhaden (juvenile), <i>Brevoortia tyrannus</i>	F,M	TBT	10	<u>5.2</u>	4.944	Bushong et al. 1987;1988
Sheepshead minnow (juvenile), <i>Cyprinodon</i> <i>variegatus</i>	S,U	TBTO	20	16.54	-	EG&G Bionomics 1979
Sheepshead minnow (juvenile), <i>Cyprinodon</i> <i>variegatus</i>	S,U	TBTO	20	16.54	-	EG&G Bionomics 1979
Sheepshead minnow (juvenile), <i>Cyprinodon</i> <i>variegatus</i>	S,U	TBTO	20	12.65	-	EG&G Bionomics 1979
Sheepshead minnow (33-49 mm), <i>Cyprinodon</i> <i>variegatus</i>	F,M	TBTO	28-32	<u>2.315</u> ^f	-	EG&G Bionomics 1981d
Sheepshead minnow (juvenile), <i>Cyprinodon</i> <i>variegatus</i>	F,M	TBTO	15	<u>12.31</u>	-	Walker 1989a
Sheepshead minnow (subadult), <i>Cyprinodon</i> <i>variegatus</i>	F,M	TBT	10	<u>25.9</u>	9.037	Bushong et al. 1988
Mummichog (adult), <i>Fundulus</i> <i>heteroclitus</i>	S,U	TBTO (95%)	25	23.36	-	EG&G Bionomics 1976
Mumichog (juvenile), <i>Fundulus</i> <i>heteroclitus</i>	F,M	TBTO	2	<u>17.2</u>	-	Pinkney et al. 1989a,b
Mummichog (larva), <i>Fundulus</i> <i>heteroclitus</i>	F,M	TBT	10	<u>23.4</u>	-	Bushong et al. 1988

Table 1. Acute Toxicity of Tributyltin to Aquatic Animals (continued)

<u>Species</u>	<u>Method</u> ^a	<u>Chemical</u> ^b	<u>Salinity</u> <u>(g/kg)</u>	<u>LC50</u> <u>or EC50</u> <u>(µg/L)</u> ^c	<u>Species Mean</u> <u>Acute Value</u> <u>(µg/L)</u>	<u>References</u>
<u>SALTWATER SPECIES</u>						
Mummichog (subadult), <i>Fundulus</i> <i>heteroclitus</i>	F,M	TBT	10	<u>23.8</u>	21.34	Bushong et al. 1988
Inland silverside (larva), <i>Menidia beryllina</i>	F,M	TBT	10	<u>3.0</u>	3.0	Bushong et al. 1987;1988
Atlantic silverside, <i>Menidia menidia</i>	F,M	TBT	10	<u>8.9</u>	8.9	Bushong et al. 1987;1988
Starry flounder (<1-year-old), <i>Platichthys stellatus</i>	R,M	TBTCl (96%)	30.2	<u>10.1</u>	10.1	Meador 1997

^a S = static; R = renewal; F = flow-through; M = measured; U = unmeasured.

^b TBTCI = tributyltin chloride; TBTF = tributyltin fluoride; TBTO = tributyltin oxide; TBTS = tributyltin sulfide. Percent purity is given in parentheses when available.

^c Concentration of the tributyltin cation, not the chemical. If the concentrations were not measured and the published results were not reported to be adjusted for purity, the published results were multiplied by the purity if it was reported to be less than 95 percent. **Note:** The values underlined in this column were used to calculate the SMAV for the respective species.

^d Value not used in determination of Species Mean Acute Value because data are available for a more sensitive life stage.

^e The test organisms were exposed to leachate from panels coated with antifouling paint containing a tributyltin polymer and cuprous oxide. Concentrations of TBT were measured and the authors provided data to demonstrate the similar toxicity of a pure TBT compound and the TBT from the paint formulation.

^f LC50 and EC50 calculated or interpolated graphically based on the authors' data.

Table 2a. Chronic Toxicity of Tributyltin to Aquatic Animals.

<u>Species</u>	<u>Test</u> ^a	<u>Chemical</u> ^b	<u>Hardness</u> (mg/L as CaCO ₃)	<u>Chronic</u> <u>Limits</u> (µg/L) ^c	<u>Chronic</u> <u>Value</u> (µg/L)	<u>References</u>
<u>FRESHWATER SPECIES</u>						
Cladoceran, <i>Daphnia magna</i>	LC	TBTO (96%)	51.5	0.1-0.2	0.1414	Brooke et al. 1986
Cladoceran, <i>Daphnia magna</i>	LC	TBTO (100%)	160-174	0.19-0.34	0.2542	ABC Laboratories, Inc. 1990d
Fathead minnow, <i>Pimephales</i> <i>promelas</i>	ELS	TBTO (96%)	51.5	0.15-0.45	0.2598	Brooke et al. 1986
<u>SALTWATER SPECIES</u>						
Atlantic dogwhinkle, <i>Nucella lapillus</i>	ELS	TBTO (97%)	34-35	0.0074-0.0278 ^f	0.0143	Harding et al. 1996
Copepod, <i>Eurytemora</i> <i>affinis</i>	LC	TBTCl	10.3 ^e	<0.088	<0.088	Hall et al. 1987;1988a
Copepod, <i>Eurytemora</i> <i>affinis</i>	LC	TBTCl	14.6 ^e	0.094-0.224	0.145	Hall et al. 1987;1988a
Mysid, <i>Acanthomysis</i> <i>sculpta</i>	LC	- ^d	-	0.09-0.19	0.1308	Davidson et al. 1986a,1986b

^a LC = Life-cycle or partial life-cycle; ELS = early life-stage.

^b TBTO = tributyltin oxide; TBTCl = tributyltin chloride. Percent purity is given in parentheses when available.

^c Measured concentrations of the tributyltin cation.

^d The test organisms were exposed to leachate from panels coated with antifouling paint containing a tributyltin polymer and cuprous oxide. Concentrations of TBT were measured and the authors provided data to demonstrate the similar toxicity of a pure TBT compound and the TBT from the paint formulation.

^e Salinity (g/kg).

^f TBT concentrations are those reported by Bailey et al. (1991). See text for explanation.

Table 2b. Acute-Chronic Ratios

Acute-Chronic Ratios					
<u>Species</u>	<u>Hardness (mg/L as CaCO₃)</u>	<u>Acute Value (µg/L)</u>	<u>Chronic Value (µg/L)</u>	<u>Ratio</u>	<u>Reference</u>
Cladoceran, <i>Daphnia magna</i>	51.5	4.3	0.1414	30.41	Brooke et al. 1986
Cladoceran, <i>Daphnia magna</i>	160-174	11.2	0.2542	44.06	ABC Laboratories, Inc. 1990d
Fathead minnow, <i>Pimephales promelas</i>	51.5	2.6	0.2598	10.01	Brooke et al. 1986
Copepod, <i>Eurytemora affinis</i>	-	2.2	<0.088	>25.00	Hall et al. 1987;1988a
Copepod, <i>Eurytemora affinis</i>	-	2.2	0.145	15.17	Hall et al. 1987;1988a
Mysid, <i>Acanthomysis sculpta</i>	-	0.61 ^a	0.1308	4.664	Davidson et al. 1986a,1986b
Snail, <i>Nucella lapillus</i>	34-35 ^b	72.7	0.0143	5,084	Harding et al. 1996

^a Reported by Valkirs et al. (1985).

^b Salinity (g/kg).

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

Rank^a	Genus Mean Acute Value ($\mu\text{g/L}$)	Species	Species Mean Acute Value ($\mu\text{g/L}$)^b	Species Mean Acute-Chronic Ratio^c
<u>FRESHWATER SPECIES</u>				
12	24,600	Freshwater clam, <i>Elliptio camplanatus</i>	24,600	-
11	12.73	Lake trout, <i>Salvelinus namaycush</i>	12.73	-
10	10.2	Mosquito, <i>Culex</i> sp.	10.2	-
9	8.3	Bluegill, <i>Lepomis macrochirus</i>	8.3	-
8	5.5	Channel catfish, <i>Ictalurus punctatus</i>	5.5	-
7	5.4	Annelid, <i>Lumbriculus variegatus</i>	5.4	-
6	4.571	Rainbow trout, <i>Oncorhynchus mykiss</i>	4.571	-
5	4.3	Cladoceran, <i>Daphnia magna</i>	4.3	36.60
4	3.7	Amphipod, <i>Gammarus pseudolimnaeus</i>	3.7	-
3	2.6	Fathead minnow, <i>Pimephales promelas</i>	2.6	10.01
2	1.80	Hydra, <i>Chlorohydra viridissima</i>	1.80	-
1	1.170	Hydra, <i>Hydra littoralis</i>	1.201	-
		Hydra, <i>Hydra oligactis</i>	1.14	-

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

<u>Rank^a</u>	<u>Genus Mean Acute Value (µg/L)</u>	<u>Species</u>	<u>Species Mean Acute Value (µg/L)^b</u>	<u>Species Mean Acute-Chronic Ratio^c</u>
<u>SALTWATER SPECIES</u>				
30	204.4	European flat oyster, <i>Ostrea edulis</i>	204.4	-
29	108	Amphipod, <i>Rhepoxynius abronius</i>	108	-
28	83.28	Shore crab, <i>Hemigrapsus nudus</i>	83.28	-
27	72.7	Atlantic dogwhelk, <i>Nucella lapillus</i>	72.7	-
26	34.90	Mud crab, <i>Rhithropanopeus harrisi</i>	34.90	-
25	25	Polychaete, <i>Armandia brevis</i>	25	-
24	21.34	Mummichog, <i>Fundulus heteroclitus</i>	21.34	-
23	>14.60	Amphipod, <i>Orchestia traskiana</i>	>14.60	-
22	10.1	Starry flounder, <i>Platichthys stellatus</i>	10.1	-
21	<10	Amphioxus <i>Branchiostoma caribaeum</i>	<10	-
20	9.732	Shore crab, <i>Carcinus maenas</i>	9.732	-
19	9.487	Amphipod, <i>Eohaustorius estuarius</i>	10.0	-
		Amphipod, <i>Eohaustorius washingtonianus</i>	9	-
18	9.037	Sheepshead minnow, <i>Cyprinodon variegatus</i>	9.037	-
17	9.022	Grass shrimp, <i>Palaemonetes pugio</i>	20	-
		Grass shrimp, <i>Palaemonetes</i> sp.	4.07	-

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

<u>Rank^a</u>	<u>Genus Mean Acute Value (µg/L)</u>	<u>Species</u>	<u>Species Mean Acute Value (µg/L)^b</u>	<u>Species Mean Acute-Chronic Ratio^c</u>
<u>SALTWATER SPECIES</u>				
16	6.812	Polychaete, <i>Neanthes arenacedentata</i>	6.812	-
15	5.3	Amphipod, <i>Gammarus</i> sp.	5.3	-
14	5.167	Inland silverside, <i>Menidia beryllina</i>	3.0	-
		Atlantic silverside, <i>Menidia menidia</i>	8.9	-
13	4.944	Atlantic manhaden, <i>Brevoortia tyrannus</i>	4.944	-
12	- 4.74	Lugworm, <i>Arenicola cristata</i>	- 4.74	-
11	3.183	Mysid, <i>Metamysidopsis elongata</i>	3.183	-
10	2.483	Pacific oyster, <i>Crassostrea gigas</i>	1.557	-
		Eastern oyster, <i>Crassostrea virginica</i>	3.96	-
9	2.238	Blue mussel, <i>Mytilus edulis</i>	2.238	-
8	1.975	Copepod, <i>Eurytemora affinis</i>	1.975	15.17
7	1.911	Copepod, <i>Nitocra spinipes</i>	1.911	-
6	1.745	American lobster, <i>Homarus americanus</i>	1.745	-
5	1.692	Mysid, <i>Amenicamysis bahia</i>	1.692	-
4	1.65	Hard clam, <i>Mercenaria mercenaria</i>	1.65	-
3	1.460	Chinook salmon, <i>Oncorhynchus tshawytscha</i>	1.460	-

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

<u>Rank^a</u>	<u>Genus Mean Acute Value (µg/L)</u>	<u>Species</u>	<u>Species Mean Acute Value (µg/L)^b</u>	<u>Species Mean Acute-Chronic Ratio^c</u>
<u>SALTWATER SPECIES</u>				
2	1.1	Copepod, <i>Acartia tonsa</i>	1.1	-
1	0.61	Mysid, <i>Acanthomysis sculpta</i>	0.61	4.664

^a Ranked from most resistant to most sensitive based on Genus Mean Acute Value. Inclusion of "greater than" value does not necessarily imply a true ranking, but does allow use of all genera for which data are available so that the Final Acute Value is not unnecessarily lowered.

^b From Table 1.

^c From Table 2.

Fresh Water

Final Acute Value = 0.9177 µg/L

Criterion Maximum Concentration = (0.9177 µg/L) ÷ 2 = 0.4589 µg/L

Final Acute-Chronic Ratio = 12.69 (see text)

Final Chronic Value = (0.9177 µg/L) ÷ 12.69 = 0.0723 µg/L

Salt Water

Final Acute Value = 0.8350 µg/L

Criterion Maximum Concentration = (0.8350 µg/L) ÷ 2 = 0.4175 µg/L

Final Acute-Chronic Ratio = 12.69 (see text)

Final Chronic Value = (0.8350 µg/L) ÷ 12.69 = 0.0658 µg/L

Final Chronic Value = 0.0074 µg/L (lowered to protect growth of commercially important molluscs, survival of the ecologically important copepod *Acartia tonsa*, and survival of the ecologically important gastropod *Nucella lapillus*; see text)

Table 4. Toxicity of Tributyltin to Aquatic Plants

<u>Species</u>	<u>Chemical^a</u>	<u>Hardness (mg/L as CaCO₃)</u>	<u>Duration (days)</u>	<u>Effect</u>	<u>Concentration (µg/L)^b</u>	<u>Reference</u>
<u>FRESHWATER SPECIES</u>						
Alga, <i>Bumilleriopsis filiformis</i>	TBTCI	-	14	No growth	111.4	Blanck 1986; Blanck et al. 1984
Alga, <i>Klebsormidium marinum</i>	TBTCI	-	14	No growth	222.8	Blanck 1986; Blanck et al. 1984
Alga, <i>Monodus subterraneus</i>	TBTCI	-	14	No growth	1,782	Blanck 1986; Blanck et al. 1984
Alga, <i>Raphidonema longiseta</i>	TBTCI	-	14	No growth	56.1	Blanck 1986; Blanck et al. 1984
Alga, <i>Tribonema aequale</i>	TBTCI	-	14	No growth	111.4	Blanck 1986; Blanck et al. 1984
Blue-green alga, <i>Oscillatoria</i> sp.	TBTCI	-	14	No growth	222.8	Blanck 1986; Blanck et al. 1984
Blue-green alga, <i>Synechococcus leopoliensis</i>	TBTCI	-	14	No growth	111.4	Blanck 1986; Blanck et al. 1984
Green alga, <i>Chlamydomonas dysosmas</i>	TBTCI	-	14	No growth	111.4	Blanck 1986; Blanck et al. 1984
Green alga, <i>Chlorella emersonii</i>	TBTCI	-	14	No growth	445.5	Blanck 1986; Blanck et al. 1984
Green alga, <i>Kirchneriella contorta</i>	TBTCI	-	14	No growth	111.4	Blanck 1986; Blanck et al. 1984
Green alga, <i>Monoraphidium pusillum</i>	TBTCI	-	14	No growth	111.4	Blanck 1986; Blanck et al. 1984
Green alga, <i>Scenedesmus obtusiusculus</i>	TBTCI	-	14	No growth	445.5	Blanck 1986; Blanck et al. 1984

Table 4. Toxicity of Tributyltin to Aquatic Plants (continued)

<u>Species</u>	<u>Chemical^a</u>	<u>Hardness (mg/L as CaCO₃)</u>	<u>Duration (days)</u>	<u>Effect</u>	<u>Concentration (µg/L)^b</u>	<u>Reference</u>
<u>FRESHWATER SPECIES</u>						
Green alga, <i>Scenedesmus quadricauda</i>	TBTCI	-	12	Reduced growth (87.6%)	1	Fargasova and Kizlink 1996
Green alga, <i>Scenedesmus quadricauda</i>	TBTO	-	7	EC50 chlorophyll production	5.0	Fargasova 1996
Green alga, <i>Scenedesmus quadricauda</i>	TBTO	0.67	12	Reduced growth 87.6% 95.9% 100%	1 10 100	Fargasova and Kizlink 1996
Green alga, <i>Scenedesmus obliquus</i>	TBT	72.7	4	EC50 (reduced growth)	3.4	Huang et al. 1993
Green alga, <i>Selenastrum capricornutum</i>	TBTCI	-	14	No growth	111.4	Blanck 1986; Blanck et al. 1984
Green alga, <i>Selenastrum capricornutum</i>	TBTCI	-	4	EC50	12.4	Miana et al. 1993
<u>SALTWATER SPECIES</u>						
Diatom, <i>Skeletonema costatum</i>	TBTO	-	5	Algistatic Algicidal	0.9732-17.52 >17.52	Thain 1983
Diatom, <i>Skeletonema costatum</i>	TBTO (BioMet Red)	30°	14	EC50 (dry cell weight)	>0.1216; <0.2433	EG&G Bionomics 1981c
Diatom, <i>Skeletonema costatum</i>	TBTO	30°	14	EC50 (dry cell weight)	0.06228	EG&G Bionomics 1981c
Diatom, <i>Nitzschia</i> sp.	TBTO	-	8	EC50 (reduced growth)	1.19	Delupis et al. 1987

Table 4. Toxicity of Tributyltin to Aquatic Plants (continued)

<u>Species</u>	<u>Chemical</u> ^a	<u>Hardness</u> (mg/L as <u>CaCO₃</u>)	<u>Duration</u> (<u>days</u>)	<u>Effect</u>	<u>Concentration</u> (<u>µg/L</u>) ^b	<u>Reference</u>
<u>SALTWATER SPECIES</u>						
Flagellate alga, <i>Dunaliella</i> <i>tertiolecta</i>	TBTO	-	8	EC50 (reduced growth)	4.53	Delupis et al. 1987
Mixed algae, <i>Dunaliella</i> <i>salina</i> and <i>D. viridis</i>	TBT	-	4	EC50 (reduced growth)	0.68	Huang et al. 1993

^a TBTCI = tributyltin chloride; TBTO = tributyltin oxide. Percent purity is given in parentheses when available.

^b Concentration of the tributyltin cation, not the chemical. If the concentrations were not measured and the published results were not reported to be adjusted for purity, the published results were multiplied by the purity if it was reported to be less than 95%.

^c Salinity (g/kg).

Table 5. Bioaccumulation of Tributyltin by Aquatic Organisms

<u>Species</u>	<u>Chemical^a</u>	<u>Hardness (mg/L as (CaCO₃))</u>	<u>Conc. in Water (µg/L)^b</u>	<u>Duration (days)</u>	<u>Tissue</u>	<u>BCF or BAF^c</u>	<u>Reference</u>
<u>FRESHWATER SPECIES</u>							
Zebra mussel (1.76 ±0.094 cm), <i>Dreissena polymorpha</i>	TBT	-	0.0703	105	Soft parts	17,483 ^d	Becker-van Slooten and Tarradellas 1994
Rainbow trout (13.8 g), <i>Oncorhynchus mykiss</i>	TBTO (97%)	135	0.513	64	Whole body	406	Martin et al. 1989
Rainbow trout (32.7 g), <i>Oncorhynchus mykiss</i>	TBTO (97%)	135	1.026	15	Liver Gall bladder/bile Kidney Carcass Peritoneal fat Gill Blood Gut Muscle	1,179 331 2,242 1,345 5,419 1,014 653 487 312	Martin et al. 1989
Carp (9.5-11.5 cm; 20.0- 27.5 g); <i>Cyprinus carpio</i>	TBTO	-	2.1	14	Muscle	501.2	Tsuda et al. 1988a
Carp (8.5-9.5 cm; 16.5-22.1 g); <i>Cyprinus carpio</i>	TBTO	34.5-39.0	1.8 (pH=6.0) 1.6 (pH=6.8) 1.7 (pH=7.8)	14	Whole body	- 1190 - 1523 - 2250	Tsuda et al. 1990a
Goldfish (3.5-4.0 cm; 1.6-2.9 g); <i>Carassius auratus</i>	TBTCl	36	0.13	28	Whole body	1,976	Tsuda et al. 1991b
Guppy (8; 2.4-2.7 cm; 0.41-0.55 g); <i>Poecilia reticulatus</i>	TBTC (95%)	-	0.28	14	Whole body	240	Tsuda et al. 1990b
Guppy (2.4-2.7 cm; 0.41-0.55 g); <i>Poecilia reticulatus</i>	TBTO (95%)	-	0.54	14	Whole body	460	Tsuda et al. 1990b

Table 5. Bioaccumulation of Tributyltin by Aquatic Organisms (continued)

<u>Species</u>	<u>Chemical^a</u>	<u>Salinity (g/kg)</u>	<u>Conc. in Water (µg/L)^b</u>	<u>Duration (days)</u>	<u>Tissue</u>	<u>BCF or BAF^c</u>	<u>Reference</u>
<u>SALTWATER SPECIES</u>							
Snail (adults), <i>Littorina littorina</i>	TBTCI	-	0.488	182	Soft parts	1,420	Bauer et al. 1997
			0.976	182	Soft parts	1,020	
Atlantic dogwhelk (female), <i>Nucella lapillus</i>	TBT	-	0.0038 to 0.268	249 to 408	Soft parts	11,000 to 38,000	Bryan et al. 1987a
Atlantic dogwhelk (female), <i>Nucella lapillus</i>	Field	-	0.070	529 to 634	Soft parts	17,000	Bryan et al. 1987a
Atlantic dog whelk (18-22 mm), <i>Nucella lapillus</i>	TBTCI	35	0.0205	49	Soft parts	30,000	Bryan et al. 1989b
Atlantic dogwhelk (1 year-old), <i>Nucella lapillus</i>	TBTO	34-35	0.0027	365	Soft parts	18,727	Bailey et al. 1991
			0.0077	365	Soft parts	21,964	
			0.0334	365	Soft parts	16,756	
			0.1246	365	Soft parts	7,625	
Atlantic dogwhelk (1 year-old), <i>Nucella lapillus</i>	TBTO	34-35	0.0026	365	Soft parts	<7782	Harding et al. 1996
			0.0074	365	Soft parts	10,121	
			0.0278	365	Soft parts	8,088	
			0.1077	365	Soft parts	6,172	
Blue mussel (spat), <i>Mytilus edulis</i>	- ^e	28.5-34.2	0.24	45	Soft parts	6,833 ^f	Thain and Waldock 1985; Thain 1986
Blue mussel (adult), <i>Mytilus edulis</i>	Field	-	<0.1	60	-	11,000	Salazar and Salazar 1990a
Blue mussel (juvenile), <i>Mytilus edulis</i>	Field	-	<0.1	60	-	25,000	Salazar and Salazar 1990a
Blue mussel, <i>Mytilus edulis</i>	- ^e	-	0.452 0.204 0.204 0.079	56	Soft parts	23,000 27,000 10,400 37,500	Salazar et al. 1987
Blue mussel (juvenile), <i>Mytilus edulis</i>	Field	-	<0.105	84	Soft parts	5,000- 60,000	Salazar and Salazar, 1991

Table 5. Bioaccumulation of Tributyltin by Aquatic Organisms (continued)

<u>Species</u>	<u>Chemical^a</u>	<u>Salinity (g/kg)</u>	<u>Conc. in Water (µg/L)^b</u>	<u>Duration (days)</u>	<u>Tissue</u>	<u>BCF or BAF^c</u>	<u>Reference</u>
<u>SALTWATER SPECIES</u>							
Blue mussel (3.0 - 3.5 cm), <i>Mytilus edulis</i>	TBTCl	25.1-26.3	0.020	60	Muscle and mantle	7,700	Guolan and Yong 1995
					Muscle and mantle	11,000	
Eastern Oyster (6-9 cm length), <i>Crassostrea virginica</i>	-	18±1	0.283	28	Soft parts	15,460	Roberts et al. 1996
Pacific oyster, <i>Crassostrea gigas</i>	TBTO	28-31.5	1.216	21	Soft parts	1,874 ^f	Waldock et al. 1983
Pacific oyster, <i>Crassostrea gigas</i>	TBTO	28-31.5	0.1460	21	Soft parts	6,047 ^f	Waldock et al. 1983
Pacific oyster, <i>Crassostrea gigas</i>	- ^e	28.5-34.2	0.24	45	Soft parts	7,292 ^f	Thain and Waldock 1985; Thain 1986
Pacific oyster, <i>Crassostrea gigas</i>	TBTO	29-32	1.557	56	Soft parts	2,300	Waldock and Thain 1983
Pacific oyster, <i>Crassostrea gigas</i>	TBTO	29-32	0.1460	56	Soft parts	11,400	Waldock and Thain 1983
Pacific oyster (spat), <i>Crassostrea gigas</i>	TBTO	-	0.29 0.92 2.83	30	Soft parts	2275 1369 621	Osada et al. 1993
European flat oyster, <i>Ostrea edulis</i>	TBTO	28-31.5	1.216	21	Soft parts	960 ^f	Waldock et al. 1983
European flat oyster, <i>Ostrea edulis</i>	TBTO	28-34.2	0.24	75	Soft parts	875 ^f	Waldock et al. 1983
European flat oyster, <i>Ostrea edulis</i>	TBTO	28-34.2	2.62	75	Soft parts	397 ^f	Thain 1986
European flat oyster, <i>Ostrea edulis</i>	- ^e	28.5-34.2	0.24	45	Soft parts	1,167 ^f	Thain and Waldock 1985; Thain 1986

SALTWATER SPECIES

Table 5. Bioaccumulation of Tributyltin by Aquatic Organisms (continued)

<u>Species</u>	<u>Chemical</u> ^a	<u>Salinity</u> <u>(g/kg)</u>	<u>Conc.</u> <u>in Water</u> <u>(µg/L)</u> ^b	<u>Duration</u> <u>(days)</u>	<u>Tissue</u>	<u>BCF</u> <u>or</u> <u>BAF</u> ^c	<u>Reference</u>
European flat oyster, <i>Ostrea edulis</i>	- ^e	28.5-34.2	2.62	45	Soft parts	192.3 ^f	Thain and Waldoock 1985; Thain 1986

^a TBTO = tributyltin oxide; Field = field study. Percent purity is given in parentheses when available.

^b Measured concentration of the tributyltin cation.

^c Bioconcentration factors (BCFs) and bioaccumulation factors (BAFs) are based on measured concentrations of TBT in water and tissue.

^d BCF normalized to 1% lipid concentration and converted to wet weight estimate based upon 85% moisture.

^e Test organisms were exposed to leachate from panels coated with antifouling paint containing tributyltin.

^f BCFs were calculated based on the increase above the concentration of TBT in control organisms.

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

<u>Species</u>	<u>Chemical^a</u>	<u>Hardness (mg/L as CaCO₃)</u>	<u>Duration</u>	<u>Effect</u>	<u>Concentration (µg/L)^b</u>	<u>Reference</u>
<u>FRESHWATER SPECIES</u>						
Microcosm natural assemblage	TBTO	-	55 days	<i>Daphnia magna</i> disappeared; Ostracoda increased; algae increased immediately then gradually disappeared	80	Delupis and Miniero 1989
Microcosm natural assemblage	TBTO	-	24 days	Metabolism reduced (2.5 days)	4.7	Miniero and Delupis 1991
				Metabolism returned to normal (14.1 days)		
				Metabolism reduced (1 day)	14.9	
				Metabolism returned to normal (16 days)		
Alga, Natural assemblage	-	-	4 hr	EC50 (production)	5	Wong et al. 1982
Blue-green alga, <i>Anabaena flos-aquae</i>	-	-	4 hr	EC50 (production)	13	Wong et al. 1982
Green alga, <i>Ankistrodesmus falcatus</i>	-	-	4 hr	EC50 (production)	20	Wong et al. 1982
				EC50 (reproduction)	5	
Green alga, <i>Ankistrodesmus falcatus</i>	TBTO (97%)	-	7 days	BCF = 300	5.2	Maguire et al. 1984
			14 days	BCF = 253	4.7	
			21 days	BCF = 448	2.1	
			28 days	BCF = 467	1.5	
Green alga, <i>Scenedesmus quadricauda</i>	-	-	4 hr	EC50 (production)	16	Wong et al. 1982
Hydra, <i>Hydra</i> sp.	TBTO (96%)	51.0	96 hr	EC50 (clubbed tentacles)	0.5	Brooke et al. 1986
Rotifer, <i>Brachionus calyciflorus</i>	TBTCl	-	24 hr	EC50 (hatching)	72	Crisinel et al. 1994
Asiatic clam (larva), <i>Corbicula fluminea</i>	TBTO	-	24 hr	EC50	1,990	Foster 1981

Table 6. Other Data on Effects of Tributyltin on Aquatic Organisms (continued)

<u>Species</u>	<u>Chemical^a</u>	<u>Hardness (mg/L as CaCO₃)</u>	<u>Duration</u>	<u>Effect</u>	<u>Concentration (µg/L)^b</u>	<u>Reference</u>
<u>FRESHWATER SPECIES</u>						
Cladoceran, <i>Daphnia magna</i>	TBTO	-	24 hr	LC50	3	Polster and Halacha 1972
Cladoceran (<24 hr), <i>Daphnia magna</i>	TBTC	200	24 hr	EC50 (mobility)	11.6	Vighi and Calamari 1985
Cladoceran (<24 hr), <i>Daphnia magna</i>	TBTO	200	24 hr	EC50 (mobility)	13.6	Vighi and Calamari 1985
Cladoceran (adult), <i>Daphnia magna</i>	TBTCI	-	8 days	Altered phototaxis	0.45	Meador 1986
Cladoceran (14-d-old), <i>Daphnia magna</i>	TBTCI	150	7 days	Altered behavior Reproductive failure Digestive storage cells reduced	1 1 5	Bodar et al. 1990
Cladoceran (<24- h old), <i>Daphnia magna</i>	TBTCI	312.8	48 hr	EC50 (mobility)	9.8	Miana et al. 1993
Fairy shrimp (cysts), <i>Streptocephalus texanus</i>	TBTCI	250	24 hr	EC50 (hatching)	15	Crisinel et al. 1994
Rainbow trout (yearling), <i>Oncorhynchus mykiss</i>	TBTO	-	24 hr 48 hr	LC50	25.2 18.9	Alabaster 1969
Rainbow trout, <i>Oncorhynchus mykiss</i>	TBTO	-	24 hr	EC50 (rheotaxis)	30.8	Chliamovitch and Kuhn 1977
Rainbow trout (embryo, larva), <i>Oncorhynchus mykiss</i>	TBTCI	94-102	110 days	20% reduction in growth 23% reduction in growth; 6.6% mortality 100% mortality	0.18 0.89 4.46	Seinen et al. 1981
Rainbow trout (fry), <i>Oncorhynchus mykiss</i>	TBTCI	96-105	110 days	NOEC (mortality and growth) LOEC (mortality and growth)	0.040 0.200	de Vries et al. 1991

Table 6. Other Data on Effects of Tributyltin on Aquatic Organisms (continued)

<u>Species</u>	<u>Chemical^a</u>	<u>Hardness (mg/L as CaCO₃)</u>	<u>Duration</u>	<u>Effect</u>	<u>Concentration (µg/L)^b</u>	<u>Reference</u>
<u>FRESHWATER SPECIES</u>						
Rainbow trout, <i>Oncorhynchus mykiss</i>	TBTO	-	28 days	BCF = 3,833 (whole body) BCF = 2,850 (whole body) BCF = 2,700 (whole body) BCF = 1,850 (whole body) Cell necrosis within gill lamellae	0.6 1.0 2.0 4.0 4.0	Schwaiger et al. 1992
Rainbow trout, <i>Oncorhynchus mykiss</i>	TBTO	-	28 days	BCF = 3,833 (whole body) BCF = 2,850 (whole body) BCF = 2,700 (whole body) BCF = 1,850 (whole body) Cell necrosis within gill lamellae	0.6 1.0 2.0 4.0 4.0	Schwaiger et al. 1992
Rainbow trout (3 wk), <i>Oncorhynchus mykiss</i>	TBTO (98%)	400	21 days	Reduced growth Reduced avoidance BCF = 540 (no head; no plateau) BCF = 990 (no head; no plateau)	0.5 2	Triebkorn et al. 1994
Goldfish (2.8-3.5 cm; 0.9-1.7 g), <i>Carassius auratus</i>	TBTO (reagent grade)	-	14 days	BCF = 1230 (no plateau)	2.0	Tsuda et al. 1988b
Carp (10.0-11.0 cm; 22.9-30.4 g), <i>Cyprinus carpio</i>	TBTO	-	7 days	BCF in muscle = 295 Half-life = 1.67 days	1.80	Tsuda et al. 1987
Guppy (3-4 wk), <i>Poecilia reticulata</i>	TBTO	209	3 mo	Thymus atrophy Hyperplasia of kidney hemopoietic tissue Marked liver vacuolation Hyperplasia of corneal epithelium	0.32 1.0 1.0 10.0	Wester and Canton 1987
Guppy (4 wk), <i>Poecilia reticulata</i>	TBTO	-	1 mo 3 mo	NOEC NOEC	1.0 0.32	Wester and Canton 1991
Frog (embryo, larva), <i>Rana temporaria</i>	TBTO TBTF TBTO TBTF	- - - -	5 days	LC40 LC50 Loss of body water Loss of body water	28.4 28.2 28.4 28.2	Laughlin and Linden 1982

Table 6. Other Data on Effects of Tributyltin on Aquatic Organisms (continued)

<u>Species</u>	<u>Chemical^a</u>	<u>Salinity (g/kg)</u>	<u>Duration</u>	<u>Effect</u>	<u>Concentration (µg/L)^b</u>	<u>Reference</u>
<u>SALTWATER SPECIES</u>						
Natural microbial populations	TBTCI	2 and 17	1 hr (incubated 10 days)	Significant decrease in metabolism of nutrient substrates	4.454	Jonas et al. 1984
Natural microbial populations	TBTCI	2 and 17	1 hr (incubated 10 days)	50% mortality	89.07	Jonas et al. 1984
Fouling communities	^c	33-36	2 months	Reduced species and diversity; no effect at 0.04 µg/L	0.1	Henderson 1986
Fouling communities	^c	-	126 days	No effect	0.204	Salazar et al. 1987
Microcosm (seagrass bed)	TBT (>95%)	21.5-28.9	6 wks	Fate of TBT Sediments 81-88% Plants 9-17% Animals 2-4%	0.2-20	Levine et al. 1990
Microcosm (seagrass bed)	TBTCI	-	6 wks	Reduced plant material loss; loss of amphipod <i>Cymadusa compta</i>	22.21	Kelly et al. 1990a
Periphyton communities	TBTCI	-	15 min	EC50 (reduced photosynthesis)	28.7	Blanck and Dahl 1996
Periphyton communities	TBTO	-	15 min	EC50 (reduced photosynthesis)	27.9	Blanck and Dahl 1996
Green alga, <i>Dunaliella tertiolecta</i>	TBTO	34-40	18 days	Population growth	1.0	Beaumont and Newman 1986
Green alga, <i>Dunaliella</i> sp.	TBTO	-	72 hr	Approx. EC50 (growth)	1.460	Salazar 1985
Green alga, <i>Dunaliella</i> sp.	TBTO	-	72 hr	100% mortality	2.920	Salazar 1985
Green alga, <i>Dunaliella tertiolecta</i>	TBTO	-	8 days	EC50	4.53	Delupis et al. 1987
Diatom, <i>Phaeodactylum tricornutum</i>	TBTO	-	72 hr	No effect on growth	1.460-5.839	Salazar 1985
Diatom, <i>Nitzschia</i> sp.	TBTO	-	8 days	EC50	1.19	Delupis et al. 1987
Diatom, <i>Nitzschia</i> sp.	TBTO	-	8 days	EC50	1.19	Delupis et al. 1987

Table 6. Other Data on Effects of Tributyltin on Aquatic Organisms (continued)

<u>Species</u>	<u>Chemical^a</u>	<u>Salinity (g/kg)</u>	<u>Duration</u>	<u>Effect</u>	<u>Concentration ($\mu\text{g/L}$)^b</u>	<u>Reference</u>
<u>SALTWATER SPECIES</u>						
Diatom, <i>Nitzschia closterium</i>	TBTCI	-	7 days	EC50 (growth)	1.16	Nakagawa and Saeki 1992
Diatom, <i>Skeletonema costatum</i>	TBTA	30	72 hr	EC50 (population growth)	0.3097	Walsh et al. 1985;1987
Diatom, <i>Skeletonema costatum</i>	TBTA	30	72 hr	LC50	12.65	Walsh et al. 1985;1987
Diatom, <i>Skeletonema costatum</i>	TBTO	34-40	12-18 days	Population growth	1.0	Beaumont and Newman 1986
Diatom, <i>Skeletonema costatum</i>	TBTO	30	72 hr	EC50 (population growth)	0.3212	Walsh et al. 1985;1987
Diatom, <i>Skeletonema costatum</i>	TBTO	30	72 hr	LC50	13.82	Walsh et al. 1985
Diatom, <i>Skeletonema costatum</i>	TBTCI	30	72 hr	EC50 (population growth)	0.3207	Walsh et al. 1985;1987
Diatom, <i>Skeletonema costatum</i>	TBTCI	30	72 hr	LC50	10.24	Walsh et al. 1985;1987
Diatom, <i>Skeletonema costatum</i>	TBTF	30	72 hr	EC50 (population growth)	>0.2346, >0.4693	Walsh et al. 1985;1987
Diatom, <i>Skeletonema costatum</i>	TBTF	30	72 hr	LC50	11.17	Walsh et al. 1985
Diatom, <i>Skeletonema costatum</i>	TBTCI	30.5	96 hr	NOEC	1	Reader and Pelletier 1992
Diatom, <i>Skeletonema costatum</i>	TBTCI	-	7 days	EC50 (growth)	3.48	Nakagawa and Saeki 1992
Diatom, <i>Chaetoceros debilis</i>	TBTCI	-	7 days	EC50 (growth)	1.16	Nakagawa and Saeki 1992

Table 6. Other Data on Effects of Tributyltin on Aquatic Organisms (continued)

<u>Species</u>	<u>Chemical^a</u>	<u>Salinity (g/kg)</u>	<u>Duration</u>	<u>Effect</u>	<u>Concentration ($\mu\text{g/L}$)^b</u>	<u>Reference</u>
<u>SALTWATER SPECIES</u>						
Diatom, <i>Chattonella antiqua</i>	TBTCl	-	7 days	EC50 (growth)	2.05	Nakagawa and Saeki 1992
Diatom, <i>Tetraselmis tetrathele</i>	TBTCl	-	7 days	EC50 (growth)	6.06	Nakagawa and Saeki 1992
Diatom, <i>Minutocellus polymorphus</i>	TBTO	-	48 hr	EC50	- 340	Walsh et al. 1988
Diatom, <i>Minutocellus polymorphus</i>	TCTCl	-	48 hr	EC50	- 330	Walsh et al. 1988
Diatom, <i>Thalassiosira pseudonana</i>	TBTA	30	72 hr	EC50 (population growth)	1.101	Walsh et al. 1985
Diatom, <i>Thalassiosira pseudonana</i>	TBTO	30	72 hr	EC50 (population growth)	1.002	Walsh et al. 1985;1987
Alga, <i>Pavlova lutheri</i>	TBTO	34-40	12-26 days	Population growth	1.0	Beaumont and Newman 1986
Alga, <i>Pavlova lutheri</i>	TBTO	-	16 days	NOEC LOEC	5.36 21.46	Saint-Louis et al. 1994
Dinoflagellate, <i>Gymnodinium splendens</i>	TBTO	-	72 hr	100% mortality	1.460	Salazar 1985
Macroalgae, <i>Fucus vesiculosus</i>	TBT	6	7 days	Photosynthesis and nutrient uptake reduced	0.6	Lindblad et al. 1989
Giant kelp (zoospores), <i>Macrocystis pyrifera</i>	TBT	32-33	48 hr	EC50 (germination) EC50 (growth)	11.256 13.629	Brix et al. 1994a
Polychaete worm (juvenile), <i>Neanthes arenaceodentata</i>	TBTCl (96%)	30	10 wks	NOEC (survival) LOEC (survival)	0.100 0.500	Moore et al. 1991
Polychaete worm (adult), <i>Armandia brevis</i>	TBTCl (96%)	28.5	10 days	BCF = 5,100 (no plateau)	233	Meador 1997

Table 6. Other Data on Effects of Tributyltin on Aquatic Organisms (continued)

<u>Species</u>	<u>Chemical^a</u>	<u>Salinity (g/kg)</u>	<u>Duration</u>	<u>Effect</u>	<u>Concentration (µg/L)^b</u>	<u>Reference</u>
<u>SALTWATER SPECIES</u>						
Rotifer (neonates), <i>Brachionus plicatilis</i>	TBT	15	30 min	Induction of the stress protein gene SP58	20-30	Cochrane et al. 1991
Hydroid, <i>Campanularia flexuosa</i>	TBTF	35	11 days	Colony growth stimulation; no growth	0.01 1.0	Stebbing 1981
Pale sea anemone (1-2 cm oral disc), <i>Aiptasia pallida</i>	TBT	-	28 days	Reduced (90.4%) symbiotic zooxanthellae populations; increased bacterial aggregates; fewer undischarged nematocysts	0.05	Mercier et al. 1997
Sand dollar (sperm), <i>Dendraster excentricus</i>	TBT	32-33	80 min	EC50 (mortality)	0.465	Brix et al. 1994b
Starfish (79 g), <i>Leptasterias polaris</i>	TBTCI	25.9	48 hr	BCF = 41,374 (whole body)	0.072	Rouleau et al. 1995
Dogwhinkle (adult), <i>Nucella lapillus</i>	- ^c	-	120 days	41% Imposex (superimposition of male anatomical characteristics on females)	0.05	Bryan et al. 1986
Dogwhinkle (adult), <i>Nucella lapillus</i>	TBTCI	35	6 months	Imposex induced	\$0.012	Stroben et al. 1992b
Dogwhinkle (subadult), <i>Nucella lapillus</i>	TBTCI	35	22	BCF = - 20,000	0.019	Bryan et al. 1993
Mussel (juvenile), <i>Mytilus</i> sp.	Field	-	12 weeks	NOEC tissue concentration growth = 0.5 µg/g LOEC tissue concentration growth = 1.5 µg/g NOEC (growth) LOEC (growth) BAF = 5,000-100,000	- - 0.025 0.100 <0.105	Salazar and Salazar 1990b, 1996
Blue mussel (larva), <i>Mytilus edulis</i>	TBTO	-	24 hr	No effect on sister chromatid exchange	1.0	Dixon and Prosser 1986
Blue mussel (larva), <i>Mytilus edulis</i>	TBTO	-	4 days	Reduced survival	\$0.1	Dixon and Prosser 1986

Table 6. Other Data on Effects of Tributyltin on Aquatic Organisms (continued)

<u>Species</u>	<u>Chemical^a</u>	<u>Salinity (g/kg)</u>	<u>Duration</u>	<u>Effect</u>	<u>Concentration (µg/L)^b</u>	<u>Reference</u>
<u>SALTWATER SPECIES</u>						
Blue mussel (spat), <i>Mytilus edulis</i>	- ^c	28.5-34.2	45 days	100% mortality	2.6	Thain and Waldock 1985; Thain 1986
Blue mussel (larva), <i>Mytilus edulis</i>	TBTO	33	15 days	51% mortality; reduced growth	0.0973	Beaumont and Budd 1984
Blue mussel (larva), <i>Mytilus edulis</i>	- ^c	-	45 days	Reduced growth	0.24	Thain and Waldock 1986
Blue mussel (juvenile), <i>Mytilus edulis</i>	TBTO	33.7	7 days	Significant reduction in growth	0.3893	Stromgren and Bongard 1987
Blue mussel (juvenile), <i>Mytilus edulis</i>	TBT (field)	-	1-2 wk	Reduced growth; at <0.2 µg/L environmental factors most important	0.2	Salazar and Salazar 1990b
Blue mussel (juvenile), <i>Mytilus edulis</i>	TBT (field)	-	12 wks	Reduced growth	\$0.2	Salazar and Salazar 1988
Blue mussel (juvenile), <i>Mytilus edulis</i>	TBT (field)	-	12 wks	Reduced growth at tissue conc. of 2.0 µg/g	-	Salazar and Salazar 1988
Blue mussel (juvenile), <i>Mytilus edulis</i>	- ^c	-	56 days	Reduced condition	0.157	Salazar et al. 1987
Blue mussel (juvenile), <i>Mytilus edulis</i>	- ^c	-	196 days	Reduced growth (no effect at day 56 of 0.2 µg/L)	0.070	Salazar and Salazar 1987
Blue mussel (juvenile), <i>Mytilus edulis</i>	- ^c	-	56 days	No effect on growth	0.160	Salazar and Salazar 1987
Blue mussel (2.5 to 4.1 cm), <i>Mytilus edulis</i>	- ^c	-	66 days	LC50	0.97	Valkirs et al. 1985, 1987
Blue mussel (2.5 to 4.1 cm), <i>Mytilus edulis</i>	- ^c	-	66 days	Significant decrease in shell growth	0.31	Valkirs et al. 1985
Blue mussel (juveniles and adults), <i>Mytilus</i> sp.	TBT (field)	-	84 days	BCF	3,000-100,000	Salazar and Salazar 1996

Table 6. Other Data on Effects of Tributyltin on Aquatic Organisms (continued)

<u>Species</u>	<u>Chemical^a</u>	<u>Salinity (g/kg)</u>	<u>Duration</u>	<u>Effect</u>	<u>Concentration (µg/L)^b</u>	<u>Reference</u>
<u>SALTWATER SPECIES</u>						
Blue mussel (3.0-3.5 cm), <i>Mytilus edulis</i>	TBT	-	2 days	Reduced ability to survive anoxia	1	Wang et al. 1992
Blue mussel (4 cm), <i>Mytilus edulis</i>	TBTCl (>97%)	-	2.5 days	Increased respiration 0.15 µg/g tissue Reduced food absorption efficiency 10 µg/g	-	Widdows and Page 1993
Blue mussel (8-d-old larvae), <i>Mytilus edulis</i>	TBT	-	33 days	NOEC (growth) LOEC (growth)	0.006 0.050	Lapota et al. 1993
Scallop (adult), <i>Hinnites multirugosus</i>	- ^c	-	110 days	No effect on condition	0.204	Salazar et al. 1987
Pacific oyster (larva), <i>Crassostrea gigas</i>	- ^c	-	30 days	100% mortality	2.0	Alzieu et al. 1980
Pacific oyster (larva), <i>Crassostrea gigas</i>	- ^c	-	113 days	30% mortality and abnormal development	0.2	Alzieu et al. 1980
Pacific oyster (spat), <i>Crassostrea gigas</i>	TBTO	-	48 days	Reduced growth	0.020	Lawler and Aldrich 1987
Pacific oyster (spat), <i>Crassostrea gigas</i>	TBTO	-	14 days	Reduced oxygen consumption and feeding rates	0.050	Lawler and Aldrich 1987
Pacific oyster (spat), <i>Crassostrea gigas</i>	- ^c	28.5-34.2	45 days	40% mortality; reduced growth	0.24	Thain and Waldock 1985
Pacific oyster (spat), <i>Crassostrea gigas</i>	- ^c	28.5-34.2	45 days	90% mortality	2.6	Thain and Waldock 1985
Pacific oyster (spat), <i>Crassostrea gigas</i>	- ^c	-	45 days	Reduced growth	0.24	Thain and Waldock 1986

Table 6. Other Data on Effects of Tributyltin on Aquatic Organisms (continued)

<u>Species</u>	<u>Chemical^a</u>	<u>Salinity (g/kg)</u>	<u>Duration</u>	<u>Effect</u>	<u>Concentration ($\mu\text{g/L}$)^b</u>	<u>Reference</u>
<u>SALTWATER SPECIES</u>						
Pacific oyster (spat), <i>Crassostrea gigas</i>	TBT	-	49 days	Shell thickening	0.020	Thain et al. 1987
Pacific oyster (spat), <i>Crassostrea gigas</i>	TBTO	29-32	56 days	No growth	1.557	Waldock and Thain 1983
Pacific oyster (spat), <i>Crassostrea gigas</i>	TBTO	29-32	56 days	Reduced growth	0.1460	Waldock and Thain 1983
Pacific oyster (adult), <i>Crassostrea gigas</i>	TBT (field)	-	-	Shell thickening	\$0.014	Wolniakowski et al. 1987
Pacific oyster (larva), <i>Crassostrea gigas</i>	TBTF	18-21	21 days	Reduced number of normally developed larvae	0.02346	Springborn Bionomics, Inc. 1984a
Pacific oyster (larva), <i>Crassostrea gigas</i>	TBTF	18-21	15 days	100% mortality	0.04692	Springborn Bionomics, Inc. 1984a
Pacific oyster (embryo), <i>Crassostrea gigas</i>	TBTA	28	24 hr	Abnormal development; 30-40% mortality	4.304	His and Robert 1980
Pacific oyster (embryo), <i>Crassostrea gigas</i>	TBTA	-	24 hr	Abnormal development	0.8604	Robert and His 1981
Pacific oyster (larva), <i>Crassostrea gigas</i>	TBTA	-	24 hr	Abnormal development	\$0.9	Robert and His 1981
Pacific oyster (larva), <i>Crassostrea gigas</i>	TBTA	-	48 hr	100% mortality	2.581	Robert and His 1981

Table 6. Other Data on Effects of Tributyltin on Aquatic Organisms (continued)

<u>Species</u>	<u>Chemical^a</u>	<u>Salinity (g/kg)</u>	<u>Duration</u>	<u>Effect</u>	<u>Concentration ($\mu\text{g/L}$)^b</u>	<u>Reference</u>
<u>SALTWATER SPECIES</u>						
Pacific oyster (150-300 mg), <i>Crassostrea gigas</i>	- ^c	-	56 days	No effect on growth	0.157	Salazar et al. 1987
Pacific oyster (3.5-25 mm), <i>Crassostrea gigas</i>	TBT (field)	-	2-5 mo	Reduced growth rate Normal growth rate	0.040 0.010	Stephenson 1991
Pacific oyster (fertilized eggs), <i>Crassostrea gigas</i>	TBTO	-	24 hr	LC50 Delayed development	7.0 1.8	Osada et al. 1993
Pacific oyster (straight-hinge larvae), <i>Crassostrea gigas</i>	TBTO	-	24 hr	LC50	15.0	Osada et al. 1993
Pacific oyster (spat), <i>Crassostrea gigas</i>	TBTO	-	48 hr	LC50	35.0	Osada et al. 1993
Pacific oyster (24-h-old), <i>Crassostrea gigas</i>	TBTA	-	12 days	LC50	0.04	His 1996
Eastern oyster (2.7-5.3 cm), <i>Crassostrea virginica</i>	- ^d	-	67 days	Decrease in condition index (body weight)	0.73	Valkirs et al. 1985
Eastern oyster (2.7-5.3 cm), <i>Crassostrea virginica</i>	- ^d	-	67 days	No effect on survival	1.89	Valkirs et al. 1985
Eastern oyster (adult), <i>Crassostrea virginica</i>	- ^c	33-36	57 days	Decrease in condition index	0.1	Henderson 1986
Eastern oyster (adult), <i>Crassostrea virginica</i>	- ^c	33-36	30 days	LC50	2.5	Henderson 1986

Table 6. Other Data on Effects of Tributyltin on Aquatic Organisms (continued)

<u>Species</u>	<u>Chemical^a</u>	<u>Salinity (g/kg)</u>	<u>Duration</u>	<u>Effect</u>	<u>Concentration ($\mu\text{g/L}$)^b</u>	<u>Reference</u>
<u>SALTWATER SPECIES</u>						
Eastern oyster (embryo), <i>Crassostrea virginica</i>	TBTCI	18-22	48 hr	Abnormal shell development	0.77	Roberts 1987
Eastern oyster (juvenile), <i>Crassostrea virginica</i>	TBTO	11-12	96 hr	EC50; shell growth	0.31	Walker 1989b
Eastern oyster (adult), <i>Crassostrea virginica</i>	- ^c	-	8 wks	No affect on sexual development, fertilization	1.142	Roberts et al. 1987
Eastern oyster (adult), <i>Crassostrea virginica</i>	TBT	-	21 wks	Immune response not weakened	0.1	Anderson et al. 1996
European flat oyster (spat), <i>Ostrea edulis</i>	TBTO	30	20 days	Significant reduction in growth	0.01946	Thain and Waldock 1985
European flat oyster (spat), <i>Ostrea edulis</i>	- ^c	28.5-34.2	45 days	Decreased growth	0.2392	Thain and Waldock 1985; Thain 1986
European flat oyster (spat), <i>Ostrea edulis</i>	- ^c	28.5-34.2	45 days	70% mortality	2.6	Thain and Waldock 1985; Thain 1986
European flat oyster (spat), <i>Ostrea edulis</i>	- ^c	-	20 days	Reduced growth	0.02	Thain and Waldock 1986
European flat oyster (adult), <i>Ostrea edulis</i>	- ^c	28-34	75 days	Complete inhibition of larval production	0.24	Thain 1986
European flat oyster (adult), <i>Ostrea edulis</i>	- ^c	28-34	75 days	Retardation of sex change from male to female	0.24	Thain 1986
European flat oyster (adult), <i>Ostrea edulis</i>	- ^c	28-34	75 days	Prevented gonadal development	2.6	Thain 1986
European flat oyster (140-280 mg), <i>Ostrea edulis</i>	- ^c	-	56 days	No effect on growth	0.157	Salazar et al. 1987

Table 6. Other Data on Effects of Tributyltin on Aquatic Organisms (continued)

<u>Species</u>	<u>Chemical^a</u>	<u>Salinity (g/kg)</u>	<u>Duration</u>	<u>Effect</u>	<u>Concentration ($\mu\text{g/L}$)^b</u>	<u>Reference</u>
<u>SALTWATER SPECIES</u>						
Native Pacific oyster (100-300 mg), <i>Ostrea lurida</i>	- ^c	-	56 days	No effect on growth	0.157	Salazar et al. 1987
Quahog clam (embryo, larva), <i>Mercenaria mercenaria</i>	TBTO	-	14 days	Reduced growth	\$0.010	Laughlin et al. 1987;1988
Clam (adult), <i>Macoma nasuta</i>	- ^c	-	110 days	No effect on condition	0.204	Salazar et al. 1987
Quahog clam (veligers), <i>Mercenaria mercenaria</i>	TBTO	-	8 days	Approx. 35% dead; reduced growth; \$1.0 $\mu\text{g/L}$ 100 mortality	0.6	Laughlin et al. 1987;1989
Quahog clam (post larva), <i>Mercenaria mercenaria</i>	TBTO	-	25 days	100% mortality	10	Laughlin et al. 1987;1989
Quahog clam (larva), <i>Mercenaria mercenaria</i>	TBTCl	18-22	48 hr	Delayed development	0.77	Roberts 1987
Common Pacific Littleneck (adult), <i>Protothaca stamina</i>	TBTO	33-34	96 hr	100% survival	\$2.920	Salazar and Salazar 1989
Copepod (subadult), <i>Eurytemora affinis</i>	TBT	10	72 hr	LC50	0.5	Bushong et al. 1988
Copepod (subadult), <i>Eurytemora affinis</i>	TBT	10	72 hr	LC50	0.6	Bushong et al. 1988
Copepod, <i>Acartia tonsa</i>	TBTO	-	6 days	EC50	0.3893	U'ren 1983
Copepod (adult), <i>Acartia tonsa</i>	TBTO	28	5 days	Reduced egg production	0.010	Johansen and Mohlenberg 1987
Copepod (nauplii), <i>Acartia tonsa</i>	TBTCl	10-12	9 days	Reduced survival	\$0.029	Bushong et al. 1990

Table 6. Other Data on Effects of Tributyltin on Aquatic Organisms (continued)

<u>Species</u>	<u>Chemical^a</u>	<u>Salinity (g/kg)</u>	<u>Duration</u>	<u>Effect</u>	<u>Concentration ($\mu\text{g/L}$)^b</u>	<u>Reference</u>
<u>SALTWATER SPECIES</u>						
Copepod (nauplii), <i>Acartia tonsa</i>	TBTCl	10-12	6 days	Reduced survival; no effect 0.012 $\mu\text{g/L}$	0.023	Bushong et al. 1990
Copepod (nauplii), <i>Acartia tonsa</i>	TBTCl	10-12	6 days	Reduced survival; no effect 0.010 $\mu\text{g/L}$	0.024	Bushong et al. 1990
Copepod (nauplii), <i>Acartia tonsa</i>	TBTCl	18	8 days	Inhibition of development EC50 (survival)	0.003 0.015-0.020	Kusk and Peterson 1997
Amphipod (larva, juvenile), <i>Gammarus oceanicus</i>	TBTO	7	8 wk	100% mortality	2.920	Laughlin et al. 1984b
Amphipod (larva, juvenile), <i>Gammarus oceanicus</i>	TBTF	7	8 wk	100% mortality	2.816	Laughlin et al. 1984b
Amphipod (larva, juvenile), <i>Gammarus oceanicus</i>	TBTO	7	8 wk	Reduced survival and growth	0.2920	Laughlin et al. 1984b
Amphipod (larva, juvenile), <i>Gammarus oceanicus</i>	TBTF	7	8 wk	Reduced survival and increased growth	0.2816	Laughlin et al. 1984b
Amphipod, <i>Gammarus</i> sp.	TBTCl	10	24 days	No effect	0.579	Hall et al. 1988b
Amphipod (adult), <i>Orchestia traskiana</i>	TBTO	30	9 days	Approx. 80% mortality	9.732	Laughlin et al. 1982
Amphipod (adult), <i>Orchestia traskiana</i>	TBTF	30	9 days	Approx. 90% mortality	9.732	Laughlin et al. 1982
Amphipod (adult), <i>Eohaustorius estuarius</i>	TBTCl (96%)	28.8-29.5	10 days	BCF = 41,200 (no plateau)	0.48	Meador et al. 1993

Table 6. Other Data on Effects of Tributyltin on Aquatic Organisms (continued)

<u>Species</u>	<u>Chemical^a</u>	<u>Salinity (g/kg)</u>	<u>Duration</u>	<u>Effect</u>	<u>Concentration ($\mu\text{g/L}$)^b</u>	<u>Reference</u>
<u>SALTWATER SPECIES</u>						
Amphipod (adult), <i>Eohaustorius washingtonianus</i>	TBTCl (96%)	32.7	10 days	BCF = 60,300 (no plateau)	109	Meador 1997
Amphipod (adult), <i>Rhepoxynius abronius</i>	TBTCl (96%)	32.3	10 days	BCF = 1,700 (no plateau)	660	Meador 1997
Grass shrimp, <i>Palaemonetes pugio</i>	TBTO (95%)	9.9-11.2	20 min	No avoidance	30	Pinkney et al. 1985
Grass shrimp, <i>Palaemonetes pugio</i>	TBTO	20	14 days	Telson regeneration retarded; molting retarded	0.1	Khan et al. 1993
Mud crab (larva), <i>Rhithropanopeus harrisii</i>	TBTO	15	15 days	Reduced developmental rate and growth	14.60	Laughlin et al. 1983
Mud crab (larva), <i>Rhithropanopeus harrisii</i>	TBTS	15	15 days	Reduced developmental rate and growth	18.95	Laughlin et al. 1983
Mud crab (larva), <i>Rhithropanopeus harrisii</i>	TBTO	15	15 days	63% mortality	>24.33	Laughlin et al. 1983
Mud crab (larva), <i>Rhithropanopeus harrisii</i>	TBTS	15	15 days	74% mortality	28.43	Laughlin et al. 1983
Mud crab (zoea), <i>Rhithropanopeus harrisii</i>	TBTO	15	20 days	LC50	13.0	Laughlin and French 1989
Mud crab (zoea), <i>Rhithropanopeus harrisii</i>	TBTO	15	40 days	LC50	33.6	Laughlin and French 1989
Mud crab, <i>Rhithropanopeus harrisii</i>	TBTO	15	6 days	BCF=24 for carapace	5.937	Evans and Laughlin 1984
Mud crab, <i>Rhithropanopeus harrisii</i>	TBTO	15	6 days	BCF=6 for hepatopancreas	5.937	Evans and Laughlin 1984
Mud crab, <i>Rhithropanopeus harrisii</i>	TBTO	15	6 days	BCF=0.6 for testes	5.937	Evans and Laughlin 1984

Table 6. Other Data on Effects of Tributyltin on Aquatic Organisms (continued)

<u>Species</u>	<u>Chemical^a</u>	<u>Salinity (g/kg)</u>	<u>Duration</u>	<u>Effect</u>	<u>Concentration ($\mu\text{g/L}$)^b</u>	<u>Reference</u>
<u>SALTWATER SPECIES</u>						
Mud crab, <i>Rhithropanopeus harrisi</i>	TBTO	15	6 days	BCF=41 for gill tissue	5.937	Evans and Laughlin 1984
Mud crab, <i>Rhithropanopeus harrisi</i>	TBTO	15	6 days	BCF=1.5 for chelae muscle	5.937	Evans and Laughlin 1984
Fiddler crab, <i>Uca pugilator</i>	TBTO	25	#24 days	Retarded limb regeneration and molting	0.5	Weis et al. 1987a
Fiddler crab, <i>Uca pugilator</i>	TBTO	25	3 weeks	Reduced burrowing	0.5	Weis and Perlmutter 1987
Fiddler crab, <i>Uca pugilator</i>	TBTO	25	7 days	Limb malformation	0.5	Weis and Kim 1988; Weis et al. 1987a
Blue crab (6-8-day-old embryos), <i>Callinectes sapidus</i>	TBT	28	4 days	EC50 (hatching)	0.047	Lee et al. 1996
Brittle star, <i>Ophioderma brevispina</i>	TBTO	18-22	4 wks	Retarded arm regeneration	- 0.1	Walsh et al. 1986a
Atlantic menhaden (juvenile), <i>Brevoortia tyrannus</i>	TBTCl	10	28 days	No effect	0.490	Hall et al. 1988b
Atlantic menhaden (juvenile), <i>Brevoortia tyrannus</i>	TBTO	9-11	-	Avoidance	5.437	Hall et al. 1984
Chinook salmon (adult), <i>Oncorhynchus tshawytscha</i>	TBTO	28	96 hr	BCF=4,300 for liver	1.49	Short and Thrower 1986a, 1986c
Chinook salmon (adult), <i>Oncorhynchus tshawytscha</i>	TBTO	28	96 hr	BCF=1,300 for brain	1.49	Short and Thrower 1986a, 1986c

Table 6. Other Data on Effects of Tributyltin on Aquatic Organisms (continued)

<u>Species</u>	<u>Chemical^a</u>	<u>Salinity (g/kg)</u>	<u>Duration</u>	<u>Effect</u>	<u>Concentration ($\mu\text{g/L}$)^b</u>	<u>Reference</u>
<u>SALTWATER SPECIES</u>						
Chinook salmon (adult), <i>Oncorhynchus tshawytscha</i>	TBTO	28	96 hr	BCF=200 for muscle	1.49	Short and Thrower 1986a,1986c
Mummichog (juvenile), <i>Fundulus heteroclitus</i>	TBTO	2	6 wks	Gill pathology	17.2	Pinkney 1988; Pinkney et al. 1989a
Mummichog, <i>Fundulus heteroclitus</i>	TBTO	9.9-11.2	20 min	Avoidance	3.7	Pinkney et al. 1985
Mummichog (embryo), <i>Fundulus heteroclitus</i>	TBTO	25	10 days	Teratology	30	Weis et al. 1987b
Mummichog (5.3 cm; 1.8 g), <i>Fundulus heteroclitus</i>	TBTO (95%)	15 16-19.5	96 hr 6 wks	LC50 NOEC	17.2 2.000	Pinkney et al. 1989a
Inland silverside (larva), <i>Menidia beryllina</i>	TBTCl	10	28 days	Reduced growth	0.093	Hall et al. 1988b
Three-spined stickleback (45- 60 mm), <i>Gasterosteus aculeatus</i>	TBTO (painted panels)	15-35	7.5 mo	80% mortality (2 months) Histological effects	10 2.5	Holm et al. 1991
California grunion (gamete through embryo), <i>Leuresthes tenuis</i>	^c	-	10 days	Significantly enhanced growth and hatching success	0.14-1.71	Newton et al. 1985
California grunion (gamete through embryo), <i>Leuresthes tenuis</i>	^c	-	10 days	Significantly enhanced growth and hatching success	0.14-1.72	Newton et al. 1985
California grunion (gamete through embryo), <i>Leuresthes tenuis</i>	^c	-	10 days	50% reduction in hatching success	74	Newton et al. 1985

Table 6. Other Data on Effects of Tributyltin on Aquatic Organisms (continued)

<u>Species</u>	<u>Chemical^a</u>	<u>Salinity (g/kg)</u>	<u>Duration</u>	<u>Effect</u>	<u>Concentration (µg/L)^b</u>	<u>Reference</u>
<u>SALTWATER SPECIES</u>						
California grunion (embryo), <i>Leuresthes tenuis</i>	^c	-	10 days	No adverse effect on hatching success or growth	0.14-1.72	Newton et al. 1985
California grunion (larva), <i>Leuresthes tenuis</i>	^c	-	7 days	Survival increased as concentration increased	0.14-1.72	Newton et al. 1985
Striped bass (juvenile), <i>Morone saxatilis</i>	TBTO (95%)	9-11	-	Avoidance	24.9	Hall et al. 1984
Striped bass (juvenile), <i>Morone saxatilis</i>	TBT (painted panels)	13.0-15.0	14	NOEC (serum ion concentrations and enzyme activity)	1.09	Pinkney et al. 1989b
Striped bass (juvenile), <i>Morone saxatilis</i>	TBT (painted panels)	1.1-3.0 1.9-3.0 12.2-14.5	6 days 7 days 7 days	NOEC 0.067; LOEC 0.766 NOEC 0.444; LOEC 1.498 LOEC >0.514	-	Pinkney et al. 1990
Speckled sanddab (adult), <i>Citharichthys stigmaeus</i>	TBTO	33-34	96 hr	LC50	18.5	Salazar and Salazar 1989
Stripped mullet (3.2 g); <i>Mugil cephalus</i>	TBTO (96%)	-	8 wks	BCF 3,000 (no plateau) BCF 3,600 (no plateau)	0.122 0.106	Yamada and Takayanagi 1992
Starry flounder (<1-year-old), <i>Platichthys stellatus</i>	TBTCl (96%)	30.2	10 days	BCF 8,700 (no plateau)	194	Meador 1997

^a TBTA = tributyltin acetate; TBTCI = tributyltin chloride; TBTF = tributyltin fluoride; TBTO = tributyltin oxide; TBTS = tributyltin sulfide. Percent purity is given in parentheses when available.

^b Concentration of the tributyltin cation, not the chemical. If the concentrations were not measured and the published results were not reported to be adjusted for purity, the published results were multiplied by the purity if it was reported to be less than 95%.

^c The test organisms were exposed to leachate from panels coated with antifouling paint containing tributyltin.

^d The test organisms were exposed to leachate from panels coated with antifouling paint containing a tributyltin polymer and cuprous oxide. Concentrations of TBT were measured and the authors provided data to demonstrate the similar toxicity of a pure TBT compound and the TBT from the paint formulation.

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