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Environmental Protection
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Office of Water
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Ambient Aquatic Life Water Quality Criteria for Tributyltin (TBT) - Draft

ADDENDUM

The Environmental Protection Agency (EPA) issued a draft ambient water quality criteria document for tributyltin (TBT) on August 7, 1997. This document was issued to the public for scientific and technical input through a notice of availability published in the *Federal Register* (62 FR 42554). After consideration of peer review, scientific and technical input, and additional data which became available after the draft was published, EPA is issuing new ambient water quality criteria for TBT for scientific and technical input.

A comprehensive literature search for toxicity information on TBT was conducted before the draft criteria document for TBT was issued in 1997. In preparing the new TBT criteria document, more recent aquatic life toxicity data have been considered (* see new references at end of Addendum). The major effect of inclusion of this new information on TBT is the lowering of the draft saltwater four day average, once in three year exceedence, chronic criterion of 0.01 ug/l to a new chronic criterion of 0.001 ug/l.

EPA's Office of Pesticide Programs (OPP) has recently updated its Environmental Risk Characterization for TBT. EPA's Office of Water (OW) has coordinated closely with OPP in preparing the new ambient water quality criteria document for TBT. This collaboration has enabled OW to access more recent information on the toxicity of TBT. Consideration of the more recent data available for TBT leads to the following conclusions:

* TBT is an immunosuppressing agent and an endocrine disruptor

* TBT biomagnifies through the food chain and has been found in tissues of marine mammals

* TBT causes adverse reproductive and developmental effects in aquatic organisms at very low concentrations

* TBT degrades much more slowly in sediment than earlier studies had indicated and is likely to persist in sediments at concentrations which cause adverse biological effects

After considering peer review, scientific and technical input from the public, and more recent data, EPA has set the new saltwater chronic criterion for TBT at 0.001 ug/l.

References:

* Fisher, W.S., L.M. Oliver, W.W. Walker, C.S. Manning, T.F. Lytle. 1999. Decreased resistance of eastern oysters (*Crassostrea Virginica*) to a protozoan pathogen (*Perkinsus marinus*) after sublethal exposure to tributyltin oxide. Marine Environ. Res. 47: 185-201.

* Matthiessen, P. and P.E. Gibbs. 1998. Critical appraisal of the evidence for tributyltin-mediated endocrine disruption in mollusks. Environ. Toxicol. Chem. 17: 37-43.

* Other references are available, but were not relied upon for derivation of the criteria

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-supression in snails and bivalves, and is very 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 Tribes to establish water quality standards for TBT.

CRITERIA:

Freshwater:

For TBT, the criterion to protect freshwater aquatic life from chronic toxic effects is 0.063 ug/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 ug/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 0.001 ug/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 0.38 ug/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 significantly from the criterion that was originally proposed for public review (0.010 ug/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

(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 0.001 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.

AMBIENT AQUATIC LIFE WATER QUALITY CRITERIA FOR

TRIBUTYLTIN

CAS Registry Number (See Text)

U.S. ENVIRONMENTAL PROTECTION AGENCY

OFFICE OF WATER
OFFICE OF SCIENCE AND TECHNOLOGY
HEALTH AND ECOLOGICAL CRITERIA DIVISION
WASHINGTON D.C.

OFFICE OF RESEARCH AND DEVELOPMENT
MID-CONTINENT ECOLOGY DIVISION
DULUTH, MINNESOTA
ATLANTIC ECOLOGY DIVISION
NARRAGANSETT, RHODE ISLAND

NOTICES

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

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

FOREWORD

Under Section 304(a)of the Clean Water Act (CWA) of 1977 (P.L. 95-217), the U.S. Environmental Protection Agency (EPA)is to periodically revise water quality criteria to accurately reflect the latest scientific knowledge. This document is a revision of previous criteria based upon consideration of scientific and technical input received from other federal agencies, state agencies, special interest groups, and individual scientists. Criteria contained in this document replace any previously published U.S. EPA aquatic life criteria for tributyltin (TBT).

This document provides guidance to States and Tribes authorized to establish water quality standards under the CWA to protect aquatic life from toxic effects of TBT. Under the CWA, States and Tribes are to establish water quality standards to protect designated uses. While this document constitutes the U.S. EPA’s scientific recommendations regarding ambient concentrations of TBT, this document does not substitute for the CWA or the U.S. EPA’s regulations, nor is it a regulation itself. Thus, it cannot impose legally binding requirements on the U.S. EPA, States, Tribes or the regulated community, and might not apply to a particular situation based upon the circumstances. The U.S. EPA may change this guidance in the future.

ACKNOWLEDGMENTS

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Introduction

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 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, 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).

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 µg/L at pH of 6.6, 31,000 µg/L at pH of 8.1 and 30,000 µg/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 in water to months or more than a year in sediments (Clark et al. 1988; de Mora et al. 1989; Lee et al. 1987; Maguire and Tkacz 1985; Seligman et al. 1986, 1988, 1989; Stang and Seligman 1986). Breakdown products include di-, monobutyltins and tin with some methyltins detected (Yonezawa et al. 1994) when sulfate reducing conditions were present. 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 TBT with the green alga, Ankistrodesmus falcatus, 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 and evidence suggests that competitive inhibition of cytochrome P450-dependent aromatase is probably occurring in TBT exposed gastropods (Matthiessen and Gibbs 1998). TBT may interfer 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). However, TBT readily sorbs to sediments and suspended solids and can persist there (Cardarelli and Evans 1980; Harris et al. 1996; Seligman et al. 1996). TBT accumulates in sediments with sorption coefficients which range from $1.1x10^2$ to $8.2x10^3$ L/Kg and desorption appears to be a two step process (Unger et al. 1987,1988). At environmentally realistic concentrations of 10 ng/L, TBT partitioning coefficients were closer to $2.5\ x10^4$ (Langston and Pope, 1995). 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.

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 (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 TBTCl (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 TBTCl, 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), hereinafter 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 mollusicidal 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. The freshwater clam, Elliptio camplanatus, had an unusually high LC50 value of 24,600 µg/L.

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. 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. Removing the least sensitive genera, Elliptio, the remainder differ from one another by a maximum factor of 11.2 times. 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, Acatia tonsa (Kusk and Petersen 1997) to 282.2 µg/L for adult Pacific oysters, Crassostrea gigas (Thain 1983). The 96-hr LC50s for six saltwater fish species range from 1.460 µg/L for juvenile chinook salmon, Oncorhynchus tshawytscha (Short and Thrower 1986b) 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 crustaceans Acanthomysis sculpta and Metamysidopsis elongata were slightly more sensitive to TBT than adults (Davidson et al. 1986a,1986b; Valkirs et al. 1985; Salazar and Salazar 1989). Four genera of amphipods were tested and sensitivity to TBT ranged from 1.3 to 22.8 µg/L. As with bivalve molluscs and other crustaceans, one genus (Gammarus) demonstrated greater sensitivity to TBT at the younger life-stage (Bushong et al. 1988).

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 12 most sensitive genera differ by a factor of less than four. Included within these genera are four species of molluscs, eight species of crustaceans, and one species of fish. The saltwater Final Acute Value (FAV) for TBT was calculated to be 0.7673 µg/L (Table 3), which is greater than the lowest saltwater Species Mean Acute Value of 0.61 µg/L. The saltwater Criterion Maximum Concentation is 0.3836 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.

Daphnia 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#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 F_1 generation) was conducted with the 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 concentrations similar to those used by Bailey et al. (1991). 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, 0.0077, 0.0334, and 0.1246 µgTBT/L for one year in the previous study (Bailey et al. 1991), 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 0.0334 µg/L and the no observed effect concentration (NOEC) of 0.0077 µg/L which is 0.0153 µg/L. Survival and growth were not affected at any TBT tested concentration. An acute-chronic ratio of 4,752 can be calculated using the acute value from this test of 72.7 µg/L. The acute-chronic ratio for N. lapillus is about 108 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 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 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 0.0605 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-one species of algae and diatoms were tested in fresh and salt water. The saltwater species are more sensitive to TBT than the freshwater species for which data are available. No explanation is apparent.

Blanck et al. (1984) 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, Nitzshia sp., flagellate green alga, 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 Skeletonema 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.

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 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, Oncorhyncus 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 µgTBT/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 and 460, respectively. Goldfish, Carassius auratus, reached a much higher BCF (1,976) in the whole body than the other fish species tested.

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 three species of bivalve molluscs, two species of snails, and a fish (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 and Salazar (1987) observed BCFs of 10,400 to 37,500 after 56 days of exposure. BAFs from field deployments of mussels were similar to 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 the snail 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. One fish species, Poecilia reticulatus, was exposed in salt water to 0.28 µg/L for 14 days and a plateau BCF of 240 was demonstrated (Tsuda et al. 1990b). The BCF agrees reasonably well with the freshwater BCF (460) with the same species.

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

Some data (Table 6) were located on the lethal and sublethal effects of TBT on aquatic species that were insufficient to meet the criteria for inclusion in the tables for acute toxicity, chronic toxicity, plant toxicity, or bioconcentration in this document. These data are potentially useful and sometimes support data in other tables. Sometimes the data are unique and useful to evaluate TBT affects on aquatic organisms.

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 herbivor, 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. 1990). 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 tricornutum, Skeletonema costatum,

and Thallassiosira 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 <0.5 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).

The freshwater invertebrates, a rotifer (Brachionus calyciflorus) and a coelentrate (Hydra sp.), showed widely differing sensitivites 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 a eight-day exposure. 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. Larvae of the mud crab, Rhithropanopeus harrisii, 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. harrisii,

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). 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%.

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. Triebskorn et al. (1994) found reduced growth and behavior changes in the fish at 21 days when exposed to 0.5 µg/L. Hall et al. (1988b) observed reduced growth in the inland silverside, Menidia beryllina, at 0.093 µg/L in a 28 day exposure. The frog, Rana temporaria, has a LC50 of 28.2 µg/L for a 5-day exposure to TBT.

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 990 (Triebskorn 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).

TBT has been shown to produce the superimposition of male sexual characteristics on female neogastropod (stenoglossan) snails (Smith 1981b, Gibbs and Bryan 1987) 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 volume^3 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 dogwhinkle (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. 1988a). 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 µgTBT/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 Pattisima 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 reticulta), 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 fuctional male (Bryan et al. 1986; Gibbs et al. 1987,1988; Gibbs and Bryan 1986,1987). 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 were not blocked and indirect development (plantonic larvae) facilitating recruitment from other areas 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 1). 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

Text Table 1. Summary of Available Laboratory and Field Studies relating the Extent of Imposex of Female Snails, Measured by Relative Penis Size (Volume^3 Female Penis/Male Penis = RPSI) and the Vas Deferens Sequence Index (VDSI), as a Function of Tributyltin Concentration in Water and Dry Tissue

		TBT Concentration		Imposex			
Species	Experimental Design	Water µg/L	Snail Tissue µg/g dry	RPSI	VDSI	Comments	Reference
Eastern mud snail, Ilyanassa obsoleta	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	
Snail, Hinia reticulata	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, Thais orbita	Field-Queens-cliff, UK	-	0.365*	19.55	-	100% incidence	Foale 1993
	-Sandringham	-	0.224*	12.16	-	100% incidence	
	-Brighton	-	<0.002*	7.34	-	100% incidence	
	-Portarlington	-	0.255*	3.67	-	92.3% incidence	
	-Mornington	-	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	

Text Table 1. Continued

		TBT Concentration		Imposex			
Species	Experimental Design	Water µg/L	Snail Tissue µg/g dry	RPSI	VDSI	Comments	Reference
File dogwhinkle, *Nucella lima*	Field -Auke Bay, AK -Auke Bay, AK	- -	ND(<0.01) 0.03-0.16	0.0 14-34	0.0 2.2-4.3	0% incidence 100% incidence, reduced abundance	Short et al. 1989
Atlantic dogwhinkle, (adults), *Nucella lapillus*	Crooklets Beach, UK Laboratory: 2 year exposure	<0.0012* 0.0036* 0.0083* 0.046* 0.26*	0.14-0.25* 0.41* 0.74* 4.5* 8.5*	2-65 10/14.2 43.8 56.4 63.3	2.9 3.7/3.7 3.9 4.0 4.1	- - - - Some sterilization	Bryan et al. 1987a
Atlantic dogwhinkle, *Nucella lapillus*	Laboratory, spires painted, 8 mo.	-	~5.1*	10-50	-	-	Bryan et al. 1987b
Atlantic dogwhinkle, (egg capsule to adult), *Nucella lapillus*	Crooklets Beach, UK Laboratory; 2 year exposure	<0.0012 0.0036 0.0093 0.049 0.24	0.19 0.58 1.4 4.1 7.7	3.7 48.4 96.6 109 90.4	3.2 4.4 5.1 5.0 5.0	Normal females 1/3 sterile, 160 capsules All sterile, 2 capsules All sterile, 0 capsules All sterile, 0 capsules	Gibbs et al. 1988

Text Table 1. Continued

		TBT Concentration		Imposex			
Species	Experimental Design	Water µg/L	Snail Tissue µg/g dry	RPSI	VDSI	Comments	Reference
European sting winkle, Ocenebra erinacea	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 somewhat deformed	
		-	0.122	7.0	-	Females somewhat deformed	
		-	0.703	36.0	-	Females highly deformed	
		-	0.764	52.7	-	Females highly deformed	
		-	0.527	46.5	-	Females highly deformed	
		-	0.488	42.3	-	Females highly deformed	
		-	0.366	0.04	-	Females somewhat deformed	
		-	0.253	33.9	-	Females highly deformed	
		-	0.832	58.0	-	Females highly deformed	
		-	1.010	79.3	-	Females highly deformed	
		-	0.510	59.7	-	Females highly deformed	

Atlantic dogwhinkle, Nucella lapillus	Field, S.W. UK	0.002-0.005*	<0.5*	-20-60	-2.0-4.5	Limited sterility	Gibbs et al. 1987
		-0.010	0.5-1.0*	-30-70	-4.5-6.0	-50% sterile	
		-0.017-0.025	<1.0*	-30-100	-4.5-6.0	All sterile	
Atlantic dogwhinkle, Nucella lapillus	Port Joke, UK	-	0.11*	0.0	-	0% aborted egg capsules	Gibbs and Bryan 1986; Gibbs et al. 1987
	Crooklets Beach	-	0.21*	2.0	-	0% aborted egg capsules	
	Meadfoot	-	0.32*	30.6	-	15% aborted egg capsules	
	Renney Rocks	-	0.43*	38.9	-	38% aborted egg capsules	
	Batten Bay	-	1.54*	22.9	-	79% aborted egg capsules	
Atlantic dogwhinkle, Nucella lapillus	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	
		0.1246	4.85*	63.40	4.25	18.9% reduced growth, 100% imposex	
Atlantic dogwhinkle, Nucella lapillus	Laboratory, flow-through, one year	<0.0015	<0.10	0.07	1.28	Control, 42.2% imposex	Harding et al. 1996
		<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	

* Concentrations changed from µg Sn/L or µg Sn/g to µg TBT/L or µg TBT/g dry weight. Dry weight estimated as 20% of wet weight.

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 and there was a 40 to 100% incidence of imposex. Short et al. (1989) collected file dogwinkle 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. undersized egg capsules produced. Concentrations of TBT in females were 0.19 µg/g in the field, 0.58 µg/g in the 0.0036 µg/L treatment and from 1.39 to 7.71 µg/g in >0.0093 µg/L. Similar concentrations of TBT (9.7 µg/g) 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 µg/L. 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 µg/g. In two studies conducted concurrently with N. 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 $0.0027 µg/L at the end of the study. Harding et al. (1996) exposed the offspring from parents exposed the study by Bailey et al. (1991) for one 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. Also, the RPSI and VDSI values were higher for the same treatments in the second generation. Harding et al. (1996) found $98.7% imposex in females at TBT concentrations $0.0026 µg/L.

In summary, in both field and laboratory studies, concentrations of TBT in water of about 0.001 µg/L or less and in tissues of about 0.2 µg/g or less appear to not cause imposex in N. lapillus. Imposex begins to occur, and cause some reproductive failure at about 0.004 µg/L with complete sterility

occurring after chronic exposure of sensitive early life-stages at ≥0.009 µg/L and for less sensitive stages at 0.02 µg/L in some studies and greater

than 0.2 µg/L in others. If N. lapillus or similarly sensitive species are ecologically important at specific sites, TBT concentrations ≤0.001 µg/L may be required to limit development of imposex.

Reproductive abnormalities have also been observed in the European flat oyster (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. 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.

Four species of snails (Hinia reticulata, Thais orbita, T. clavigera, Ocenebra erinacea) not resident to North America also demonstrated imposex effects when exposed to TBT in field studies (Text Table 1). The snail H. reticulata is less sensitive to TBT than other snails having higher body burdens (>1.5 µg/g) before showing affects of imposex. Thais sp. showed high imposex incidence at tissue concentrations as low as 0.005 µg/g and no imposex at other locations with tissue concentrations of 0.108 µg/g. Ocenebra erinacea did not show imposex in a field study at body burdens as high as 0.185 µg/g, but females were deformed at all higher concentrations.

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. 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 $\geq$0.8604 µg/L (Robert and His 1981). Waldock 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. Shell thickening in Crassostrea gigas was associated with tissue concentrations of $\geq$0.2 mg/kg (Davies et al. 1988). 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 $\geq$0.1 µg/L (Henderson 1986; Valkirs et al. 1985). Salazar et al. (1987) found no effect on growth after 56 days exposure to 0.157 µg/L to the oysters C. virginica, 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 lessor 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 $\geq$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 harrisii, were reduced by a 15-day exposure to ≥14.60 µg/L. R. harrisii 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. Harrisii) 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). Arm regeneration was reduced in brittle stars exposed to 0.1 µg/L (Walsh et al. 1986a). 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 1,490 µg/L for 96 hours (Short and Thrower 1986a,1986c).

TBT concentrations less than the Final Chronic Value of 0.0605 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 2). Survival of the copepod A. tonsa was significantly reduced in three tests in 0.029, 0.023 and 0.024 µg/L; 30, 27

and 51 percent of control survival, respectively (Bushong et al. 1990). Survival decreased with increase in exposure concentration but was not significantly affected in the 0.012 µg/L exposure concentration.

Laughlin et al. (1987, 1988) observed a significant decrease in growth of hard clam (Mercenaria mercinaria) larvae exposed for 14 days to ≥0.01 µg/L (Text Table 2). 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 2). 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

Text Table 2. Summary of laboratory and field data on the effects of tributyltin on saltwater organisms at concentrations less than the Final Chronic Value of 0.0605 µg/L

Species	Experimental Design[a]	Concentration (µg/L)	Response	Reference
		Measured		
Copepod (nauplii-adult), Acartia tonsa	#1: F,M, 9-day duration, $10 copepods/replicate, 4 replicates	control 0.029 0.05-0.5	77% survival 23% survival[b] 0-2% survival[b]	Bushong et al. 1990
	#2: F,M, 6-day duration, $10 copepods/replicate, 4 replicates	control 0.007-0.012 0.023 0.048-0.102	71% survival 32% survival 19% survival[b] 0-14% survival[b]	
	#3: F,M, 6-day duration, $10 copepods/replicate, 4 replicates	control 0.006-0.010 0.024 0.051-0.115	59% survival 44-46% survival 30% survival[b] 2-35% survival[b]	
		Nominal		
Hard clam (4 hr larvae - metamorphosis), Mercenaria mercenaria	R,M, 14-day duration, <150 larvae/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 (Value length)[b]	Laughlin et al. 1987,1988
		Nominal		

Pacific oyster (spat), Crassostrea gigas	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 (Value length) 0-72% Growth (Value length)[b]	Lawler and Aldrich 1987

Text Table 2.
(Continued)

Species	Experimental Design[a]	Concentration (µg/L)	Response	Reference
		Measured		
	Field	0.011-0.015 ~0.018-0.060	No shell thickening Shell thickening and decreased meat weight	
Pacific oyster (larvae and spat), Crassostrea gigas	R,M/N, 21-day duration, 75,000 larvae/replicate	0.24,0.29,0.69	Mortality 100% by day 1	Springborn Bionomics, Inc. 1984a
		Measured		
Pacific oyster (spat), Crassostrea gigas	R,M, 4 week duration, 4 replicates, 30 spat each	Control,0.005,0.010,0.015,0.020	Growth decreased 79% in 0.005, 78% in 0.010, 78% in 0.015, 84% in 0.020	Nell and Chvojka 1992
		Nominal		
		control, 0.1, 0.05, 0.025	Mortality 100% in 0.05 and 1.0 µg/L; 86% in 0.025 µg/L	
		Nominal		
European oyster (spat), Ostrea edulis	R,N, 20-day duration, 50 spat/treatment	control 0.02-2.0 control 0.02-2.0	100 length 76-81% length[b] 202% weight gain 151-50% weight gain	Thain and Waldock 1985

		Nominal		
European oyster (adult), Ostrea edulis	R,N, 96-hr duration	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.

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 µg/L but not at higher concentrations. Decreased weights of oyster meats were associated with locations where there was shell thickening. Survival of Crassostrea gigas larvae exposed for 21 days was reduced in 0.025 µg/L (Springborn Bionomics 1984a). No larvae survived in ≥0.050 µg/L.

Growth of spat of the European oyster (Ostrea edulis) was reduced at ≥0.02 µg/L (Thain and Waldock 1985; Text Table 2). Spat exposed to TBT in static tests were 82% of control lengths and 75% of control weights; extent of impact increased with increased exposure. In these static and flow-through tests at exposures at about 0.02 µg/L, weight gain was identical; i.e., 35% of controls. Growth of larger spat was marginally reduced by 0.2392 µg/L (Thain 1986; Thain and Waldock 1985).

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 0.0605 µg/L derived using Final Acute Values and Acute-Chronic Ratios from Table 3. Therefore, EPA believes the Final Chronic Value should be lowered to 0.001 µg/L to limit unacceptable impacts on A. tonsa, Mercenaria mercenaria, Crassostrea gigas and Ostrea edulis observed at 0.02 µg/L. At this criteria concentration, imposex would not be expected in Ilyanassa obsoleta, N. lapillus and similarly sensitive neogastropods; populations of N. lapillus and similarly sensitive snails with direct development would not be impacted and growth of M. mercenaria would not be lowered.

Unused Data

Some data concerning the effects of TBT on aquatic organisms were not

used because the tests were conducted with species that are not resident in North America (e.g., Ali et al. 1990; Allen et al. 1980; Axiak et al. 1995b; Batley et al. 1989,1992; Burridge et al. 1995; Camey and Paulini 1964; Danil'chenko 1982; Deschiens and Floch 1968; Deschiens et al. 1964,1966a, 1966b; de Sousa and Paulini 1970; Fent 1991, 1992; Fent and Hunn 1993; Fent and Meier 1992; Frick and DeJimenez 1964; Girard et al. 1996; Helmstetter and Alden 1995; Hopf and Muller 1962; Jantataeme 1991; Karande and Ganti 1994; Karande et al. 1993; Kubo et al. 1984; Langston and Burt 1991; Lewis et al. 1995; Nagabhushanam et al. 1991; Nagase et al. 1991; Nias et al. 1993; Nishuichi and Yoshida 1972; Oehlmann et al. 1996; Reddy et al. 1992; Ringwood 1992; Ritchie et al. 1964; Ruiz et al. 1994a, 1994b, 1995a, 1995b, 1995c, 1997; Sarojini et al. 1991, 1992; Scadding 1990; Scammell et al. 1991; Seiffer and Schoof 1967; Shiff et al. 1975; Shimizu and Kimura 1992; Smith et al. 1979; Spence et al. 1990b; Stebbing et al. 1990; Sujatha et al. 1996; Tsuda et al. 1986, 1991a; Upatham 1975; Upatham et al. 1980a, 1980b; Vitturi et al. 1992; Webbe and Sturrock 1964; Yamada et al. 1994; Yla-Mononen 1989).

Alzieu (1986), Cardarelli and Evans (1980), Cardwell and Sheldon (1986), Cardwell and Vogue (1986), Champ (1986), Chau (1986), Eisler (1989), Envirosphere Company (1986), Evans and Leksono (1995), Gibbs and Bryan (1987), Gibbs et al. (1991a), Good et al. (1980), Guard et al. (1982), Hall (1988, 1991), Hall and Pinkney (1985), Hall et al. (1991), Hodge et al. (1979), International Joint Commission (1976), Jensen (1977), Kimbrough (1976), Kumpulainen and Koivistoinen (1977), Lau (1991), Laughlin (1986), Laughlin and Linden (1985), Laughlin et al. (1984a), McCullough et al. (1980), Monaghan et al. (1980), North Carolina Department of Natural Resources and Community Development (1983,1985), Rexrode (1987), Salazar (1989), Seligman et al. (1986), Slesinger and Dressler (1978), Stebbing (1985), Thayer (1984), Thompson et al. (1985), U.S. EPA (1975,1985b), U.S. Navy (1984), von Rumker et al. (1974), Walsh (1986) and Zuckerman et al. (1978) compiled data from other sources. 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).

Results were not used when the test procedures, test material, or

results were not adequately described (e.g., Bruno and Ellis 1988; Cardwell and Stuart 1988; Chau et al. 1983; Danil'chenko and Buzinova 1982; de la Court 1980; Deschiens 1968; EG&G Bionomics 1981b; Filenko and Isakova 1980; Holwerda and Herwig 1986; Kelly et al. 1990b; Kolosova et al. 1980; Laughlin 1983; Mercier et al. 1994; Nosov and Kolosova 1979; Smith 1981c; Stroganov et al. 1972,1977). 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.

Data were not used when TBT was a component of a formulation, mixture, paint, or sediment (Boike and Rathburn 1973; Cardarelli 1978; Deschiens and Floch 1970; Goss et al. 1979; Henderson and Salazar 1996; Mattiessen and Thain 1989; North Carolina Department of Natural Resources and Community Development 1983; Pope 1981; Quick and Cardarelli 1977; Salazar and Salazar 1985a, 1985b; Santos et al. 1977; Sherman 1983; Sherman and Hoang 1981; Sherman and Jackson 1981; Walker 1977; Weisfeld 1970), unless data were available to show that the toxicity was the same as for TBT alone. 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).

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. 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; Bruschweiler 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). 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 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 naymaycush. 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 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. The species-mean 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. Weight of fathead minnows (P. promelas) 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 180,427 times the water concentration for the soft parts and in rainbow trout, Oncorhyncus mykiss, at 406 times the water concentration for the whole body. Growth of thirteen species of freshwater algae was inhibited by concentrations ranging from 56.1 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 twelve 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 0.0153 µg/L. No Acute-Chronic Ratio is available for this species. 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 factors for three species of bivalve molluscs range from 192.3 for soft parts of the European flat oyster to 11,400 for soft parts of the Pacific oyster, Crassostrea gigas.

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 0.0605µg/L (Table 3). Both of these Chronic Values are below the experimentally determined chronic values from life-cycle or early life-stage tests (0.144 µg/L for D. magna and 0.1308 µg/L for A. sculpta).

Tributyltin chronically affects certain saltwater copepods, gastropods, and pelecypods at concentrations less than those predicted from "standard" acute and chronic toxicity tests. The data show that reductions in growth occur in commercially or ecologically important saltwater species at concentrations of TBT less than the Final Chronic Value of 0.0605 µg/L derived using Final Acute Values and Acute-Chronic Ratios from Table 3. Survival of the copepod A. tonsa was reduced in $\geq$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 was observed in the C. gigas at 0.01 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 0.0605 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 0.001 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 on Table 6, adult dogwhinkle, Nucella lapillus, exposed to 0.05 µg/L TBT for 120 days showed 41% 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% of the female N. Lapillus exposed to TBT at 0.0027 µg/L exhibiting imposex; a followup study of offspring showed almost 99% imposex in females at TBT concentrations of 0.0026 µg/L. Thus, numerous studies show imposex effects at doses well below the calculated Final Chronic Value of 0.0605 µg/L. Many of the studies did not produce a No Observed Adverse 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, Carassius 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 1990 and 1991); the American 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 of from 0.24 to 1.5 µg/L.

The National Research Council (NRC) conducted a four year study to "...review critically the literature on hormonally-active agents in the environment..." and "...identify the known and suspected toxicologic mechanisms and impacts on fish, wildlife and humans...." The report, entitled *Hormonally Active Agents in the Environment* (National Research Council, 1999), cited Bettin et. al. (1996) who reported that TBT "is thought to cause penis growth in female molluscs by affecting steroid metabolism."[p. 102] Immunologic effects have been observed in eastern oysters exposed to 0.03 ug/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. The first line of evidence is the *traditional* endpoints of adverse effects on survival, growth, and reproduction

as demonstrated in numerous laboratory studies, recognizes that a number of these studies have unbounded LOAELs at or near 0.01 µg/L, and recognizes further that only one study included levels below 0.01 µg/L and that study (on Acartia tonsa at 0.003µg/L) showed inhibition of development.

The next three lines of evidence are *additional factors.* These are: 1) the production of imposex in field studies and the impact of imposex on commercially significant species population levels, 2) the accumulation and/or concentration of TBT in commercially and recreationally important freshwater and saltwater species, and 3) 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 only the *traditional* endpoints of adverse effects on survival, growth, and reproduction, and the criteria calculation procedures described in the National Guidelines, the Final Chronic Value would be set at 0.06 µg/L. However, the Agency believes that this level would not be adequately protective because of the *additional factors* cited above. These types of effects are unusual and seem to be characteristic of TBT's ability to produce toxicity through multiple mechanisms.

The Agency is faced with the uncertainty created by the lack of understanding of the relationship of these multiple factors. TBT does not lend itself to the ordinary application of the existing criteria calculation procedures described in the National Guidelines. Therefore, considering the low levels at which adverse effects have been observed, the lack of data showing no effect below these levels, and the importance of the species affected, a lower criterion must be established for TBT.

The National Guidelines require that a criterion be consistent with sound scientific evidence, based on all available pertinent laboratory and field information. The available information for TBT indicates that it causes imposex to occur in saltwater snails at concentrations less than 0.003 ug/L.

Considering that less than 0.003 ug/L is an effect level and the weight of evidence for multiple adverse effects, EPA believes that a Final Chronic Value for TBT in saltwater of 0.001 ug/L is likely to be protective in most situations.

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 0.063 :g/L more than once every three years on the average and if the one-hour average concentration does not exceed 0.46 :g/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 0.001 :g/L more than once every three years on the average and if the one-hour average concentration does not exceed 0.38 :g/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).

Table 1. Acute Toxicity of Tributyltin to Aquatic Animals

Species	Method[a]	Chemical[b]	Hardness (mg/L as $CaCO_3$)	LC50 or EC50 (µg/L)[c]	Species mean Acute value (µg/L)	References
			FRESHWATER SPECIES			
Hydra, *Hydra littoralis*	S,M	TBTO (97.5%)	100	1.11	-	TAI Environmental Sciences, Inc. 1989a
Hydra, *Hydra littoralis*	S,M	TBTO (97.5%)	120	1.30	1.201	TAI Environmental Sciences, Inc. 1989b
Hydra, *Hydra oligactis*	S,M	TBTO (97.5%)	100	1.14	1.14	TAI Environmental Sciences, Inc. 1989a
Hydra, *Chlorohydra viridissmia*	S,M	TBTO (97.5%)	120	1.80	1.80	TAI Environmental Sciences, Inc. 1989b
Annelid (9 mg), *Lumbriculus variegatus*	F,M	TBTO (96%)	51.8	5.4	5.4	Brooke et al. 1986
Freshwater clam, (113 mm TL; 153 g) *Elliptio complanatus*	S,U	TBTO (95%)	-	24,600	24,600	Buccafusco 1976a
Cladoceran, *Daphnia magna*	S,U	TBTO	-	66.3	-	Foster 1981
Cladoceran (adult), *Daphnia magna*	S,U	TBTCl	-	5.26	-	Meador 1986
Cladoceran (<24 hr), *Daphnia magna*	S,U	TBTO (95%)	-	1.58	-	LeBlanc 1976
Cladoceran (<24 hr), *Daphnia magna*	R,M	TBTO (97.5%)	172	11.2	-	ABC Laboratories, Inc. 1990c
Cladoceran (<24 hr), *Daphnia magna*	F,M	TBTO (96%)	51.5	4.3	4.3	Brooke et al. 1986
Cladoceran (<24 hr), *Daphnia magna*	S,U	TBTCl	250	18	-	Crisinel et al. 1994
Amphipod, *Gammarus pseudolimnaeus*	F,M	TBTO (96%)	51.8	3.7	3.7	Brooke et al. 1986

Mosquito (larva), *Culex* sp.	S,M	TBTO (96%)	51.5	10.2	10.2	Brooke et al. 1986

Table 1. (continued)

Species	Method[a]	Chemical[b]	Hardness (mg/L as $CaCO_3$)	LC50 or EC50 (:g/L)[c]	Species Mean Acute Value (:g/L)	References
Rainbow trout, (45 mm TL; 0.68 g) Oncorhynchus mykiss	S,U	TBTO (95%)	-	6.5	-	Buccafusco et al. 1978
Rainbow trout (juvenile), Oncorhynchus mykiss	F,M	TBTO (96%)	50.6	3.9	-	Brooke et al. 1986
Rainbow trout (1.47 g), Oncorhynchus mykiss	F,M	TBTO (97%)	135	3.45	-	Martin et al. 1989
Rainbow trout (1.4 g), Oncorhynchus mykiss	F,M	TBTO (97.5%)	44	7.1	4.571	ABC Laboratories, Inc. 1990a
Lake trout (5.94 g), Salvelinus naymaycush	F,M	TBTO (97%)	135	12.73	12.73	Martin et al. 1989
Fathead minnow (juvenile), Pimephales promelas	F,M	TBTO (96%)	51.5	2.6	2.6	Brooke et al. 1986
Channel catfish, (65 mm TL; 1.9 g) Ictalurus punctatus	S,U	TBTO (95%)	-	11.4	-	Buccafusco 1976a
Channel catfish (juvenile), Ictalurus punctatus	F,M	TBTO (96%)	51.8	5.5	5.5	Brooke et al. 1986
Bluegill, Lepomis macrochirus	S,U	TBTO	-	227.4	-	Foster 1981
Bluegill, (36 mm TL: 0.67 g), Lepomis macrochirus	S,U	TBTO (95%)	-	7.2	-	Buccafusco 1976b
Bluegill (1.01 g), Lepomis macrochirus	F,M	TBTO (97.5%)	44	8.3	8.3	ABC Laboratories, Inc. 1990b

SALTWATER SPECIES

Lugworm (larva), Arenicola cristata	S,U	TBTO	28[c]	-2-4	-	Walsh et al. 1986b
Lugworm (larva), Arenicola cristata	S,U	TBTA	28	-5-10	-5.03	Walsh et al. 1986b

Table 1. (Continued)

Species	Method[a]	Chemical[b]	Salinity (g/kg)	LC50 or EC50 (µg/L)[c]	Species Mean Acute Value (µg/L)	Reference
Polychaete (juvenile), Neanthes arenaceodentata	S,U	TBTO	33-34	6.812	-	Salazar and Salazar 1989
Polychaete (adult), Neanthes arenaceodentata	S,U	TBTO	33-34	21.41[e]	6.812	Salazar and Salazar 1989
Polychaete (adult), Armandia brevis	R,M	TBTCl (96%)	28.5	25	25	Meador 1997
Blue mussel (larva), Mytilus edulis	R,-	TATO	-	2.238	-	Thain 1983
Blue mussel (adult), Mytilus edulis	R,-	TBTO	-	36.98[e]	-	Thain 1983
Blue mussel (adult), Mytilus edulis	S,U	TBTO	33-34	34.06[e]	2.238	Salazar and Salazar 1989
Pacific oyster (larva), Crassostrea gigas	R,-	TBTO	-	1.557	-	Thain 1983
Pacific oyster (adult), Crassostrea gigas	R,-	TBTO	-	282.2[e]	1.557	Thain 1983
Eastern oyster (embryo), Crassostrea virginica	S,U	TBTO	22	0.8759	-	EG&G Bionomics 1976a, 1977
Eastern oyster (embryo), Crassostrea virginica	R,U	TBTCl	18-22	1.30	-	Roberts 1987
Eastern oyster (embryo), Crassostrea virginica	R,U	TBTCl	18-22	0.71	-	Roberts 1987

Eastern oyster, Crassostrea virginica	R,U	TBTCl	18-22	3.96[e]	0.9316	Roberts 1987
European flat oyster (adult), Ostrea edulis	R,-	TBTO	-	204.4	204.4	Thain 1983
Atlantic dogwhinkle (<24 hr-old), Nucella lapillus	R,M	TBTO	34-35	72.7	72.7	Harding et al. 1996

Table 1. (Continued)

Species	Method[a]	Chemical[b]	Salinity (g/kg)	LC50 or EC50 µg/L)[c]	Species Mean Acute Value (µg/L)	References
Hard clam (post larva), Mercenaria mercenaria	S,U	TBTCl	-	0.01466[f]	-	Becerra-Huencho 1984
Hard clam (embryo), Mercenaria mercenaria	R,U	TBTCl	18-22	1.13	-	Roberts 1987
Hard clam (larva), Mercenaria mercenaria	R,U	TBTCl	18-22	1.65	1.365	Roberts 1987
Copepod (juvenile), Eurytemora affinis	F,M	TBTCl	10.6	2.2	-	Hall et al. 1988a
Copepod (subadult), Eurytemora affinis	F,M	TBT	10	2.5	-	Bushong et al. 1987;1988
Copepod (subadult), Eurytemora affinis	F,M	TBT	10	1.4	1.975	Bushong et al. 1987;1988
Copepod (adult), Acartia tonsa	R,U	TBTO (95%)	-	0.6326	-	U'ren 1983
Copepod (subadult), Acartia tonsa	F,M	TBT	10	1.1	1.1	Bushong et al. 1987;1988
Copepod (10-12-d-old), Acartia tonsa	S,U	TBTCl (99.3%)	18	0.47	-	Kusk and Petersen 1997

Copepod (10-12-d-old), Acartia tonsa	S,U	TBTCl (99.3%)	28	0.24	-	Kusk and Petersen 1997
Copepod (adult), Nitocra spinipes	S,U	TBTF	7	1.877	-	Linden et al. 1979
Copepod (adult), Nitocra spinipes	S,U	TBTO	7	1.946	1.911	Linden et al. 1979
Mysid (juvenile), Acanthomysis sculpta	R,M	[g]	-	0.42	-	Davidson et al. 1986a,1986b
Mysid (adult), Acanthomysis sculpta	F,M	[g]	-	1.68[e]	-	Valkirs et al. 1985
Mysid (juvenile), Acanthomysis sculpta	F,M	[g]	-	0.61	0.61	Valkirs et al. 1985
Mysid (juvenile), Metamysidopsis elongata	S,U	TBTO	33-34	<0.9732	-	Salazar and Salazar 1989

Table 1. (Continued)

Species	Method[a]	Chemical[b]	Salinity (g/kg)	LC50 or EC50 (µg/L)[c]	Species Mean Acute Value (µg/L)	Reference
Mysid (subadult), Metamysidopsis elongata	S,U	TBTO	33-34	1.946[e]	-	Salazar and Salazar 1989
Mysid (adult), Metamysidopsis elongata	S,U	TBTO	33-34	2.433[e]	-	Salazar and Salazar 1989
Mysid (adult), Metamysidopsis elongata	S,U	TBTO	33-34	6.812[e]	<0.9732	Salazar and Salazar 1989
Mysid (<1 day), Mysidopsis bahia	F,M	TBTCl	19-22	1.1	-	Goodman et al. 1988
Mysid (5 day), Mysidopsis bahia	F,M	TBTCl	19-22	2.0	-	Goodman et al. 1988
Mysid (10 day), Mysidopsis bahia	F,M	TBTCl	19-22	2.2	1.692	Goodman et al. 1988
Amphipod (subadult), Gammarus sp.	F,M	TBT	10	1.3	-	Bushong et al. 1988
Amphipod (adult), Gammarus sp.	F,M	TBT	10	5.3[e]	1.3	Bushong et al. 1988

Amphipod (adult), Orchestia traskiana	R,M	TBTO	30	>14.60[h]	>14.60	Laughlin et al. 1982
Amphipod (adults), Rhepoxynius abronius	R,M	TBTCl (96%)	32.3	108	108	Meador 1997
Amphipod (3-5 mm; 2-5 mg), Eohaustorius estuarius	R,M	TBTCl (96%)	28.8-29.5	10	10	Meador 1993; Meador et al. 1993; Meador 1997
Amphipod (adult), Eohaustorius washingtonianus	R,M	TBTCl (96%)	32.7	9	9	Meador 1997
Grass shrimp (adult), Palaemonetes pugio	F,U	TBTO	-	20	20	Clark et al. 1987
Grass shrimp (subadult), Palaemonetes sp.	F,M	TBT	10	>31	>31	Bushong et al. 1988
Grass shrimp (larvae), Palaemonetes sp.	R,U	TBTO	20	4.07	-	Kahn et al. 1993
Grass shrimp (adult), Palaemonetes sp.	R,U	TBTO	20	31.41[e]	4.07	Kahn et al. 1993

Table 1. (Continued)

Species	Method[a]	Chemical[b]	Salinity (g/kg)	LC50 or EC50 (µg/L)[c]	Species Mean Acute Value (µg/L	Reference
American lobster (larva), Homarus americanus	R,U	TBTO	32	1.745[h]	1.745	Laughlin and French 1980
Shore crab (larva), Carcinus maenas	R,-	TBTO	-	9.732	9.732	Thain 1983
Mud crab (larva), Rhithropanopeus harrisii	R,U	TBTS	15	>24.3[h]	-	Laughlin et al. 1983
Mud crab (larva), Rhithropanopeus harrisii	R,U	TBTO	15	34.90[h]	34.90	Laughlin et al. 1983
Shore crab (larva), Hemigrapsus nudus	R,U	TBTO	32	83.28[h]	83.28	Laughlin and French 1980
Amphioxus, Branchiostoma caribaeum	F,U	TBTO	-	<10	<10	Clark et al. 1987

Chinook salmon (juvenile), Oncorhynchus tshawytscha	S,M	TBTO	28	1.460	1.460	Short and Thrower 1986b;1987
Atlantic menhaden (juvenile), Brevoortia tyrannus	F,M	TBT	10	4.7	-	Bushong et al. 1987;1988
Atlantic menhaden (juvenile), Brevoortia tyrannus	F,M	TBT	10	5.2	4.944	Bushong et al. 1987;1988
Sheepshead minnow (juvenile), Cyprinodon variegatus	S,U	TBTO	20	16.54	-	EG&G Bionomics 1979
Sheepshead minnow (juvenile), Cyprinodon variegatus	S,U	TBTO	20	16.54	-	EG&G Bionomics 1979
Sheepshead minnow (juvenile), Cyprinodon variegatus	S,U	TBTO	20	12.65	-	EG&G Bionomics 1979
Sheepshead minnow (33-49 mm), Cyprinodon variegatus	F,M	TBTO	28-32	2.315[h]	-	EG&G Bionomics 1981d

Table 1. (Continued)

Species	Method[a]	Chemical[b]	Salinity (g/kg)	LC50 or EC50 (µg/L)[c]	Species Mean Acute Value (µg/L)	Reference
Sheepshead minnow (juvenile), Cyprinodon variegatus	F,M	TBTO	15	12.31	-	Walker 1989a
Sheepshead minnow (subadult), Cyprinodon variegatus	F,M	TBT	10	25.9	9.037	Bushong et al. 1988
Mummichog (adult), Fundulus heteroclitus	S,U	TBTO (95%)	25	23.36	-	EG&G Bionomics 1976a
Mumichog (juvenile), Fundulus heteroclitus	F,M	TBTO	2	17.2	-	Pinkney et al. 1989

Mummichog (larval), Fundulus heteroclitus	F,M	TBT	10	23.4	-	Bushong et al. 1988
Mummichog (subadult), Fundulus heteroclitus	F,M	TBT	10	23.8	21.34	Bushong et al. 1988
Inland silverside (larva), Menidia beryllina	F,M	TBT	10	3.0	3.0	Bushong et al. 1987;1988
Atlantic silverside, Menidia menidia	F,M	TBT	10	8.9	8.9	Bushong et al. 1987;1988
Starry flounder (<1-year-old), Platichthys stellatus	R,M	TBTCl (96%)	30.2	10.1	10.1	Meador 1997

[a] S = static; R = renewal; F = flow-through; M = measured; U = unmeasured.
[b] TBTCl = tributyltin chloride; TBTF = tributyltin fluoride; TBTO = tributyltin oxide; TBTS = tributyltin sulfide. Percent purity is given in parentheses when available.
[c] Salinity (g/kg).
[d] 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%.
[e] Value not used in determination of Species Mean Acute Value because data are available for a more sensitive life stage.
[f] Value not used in determination of Species Mean Acute Value (see text).
[g] 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.
[h] LC50 and EC50 calculated or interpolated graphically based on the authors' data.

Table 2. Chronic Toxicity of Tributyltin to Aquatic Animals.

Species	Test[a]	Chemical[b]	Hardness (mg/L as $CaCO_3$)	Chronic Limits (µg/L)[c]	Chronic Value (µg/L)	References
			FRESHWATER SPECIES			
Cladoceran, *Daphnia magna*	LC	TBTO (96%)	51.5	0.1-0.2	0.1414	Brooke et al. 1986
Cladoceran, *Daphnia magna*	LC	TBTO (100%)	160-174	0.19-0.34	0.2542	ABC Laboratories, Inc. 1990d
Fathead minnow, *Pimephales promelas*	ELS	TBTO (96%)	51.5	0.15-0.45	0.2598	Brooke et al. 1986
			SALTWATER SPECIES			
Atlantic dogwhinkle, *Nucella lapillus*	ELS	TBTO (97%)	34-35	0.0077-0.0334[f]	0.0153	Harding et al. 1996
Copepod, *Eurytemora affinis*	LC	TBTCl	10.3[e]	<0.088	<0.088	Hall et al. 1987;1988a
Copepod, *Eurytemora affinis*	LC	TBTCl	14.6[e]	0.094-0.224	0.145	Hall et al. 1987;1988a
Mysid, *Acanthomysis sculpta*	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 2. (continued)

Acute-Chronic Ratios

Species	Hardness (mg/L as $CaCO_3$)	Acute Value (µg/L)	Chronic Value (µg/L)	Ratio
Cladoceran, *Daphnia magna*	51.5	4.3	0.1414	30.41
Cladoceran, *Daphnia magna*	160-174	11.2	0.2542	44.06
Fathead minnow, *Pimephales promelas*	51.5	2.6	0.2598	10.01
Copepod, *Eurytemora affinis*	-	2.2	<0.088	>25.00
Copepod, *Eurytemora affinis*	-	2.2	0.145	15.17
Mysid, *Acanthomysis sculpta*	-	0.61[a]	0.1308	4.664
Snail, *Nucella lapillus*	34-35[b]	72.7	0.0153	4752

[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 (µg/L)	Species	Species Mean Acute Value (µg/L)[b]	Species Mean Acute-Chronic Ratio[c]
		FRESHWATER SPECIES		
12	24,600	Freshwater clam, *Elliptio camplanatus*	24,600	-
11	12.73	Lake trout, *Salvelinus naymaycush*	12.73	-
10	10.2	Mosquito, *Culex* sp.	10.2	-
9	8.3	Bluegill, *Lepomis macrochirus*	8.3	-
8	5.5	Channel catfish, *Ictalurus punctatus*	5.5	-
7	5.4	Annelid, *Lumbriculus variegatus*	5.4	-
6	4.571	Rainbow trout, *Oncorhyncus mykiss*	4.571	-
5	4.3	Cladoceran, *Daphnia magna*	4.3	36.60
4	3.7	Amphipod, *Gammarus pseudolimnaeus*	3.7	-
3	2.6	Fathead minnow, *Pimephales promelas*	2.6	10.01
2	1.80	Hydra, *Chlorohydra viridissmia*	1.80	-
1	1.170	Hydra, *Hydra littoralis*	1.201	-
		Hydra, *Hydra oligactis*	1.14	-

Table 3. (Continued)

Rank[a]	Genus Mean Acute Value (µg/L)	Species	Species Mean Acute Value (µg/L)[b]	Species Mean Acute-Chronic Ratio[c]
		SALTWATER SPECIES		
30	204.4	European flat oyster, *Ostrea edulis*	204.4	-
29	108	Amphipod, *Rhepoxynius abronius*	108	-
28	83.28	Shore crab, *Hemigrapsus nudus*	83.28	-
27	72.7	Atlantic dogwhinkle, *Nucella lapillus*	72.7	4752
26	34.90	Mud crab, *Rhithropanopeus harrisii*	34.90	-
25	25	Polychaete, *Armandia brevis*	25	-
24	24.90	Grass shrimp, *Palaemonetes pugio*	20	-
		Grass shrimp, *Palaemonetes* sp.	>31	-
23	21.34	Mummichog, *Fundulus heteroclitus*	21.34	-
22	>14.60	Amphipod, *Orchestia traskiana*	>14.60	-
21	10.1	Starry flounder, *Platichthys stellatus*	10.1	-
20	<10	Amphioxus *Branchiostoma caribaeum*	<10	-
19	9.732	Shore crab, *Carcinus maenas*	9.732	-
18	9.534	Amphipod, *Eohaustorius estuarius*	10.1	-

		Amphipod, *Eohaustorius washingtonianus*	9	-

Table 3.
(Continued)

Rank[a]	Genus Mean Acute Value (µg/L)	Species	Species Mean Acute Value (µg/L)[b]	Species Mean Acute-Chronic Ratio[c]
17	9.037	Sheepshead minnow, *Cyprinodon variegatus*	9.037	-
16	6.812	Polychaete, *Neanthes arenacedentata*	6.812	-
15	5.167	Inland silverside, *Menidia beryllina*	3.0	-
		Atlantic silverside, *Menidia menidia*	8.9	-
14	-5.0	Lugworm, *Arenicola cristata*	-5.0	-
13	4.944	Atlantic manhaden, *Brevoortia tyrannus*	4.944	-
12	2.238	Blue mussel, *Mytilus edulis*	2.238	-
11	1.975	Copepod, *Eurytemora affinis*	1.975	15.17
10	1.911	Copepod, *Nitocra spinipes*	1.911	-
9	1.745	American lobster, *Homarus americanus*	1.745	-
8	1.692	Mysid, *Mysidopsis bahia*	1.692	-
7	1.460	Chinook salmon, *Oncorhynchus tshawytscha*	1.460	-
6	1.365	Hard clam, *Mercenaria mercenaria*	1.365	-
5	1.3	Amphipod, *Gammarus* sp.	1.3	-

4	1.204	Pacific oyster, Crassostrea gigas	1.557	-
		Eastern oyster, Crassostrea virginica	0.9316	-
3	1.1	Copepod, Acartia tonsa	1.1	-

Table 3.
(Continued)

Rank[a]	Genus Mean Acute Value (µg/L)	Species	Species Mean Acute Value (µg/L)[b]	Species Mean Acute-Chronic Ratio[c]
2	<0.9732	Mysid, Metamysidopsis elongata	<0.9732[d]	-
1	0.61	Mysid, Acanthomysis sculpta	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.
[d] This was used as a quantitative value, not as a "less than" value in the calculation of the Final Acute Value.

Fresh Water

Final Acute Value = 0.9177 ˙g/L

Criterion Maximum Concentration = (0.9177 ˙g/L)/2 = 0.4589 ˙g/L

Final Acute-Chronic Ratio = 12.69 (see text)

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

Salt Water

Final Acute Value = 0.7673 µg/L

Criterion Maximum Concentration = (0.7673 µg/L)/2 = 0.3836 µg/L

Final Acute-Chronic Ratio = 12.69 (see text)

Final Chronic Value = (0.7673 µg/L)/12.69 = 0.0605 µg/L

Final Chronic Value = 0.010 µg/L (lowered to protect growth of commercially important molluscs and survival of the ecologically important copepod Acartia tonsa; see text)

Table 4. Toxicity of Tributyltin to Aquatic Plants

Species	Chemical[a]	Hardness (mg/L as $CaCO_3$)	Duration (days)	Effect	Concentration (µg/L)[b]	Reference
			FRESHWATER SPECIES			
Alga, Bumilleriopsis filiformis	TBTCl	-	14	No growth	111.4	Blanck 1986; Blanck et al. 1984
Alga, Klebsormidium marinum	TBTCl	-	14	No growth	222.8	Blanck 1986; Blanck et al. 1984
Alga, Monodus subterraneus	TBTCl	-	14	No growth	1,782.2	Blanck 1986; Blanck et al. 1984
Alga, Raphidonema longiseta	TBTCl	-	14	No growth	56.1	Blanck 1986; Blanck et al. 1984
Alga, Tribonema aequale	TBTCl	-	14	No growth	111.4	Blanck 1986; Blanck et al. 1984
Blue-green alga, Oscillatoria sp.	TBTCl	-	14	No growth	222.8	Blanck 1986; Blanck et al. 1984
Blue-green alga, Synechococcus leopoliensis	TBTCl	-	14	No growth	111.4	Blanck 1986; Blanck et al. 1984
Green alga, Chlamydomonas dysosmas	TBTCl	-	14	No growth	111.4	Blanck 1986; Blanck et al. 1984
Green alga, Chlorella emersonii	TBTCl	-	14	No growth	445.5	Blanck 1986; Blanck et al. 1984

Green alga, Kirchneriella contorta	TBTCl	-	14	No growth	111.4	Blanck 1986; Blanck et al. 1984
Green alga, Monoraphidium pusillum	TBTCl	-	14	No growth	111.4	Blanck 1986; Blanck et al. 1984
Green alga, Scenedesmus obtusiusculus	TBTCl	-	14	No growth	445.5	Blanck 1986; Blanck et al. 1984
Green alga, Scenedesmus quadricauda	TBTCl	-	12	Reduced growth (87.6%)	1	Fargasova and Kizlink 1996
Green alga, Scenedesmus quadricauda	TBTO	-	7	EC50 chlorophyll production	5.0	Fargasova 1996

Table 4. (Continued)

Species	Chemical[a]	Hardness (mg/L as $CaCO_3$)	Duration (days)	Effect	Concentration (µg/L)[b]	Reference
Green alga, Scenedesmus quadricauda	TBTO	0.67	12	Reduced growth 87.6% 95.9% 100%	1 10 100	Fargasova and Kizlink 1996
Green alga, Scenedesmus obliquus	TBT	72.7	4	EC50 (reduced growth)	3.4	Huang et al. 1993
Green alga, Selenastrum capricornutum	TBTCl	-	14	No growth	111.4	Blanck 1986; Blanck et al. 1984
Green alga, Selenastrum capricornutum	TBTCl	-	4	EC50	12.4	Miana et al. 1993

SALTWATER SPECIES

Diatom, Skeletonema costatum	TBTO	-	5	Algistatic Algicidal	0.9732-17.52 >17.52	Thain 1983
Diatom, Skeletonema costatum	TBTO (BioMet Red)	30[c]	14	EC50 (dry cell weight)	>0.1216; <0.2433	EG&G Bionomics 1981c
Diatom, Skeletonema costatum	TBTO	30[c]	14	EC50 (dry cell weight)	0.06228	EG&G Bionomics 1981c
Diatom, Nitzschia sp.	TBTO	-	8	EC50 (reduced growth)	1.19	Delupis et al. 1987
Flagellate alga, Dunaliella tertiolecta	TBTO	-	8	EC50 (reduced growth)	4.53	Delupis et al. 1987
Mixed algae, Dunaliella salina and D. viridis	TBT	-	4	EC50 (reduced growth)	0.68	Huang et al. 1993

[a] TBTCl = 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

Species	Chemical[a]	Hardness (mg/L as $CaCO_3$)	Concentration in Water (µg/L)[b]	Duration (days)	Tissue	BCF or BAF[c]	Reference
			FRESHWATER SPECIES				
Zebra mussel (1.76±0.094 cm), Dreissena polymorpha	TBT	-	0.0703	105	Soft parts	17,483[d]	Becker-van Slooten and Tarradellas 1994
Rainbow trout (13.8 g), Oncorhynchus mykiss	TBTO (97%)	135	0.513	64	Whole body	406	Martin et al. 1989
Rainbow trout (32.7 g), Oncorhynchus mykiss	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); Cyprinus carpio	TBTO	-	2.1	14	Muscle	501.2	Tsuda et al. 1988a
Carp (8.5-9.5 cm; 16.5-22.1 g); Cyprinus carpio	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); Carassius auratus	TBTCl	36	0.13	28	Whole body	1,976	Tsuda et al. 1991b
Guppy (2.4-2.7 cm; 0.41-0.55 g); Poecilia reticulatus	TBTO (95%)	-	0.54	14	Whole body	460	Tsuda et al. 1990b

Table 5.
(Continued)

Species	Chemical[a]	Salinity (g/kg)	Concentration in Water (µg/L)[b]	Duration (days)	Tissue	BCF or BAF[c]	Reference
			SALTWATER SPECIES				
Snail (adults), Littorina littorina	TBTCl	-	0.488 0.976	182 182	Soft parts Soft parts	1,420 1,020	Bauer et al. 1997
Atlantic dogwhinkle (female), Nucella lapillus	TBT	-	0.0038 to 0.268	249 to 408	Soft parts	11,000 to 38,000	Bryan et al. 1987a
Atlantic dogwhinkle (female), Nucella lapillus	Field	-	0.070	529 to 634	Soft parts	17,000	Bryan et al. 1987a
Atlantic dog whinkle (18-22 mm), Nucella lapillus	TBTCl	35	0.0205	49	Soft parts	30,000	Bryan et al. 1989b
Atlantic dogwhinkle (1 year-old), Nucella lapillus	TBTO	34-35	0.0027 0.0077 0.0334 0.1246	365 365 365 365	Soft parts Soft parts Soft parts Soft parts	18,727 21,964 16,756 7,625	Bailey et al. 1991
Atlantic dogwhinkle (1 year-old), Nucella lapillus	TBTO	34-35	0.0026 0.0074 0.0278 0.1077	365 365 365 365	Soft parts Soft parts Soft parts Soft parts	<7782 10,121 8,088 6,172	Harding et al. 1996

Blue mussel (spat), Mytilus edulis	[e]	28.5-34.2	0.24	45	Soft parts	6,833[f]	Thain and Waldock 1985; Thain 1986
Blue mussel (adult), Mytilus edulis	Field	-	<0.1	60	-	11,000	Salazar and Salazar 1990a
Blue mussel (juvenile), Mytilus edulis	Field	-	<0.1	60	-	25,000	Salazar and Salazar 1990a
Blue mussel, Mytilus edulis	[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), Mytilus edulis	Field	-	<0.105	84	Soft parts	5,000-60,000	Salazar and Salazar, 1991

Table 5.
(Continued)

Species	Chemical[a]	Salinity (g/kg)	Concentration in Water (µg/L)[b]	Duration (days)	Tissue	BCF or BAF[c]	Reference
Blue mussel (3.0 - 3.5 cm), Mytilus edulis	TBTCl	25.1-26.3	0.020	60	Muscle and mantle Muscle and mantle	7,700 11,000	Guolan and Yong 1995
Pacific oyster, Crassostrea gigas	TBTO	28-31.5	1.216	21	Soft parts	1,874[f]	Waldock et al. 1983
American Oyster (6-9 cm length), Crassostrea virginica	-	18±1	0.283	28	Soft parts	15,460	Roberts et al. 1996
Pacific oyster, Crassostrea gigas	TBTO	28-31.5	0.1460	21	Soft parts	6,047[f]	Waldock et al. 1983

Pacific oyster, Crassostrea gigas	[e]	28.5-34.2	0.24	45	Soft parts	7,292[f]	Thain and Waldock 1985; Thain 1986
Pacific oyster, Crassostrea gigas	TBTO	29-32	1.557	56	Soft parts	2,300	Waldock and Thain 1983
Pacific oyster, Crassostrea gigas	TBTO	29-32	0.1460	56	Soft parts	11,400	Waldock and Thain 1983
Pacific oyster (spat), Crassostrea gigas	TBTO	-	0.29 0.92 2.83	30	Soft parts	2275 1369 621	Osada et al. 1993
European flat oyster, Ostrea edulis	TBTO	28-31.5	1.216	21	Soft parts	960[f]	Waldock et al. 1983
European flat oyster, Ostrea edulis	TBTO	28-34.2	0.24	75	Soft parts	875[f]	Waldock et al. 1983
European flat oyster, Ostrea edulis	TBTO	28-34.2	2.62	75	Soft parts	397[f]	Thain 1986
European flat oyster, Ostrea edulis	[e]	28.5-34.2	0.24	45	Soft parts	1,167[f]	Thain and Waldock 1985; Thain 1986
European flat oyster, Ostrea edulis	[d]	28.5-34.2	2.62	45	Soft parts	192.3[f]	Thain and Waldock 1985; Thain 1986
Guppy (&; 2.4-2.7 cm; 0.41-0.55 g); Poecilia reticulatus	TBTC (95%)	-	0.28	14	Whole body	240	Tsuda et al. 1990b

[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

Species	Chemical[a]	Hardness (mg/L as $CaCO_3$)	Duration	Effect	Concentration (µg/L)[b]	Reference
				FRESHWATER SPECIES		
Microcosm natural assemblage	TBTO	-	55 days	Daphnia magna 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) Metabolism returned to normal (14.1 days)	4.7	Miniero and Delupis 1991
				Metabolism reduced (1 day) Metabolism returned to normal (16 days)	14.9	
Alga, Natural assemblage	-	-	4 hr	EC50 (production)	5	Wong et al. 1982
Blue-green alga, Anabaena flos-aquae	-	-	4 hr	EC50 (production)	13	Wong et al. 1982
Green alga, Ankistrodesmus falcatus	-	-	4 hr	EC50 (production) (reproduction)	 20 5	Wong et al. 1982
Green alga, Ankistrodesmus falcatus	TBTO (97%)	-	7 days 14 days 21 days 28 days	BCF = 300 BCF = 253 BCF = 448 BCF = 467	5.2 4.7 2.1 1.5	Maguire et al. 1984
Green alga, Scenedesmus quadricauda	-	-	4 hr	EC50 (production)	16	Wong et al. 1982

Hydra, Hydra sp.	TBTO (96%)	51.0	96 hr	EC50 (clubbed tentacles)	0.5	Brooke et al. 1986
Rotifer, Brachionus calyciflorus	TBTCl	-	24 hr	EC50 (hatching)	72	Crisinel et al. 1994
Asiatic clam (larva), Corbicula fluminea	TBTO	-	24 hr	EC50	1,990	Foster 1981

Table 6. (Continued)

Species	Chemical[a]	Hardness (mg/L as $CaCO_3$)	Duration	Effect	Concentration (µg/L)[b]	Reference
Cladoceran, Daphnia magna	TBTO	-	24 hr	LC50	3	Polster and Halacha 1972
Cladoceran (<24 hr), Daphnia magna	TBTC	200	24 hr	EC50 (mobility)	11.6	Vighi and Calamari 1985
Cladoceran (<24 hr), Daphnia magna	TBTO	200	24 hr	EC50 (mobility)	13.6	Vighi and Calamari 1985
Cladoceran (adult), Daphnia magna	TBTCl	-	8 days	Altered phototaxis	0.45	Meador 1986
Cladoceran (14-d-old), Daphnia magna	TBTCl	150	7 days	Altered behavior Reproductive failure Digestive storage cells reduced	1 1 5	Bodar et al. 1990
Cladoceran (<24-h old), Daphnia magna	TBTCl	312.8	48 hr	EC50 (mobility)	9.8	Miana et al. 1993
Fairy shrimp (cysts), Streptocephalus texanus	TBTCl	250	24 hr	EC50 (hatching)	15	Crisinel et al. 1994
Rainbow trout (yearling), Oncorhynchus mykiss	TBTO	-	24 hr 48 hr	LC50	25.2 18.9	Alabaster 1969
Rainbow trout, Oncorhynchus mykiss	TBTO	-	24 hr	EC50 (rheotaxis)	30.8	Chliamovitch and Kuhn 1977

Rainbow trout (embryo, larva), Oncorhynchus mykiss	TBTCl	94-102	110 days	20% reduction in growth	0.18	Seinen et al. 1981
				23% reduction in growth; 6.6% mortality	0.89	
				100% mortality	4.46	
Rainbow trout (fry), Oncorhynchus mykiss	TBTCl	96-105	110 days	NOEC (mortality and growth)	0.040	de Vries et al. 1991
				LOEC (mortality and growth)	0.200	
Rainbow trout, Oncorhynchus mykiss	TBTO	-	28 days	BCF = 3833 (whole body)	0.6	Schwaiger et al. 1992
				BCF = 2850 (whole body)	1.0	
				BCF = 2700 (whole body)	2.0	
				BCF = 1850 (whole body)	4.0	
				Cell necrosis within gill lamellae	4.0	

Table 6. (Continued)

Species	Chemical[a]	Hardness (mg/L as $CaCO_3$)	Duration	Effect	Concentration (µg/L)[b]	Reference
Rainbow trout, Oncorhynchus mykiss	TBTO	-	28 days	BCF = 3833 (whole body)	0.6	Schwaiger et al. 1992
				BCF = 2850 (whole body)	1.0	
				BCF = 2700 (whole body)	2.0	
				BCF = 1850 (whole body)	4.0	
				Cell necrosis within gill lamellae	4.0	
Rainbow trout (3 week), Oncorhynchus mykiss	TBTO (98%)	400	21 days	Reduced growth	0.5	Triebskorn et al. 1994
				Reduced avoidance		
				BCF = 540 (no head; no plateau)		
				BCF = 990 (no head; no plateau	2	
Goldfish (2.8-3.5 cm; 0.9-1.7 g), Carassius auratus	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), Cyprinus carpio	TBTO	-	7 days	BCF in muscle = 295 Half-life = 1.67 days	1.80	Tsuda et al. 1987
Guppy (3-4 wk), Poecilia reticulata	TBTO	209	3 mo	Thymus atrophy	0.32	Wester and Canton 1987
				Hyperplasia of kidney hemopoietic tissue	1.0	
				Marked liver vacuolation	1.0	
				Hyperplasia of corneal epithelium	10.0	
Guppy (4 wk), Poecilia reticulata	TBTO	-	1 mo 3 mo	NOEC NOEC	1.0 0.32	Wester and Canton 1991
Frog (embryo, larva), Rana temporaria	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. (Continued)

Species	Chemical[a]	Salinity (g/kg)	Duration	Effect	Concentration (µg/L)[b]	Reference
			SALTWATER SPECIES			
Natural microbial populations	TBTCl	2 and 17	1 hr (incubated 10 days)	Significant decrease in metabolism of nutrient substrates	4.454	Jonas et al. 1984
Natural microbial populations	TBTCl	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)	TBTCl	-	6 wks	Reduced plant material loss; loss of amphipod Cymadusa compta	22.21	Kelly et al. 1990a
Periphyton communities	TBTCl	-	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, Dunaliella tertiolecta	TBTO	34-40	18 days	Population growth	1.0	Beaumont and Newman 1986
Green alga, Dunaliella sp.	TBTO	-	72 hr	Approx. EC50 (growth)	1.460	Salazar 1985
Green alga, Dunaliella sp.	TBTO	-	72 hr	100% mortality	2.920	Salazar 1985
Green alga, Dunaliella tertolecta	TBTO	-	8 days	EC50	4.53	Delupis et al. 1987

Table 6. (Continued)

Species	Chemical[a]	Salinity (g/kg)	Duration	Effect	Concentration (µg/L)[b]	Reference
Diatom, Phaeodoctylum tricornutum	TBTO	-	72 hr	No effect on growth	1.460-5.839	Salazar 1985
Diatom, Nitzschia sp.	TBTO	-	8 days	EC50	1.19	Delupis et al. 1987

Diatom, Nitzschia sp.	TBTO	-	8 days	EC50	1.19	Delupis et al. 1987
Diatom, Nitzschia closterium	TBTCl	-	7 days	EC50 (growth)	1.16	Nakagawa and Saeki 1992
Diatom, Skeletonema costatum	TBTA	30	72 hr	EC50 (population growth)	0.3097	Walsh et al. 1985;1987
Diatom, Skeletonema costatum	TBTA	30	72 hr	LC50	12.65	Walsh et al. 1985;1987
Diatom, Skeletonema costatum	TBTO	34-40	12-18 days	Population growth	1.0	Beaumont and Newman 1986
Diatom, Skeletonema costatum	TBTO	30	72 hr	EC50 (population growth)	0.3212	Walsh et al. 1985;1987
Diatom, Skeletonema costatum	TBTO	30	72 hr	LC50	13.82	Walsh et al. 1985
Diatom, Skeletonema costatum	TBTCl	30	72 hr	EC50 (population growth)	0.3207	Walsh et al. 1985;1987
Diatom, Skeletonema costatum	TBTCl	30	72 hr	LC50	10.24	Walsh et al. 1985;1987
Diatom, Skeletonema costatum	TBTF	30	72 hr	EC50 (population growth)	>0.2346, >0.4693	Walsh et al. 1985;1987
Diatom, Skeletonema costatum	TBTF	30	72 hr	LC50	11.17	Walsh et al. 1985
Diatom, Skeletonema costatum	TBTCl	30.5	96 hr	NOEC	1	Reader and Pelletier 1992
Diatom, Skeletonema costatum	TBTCl	-	7 days	EC50 (growth)	3.48	Nakagawa and Saeki 1992

Table 6. (Continued)

Species	Chemical[a]	Salinity (g/kg)	Duration	Effect	Concentration (µg/L)[b]	Reference

Diatom, Chaetoceros debilis	TBTCl	-	7 days	EC50 (growth)	1.16	Nakagawa and Saeki 1992
Diatom, Chattonella antiqua	TBTCl	-	7 days	EC50 (growth)	2.05	Nakagawa and Saeki 1992
Diatom, Tetraselmis tetrathele	TBTCl	-	7 days	EC50 (growth)	6.06	Nakagawa and Saeki 1992
Diatom, Minutocellus polymorphus	TBTO	-	48 hr	EC50	-340	Walsh et al. 1988
Diatom, Minutocellus polymorphus	TCTCl	-	48 hr	EC50	-330	Walsh et al. 1988
Diatom, Thalassiosira pseudonana	TBTA	30	72 hr	EC50 (population growth)	1.101	Walsh et al. 1985
Diatom, Thalassiosira pseudonana	TBTO	30	72 hr	EC50 (population growth)	1.002	Walsh et al. 1985;1987
Alga, Pavlova lutheri	TBTO	34-40	12-26 days	Population growth	1.0	Beaumont and Newman 1986
Alga, Pavlova lutheri	TBTO	-	16 days	NOEC LOEC	5.36 21.46	Saint-Louis et al. 1994
Dinoflagellate, Gymnodinium splendens	TBTO	-	72 hr	100% mortality	1.460	Salazar 1985
Macroalgae, Fucus vesiculosus	TBT	6	7 days	Photosynthesis and nutrient uptake reduced	0.6	Lindblad et al. 1989
Giant kelp (zoospores), Macrocystis pyrifera	TBT	32-33	48 hr	EC50 (germination) EC50 (growth)	11.256 13.629	Brix et al. 1994a
Polychaete worm (juvenile), Neanthes arenaceodentata	TBTCl (96%)	30	10 wks	NOEC (survival) LOEC (survival)	0.100 0.500	Moore et al. 1991
Polychaete worm (adult), Armandia brevis	TBTCl (96%)	28.5	10 days	BCF = 5,100 (no plateau)	233	Meador 1997

Rotifer (neonates), Brachionus plicatilis	TBT	15	30 min	Induction of the stress protein gene SP58	20-30	Cochrane et al. 1991

Table 6. (Continued)

Species	Chemical[a]	Salinity (g/kg)	Duration	Effect	Concentration (µg/L)[b]	Reference
Hydroid, Campanularia flexuosa	TBTF	35	11 days	Colony growth stimulation; no growth	0.01 1.0	Stebbing 1981
Pale sea anemone (1-2 cm oral disc), Aiptasia pallida	TBT	-	28 days	Reduced (90.4%) symbiotic zooxanthellae populations; incresed bacterial aggregates; fewer undischarged nematocysts	0.05	Mercier et al. 1997
Sand dollar (sperm), Dendraster excentricus	TBT	32-33	80 min	EC50 (mortality)	0.465	Brix et al. 1994b
Starfish (79 g), Leptasterias polaris	TBTCl	25.9	48 hr	BCF = 41,374 (whole body)	0.072	Rouleau et al. 1995
Dogwhinkle (adult), Nucella lapillus	[c]	-	120 days	41% Imposex (superimposition of male anatomical characteristics on females)	0.05	Bryan et al. 1986
Dogwhinkle (adult), Nucella lapillus	TBTCl	35	6 months	Imposex induced	$0.012	Stroben et al. 1992b
Dogwhinkle (subadult), Nucella lapillus	TBTCl	35	22	BCF = -20,000	0.019	Bryan et al. 1993
Mussel (juvenile), Mytilus sp.	Field	-	12 weeks	NOEC tissue conc. growth = 0.5 µg/g LOEC tissue conc. 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), Mytilus edulis	TBTO	-	24 hr	No effect on sister chromatid exchange	1.0	Dixon and Prosser 1986
Blue mussel (larva), Mytilus edulis	TBTO	-	4 days	Reduced survival	≥0.1	Dixon and Prosser 1986
Blue mussel (spat), Mytilus edulis	[c]	28.5-34.2	45 days	100% mortality	2.6	Thain and Waldock 1985; Thain 1986
Blue mussel (larva), Mytilus edulis	TBTO	33	15 days	51% mortality; reduced growth	0.0973	Beaumont and Budd 1984

Table 6. (Continued)

Species	Chemical[a]	Salinity (g/kg)	Duration	Effect	Concentration (µg/L)[b]	Reference
Blue mussel (larva), Mytilus edulis	[c]	-	45 days	Reduced growth	0.24	Thain and Waldock 1986
Blue mussel (juvenile), Mytilus edulis	TBTO	33.7	7 days	Significant reduction in growth	0.3893	Stromgren and Bongard 1987
Blue mussel (juvenile), Mytilus edulis	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), Mytilus edulis	TBT (field)	-	12 wks	Reduced growth	≥0.2	Salazar and Salazar 1988
Blue mussel (juvenile), Mytilus edulis	TBT (field)	-	12 wks	Reduced growth at tissue conc. of 2.0 µg/g	-	Salazar and Salazar 1988
Blue mussel (juvenile), Mytilus edulis	[c]	-	56 days	Reduced condition	0.157	Salazar et al. 1987
Blue mussel (juvenile), Mytilus edulis	[c]	-	196 days	Reduced growth (no effect at day 56 of 0.2 µg/L)	0.070	Salazar and Salazar 1987
Blue mussel (juvenile), Mytilus edulis	[c]	-	56 days	No effect on growth	0.160	Salazar and Salazar 1987

Blue mussel (2.5 to 4.1 cm), Mytilus edulis	[c]	-	66 days	LC50	0.97	Valkirs et al. 1985;1987
Blue mussel (2.5 to 4.1 cm), Mytilus edulis	[c]	-	66 days	Significant decrease in shell growth	0.31	Valkirs et al. 1985
Blue mussel (juveniles and adults), Mytilus sp.	TBT (field)	-	84 days	BCF	3,000-100,000	Salazar 1996
Blue mussel (3.0-3.5 cm), Mytilus edulis	TBT	-	2 days	Reduced ability to survive anoxia	1	Wang et al. 1992

Table 6. (Continued)

Species	Chemical[a]	Salinity (g/kg)	Duration	Effect	Concentration (µg/L)[b]	Reference
Blue mussel (4 cm), Mytilus edulis	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), Mytilus edulis	TBT	-	33 days	NOEC (growth) LOEC (growth)	0.006 0.050	Lapota et al. 1993
Scallop (adult), Hinnites multirugosus	[c]	-	110 days	No effect on condition	0.204	Salazar et al. 1987
Pacific oyster (larva), Crassostrea gigas	[c]	-	30 days	100% mortality	2.0	Alzieu et al. 1980
Pacific oyster (larva), Crassostrea gigas	[c]	-	113 days	30% mortality and abnormal development	0.2	Alzieu et al. 1980
Pacific oyster (spat), Crassostrea gigas	TBTO	-	48 days	Reduced growth	0.020	Lawler and Aldrich 1987

Pacific oyster (spat), Crassostrea gigas	TBTO	-	14 days	Reduced oxygen consumption and feeding rates	0.050	Lawler and Aldrich 1987
Pacific oyster (spat), Crassostrea gigas	[c]	28.5-34.2	45 days	40% mortality; reduced growth	0.24	Thain and Waldock 1985
Pacific oyster (spat), Crassostrea gigas	[c]	28.5-34.2	45 days	90% mortality	2.6	Thain and Waldock 1985
Pacific oyster (spat), Crassostrea gigas	[c]	-	45 days	Reduced growth	0.24	Thain and Waldock 1986
Pacific oyster (spat), Crassostrea gigas	TBT	-	49 days	Shell thickening	0.020	Thain et al. 1987
Pacific oyster (spat), Crassostrea gigas	TBTO	29-32	56 days	No growth	1.557	Waldock and Thain 1983
Pacific oyster (spat), Crassostrea gigas	TBTO	29-32	56 days	Reduced growth	0.1460	Waldock and Thain 1983
Pacific oyster (adult), Crassostrea gigas	TBT (field)	-	-	Shell thickening	≥0.014	Wolniakowski et al. 1987

Table 6. (Continued)

Species	Chemical[a]	Salinity (g/kg)	Duration	Effect	Concentration (µg/L)[b]	Reference
Pacific oyster (larva), Crassostrea gigas	TBTF	18-21	21 days	Reduced number of normally developed larvae	0.02346	Springborn Bionomics, Inc. 1984a
Pacific oyster (larva), Crassostrea gigas	TBTF	18-21	15 days	100% mortality	0.04692	Springborn Bionomics, Inc. 1984a
Pacific oyster (embryo), Crassostrea gigas	TBTA	28	24 hr	Abnormal development; 30-40% mortality	4.304	His and Robert 1980

Pacific oyster (embryo), Crassostrea gigas	TBTA	-	24 hr	Abnormal development	0.8604	Robert and His 1981
Pacific oyster (larva), Crassostrea gigas	TBTA	-	24 hr	Abnormal development	≥0.9	Robert and His 1981
Pacific oyster (larva), Crassostrea gigas	TBTA	-	48 hr	100% mortality	2.581	Robert and His 1981
Pacific oyster (150-300 mg), Crassostrea gigas	[c]	-	56 days	No effect on growth	0.157	Salazar et al. 1987
Pacific oyster (3.5-25 mm), Crassostrea gigas	TBT (field)	-	2-5 mo	Reduced growth rate Normal growth rate	0.040 0.010	Stephanson 1991
Pacific oyster (fertilized eggs), Crassostrea gigas	TBTO	-	24 hr	LC50 Delayed development	7.0 1.8	Osada et al. 1993
Pacific oyster (straight-hinge larvae), Crassostrea gigas	TBTO	-	24 hr	LC50	15.0	Osada et al. 1993
Pacific oyster (spat), Crassostrea gigas	TBTO	-	48 hr	LC50	35.0	Osada et al. 1993
Pacific oyster (24-h-old), Crassostrea gigas	TBTA	-	12 days	LC50	0.04	His 1996

Table 6. (Continued)

Species	Chemical[a]	Salinity (g/kg)	Duration	Effect	Concentration (µg/L)[b]	Reference
Eastern oyster (2.7-5.3 cm), Crassostrea virginica	[d]	-	67 days	Decrease in condition index (body weight)	0.73	Valkirs et al. 1985

Eastern oyster (2.7-5.3 cm), Crassostrea virginica	[d]	-	67 days	No effect on survival	1.89	Valkirs et al. 1985
Eastern oyster (adult), Crassostrea virginica	[c]	33-36	57 days	Decrease in condition index	0.1	Henderson 1986
Eastern oyster (adult), Crassostrea virginica	[c]	33-36	30 days	LC50	2.5	Henderson 1986
Eastern oyster (embryo), Crassostrea virginica	TBTCl	18-22	48 hr	Abnormal shell development	0.77	Roberts 1987
Eastern oyster (juvenile), Crassostrea virginica	TBTO	11-12	96 hr	EC50; shell growth	0.31	Walker 1989b
Eastern oyster (adult), Crassostrea virginica	[c]	-	8 wks	No affect on sexual development, fertilization	1.142	Roberts et al. 1987
Eastern oyster (adult), Crassostrea virginica	TBT	-	21 wks	Immune response not weakened	0.1	Anderson et al. 1996
European flat oyster (spat), Ostrea edulis	TBTO	30	20 days	Significant reduction in growth	0.01946	Thain and Waldock 1985
European flat oyster (spat), Ostrea edulis	[c]	28.5-34.2	45 days	Decreased growth	0.2392	Thain and Waldock 1985; Thain 1986
European flat oyster (spat), Ostrea edulis	[c]	28.5-34.2	45 days	70% mortality	2.6	Thain and Waldock 1985; Thain 1986
European flat oyster (spat), Ostrea edulis	[c]	-	20 days	Reduced growth	0.02	Thain and Waldock 1986
European flat oyster (adult), Ostrea edulis	[c]	28-34	75 days	Complete inhibition of larval production	0.24	Thain 1986

Table 6. (Continued)

Species	Chemical[a]	Salinity (g/kg)	Duration	Effect	Concentration (µg/L)[b]	Reference
European flat oyster (adult), Ostrea edulis	[c]	28-34	75 days	Retardation of sex change from male to female	0.24	Thain 1986
European flat oyster (adult), Ostrea edulis	[c]	28-34	75 days	Prevented gonadal development	2.6	Thain 1986
European flat oyster (140-280 mg), Ostrea edulis	[c]	-	56 days	No effect on growth	0.157	Salazar et al. 1987
Native Pacific oyster (100-300 mg), Ostrea luricla	[c]	-	56 days	No effect on growth	0.157	Salazar et al. 1987
Quahog clam (embryo, larva), Mercenaria mercenaria	TBTO	-	14 days	Reduced growth	≥0.010	Laughlin et al. 1987;1988
Clam (adult), Macoma nasuta	[c]	-	110 days	No effect on condition	0.204	Salazar et al. 1987
Quahog clam (veligers), Mercenaria mercenaria	TBTO	-	8 days	Approx. 35% dead; reduced growth; ≥1.0 µg/L 100 mortality	0.6	Laughlin et al. 1987;1989
Quahog clam (post larva), Mercenaria mercenaria	TBTO	-	25 days	100% mortality	10	Laughlin et al. 1987;1989
Quahog clam (larva), Mercenaria mercenaria	TBTCl	18-22	48 hr	Delayed development	0.77	Roberts 1987
Common Pacific Littleneck (adult), Protothaca stamina	TBTO	33-34	96 hr	100% survival	≥2.920	Salazar and Salazar 1989
Copepod (subadult), Eurytemora affinis	TBT	10	72 hr	LC50	0.5	Bushong et al. 1988

Copepod (subadult), Eurytemora affinis	TBT	10	72 hr	LC50	0.6	Bushong et al. 1988
Copepod, Acartia tonsa	TBTO	-	6 days	EC50	0.3893	U'ren 1983

Table 6. (Continued)

Species	Chemical[a]	Salinity (g/kg)	Duration	Effect	Concentration (µg/L)[b]	Reference
Copepod (adult), Acartia tonsa	TBTO	28	5 days	Reduced egg production	0.010	Johansen and Mohlenberg 1987
Copepod (nauplii), Acartia tonsa	TBTCl	10-12	9 days	Reduced survival	≥0.029	Bushong et al. 1990
Copepod (nauplii), Acartia tonsa	TBTCl	10-12	6 days	Reduced survival; no effect 0.012 µg/L	0.023	Bushong et al. 1990
Copepod (nauplii), Acartia tonsa	TBTCl	10-12	6 days	Reduced survival; no effect 0.010 µg/L	0.024	Bushong et al. 1990
Copepod (nauplii), Acartia tonsa	TBTCl	18	8 days	Inhibition of development EC50 (survival)	0.003 0.015-0.020	Kusk and Peterson 1997
Amphipod (larva, juvenile), Gammarus oceanus	TBTO	7	8 wk	100% mortality	2.920	Laughlin et al. 1984b
Amphipod (larva, juvenile), Gammarus oceanus	TBTF	7	8 wk	100% mortality	2.816	Laughlin et al. 1984b
Amphipod (larva, juvenile), Gammarus oceanus	TBTO	7	8 wk	Reduced survival and growth	0.2920	Laughlin et al. 1984b
Amphipod (larva, juvenile), Gammarus oceanus	TBTF	7	8 wk	Reduced survival and increased growth	0.2816	Laughlin et al. 1984b
Amphipod, Gammarus sp.	TBTCl	10	24 days	No effect	0.579	Hall et al. 1988b

Amphipod (adult), Orchestia traskiana	TBTO	30	9 days	Approx. 80% mortality	9.732	Laughlin et al. 1982
Amphipod (adult), Orchestia traskiana	TBTF	30	9 days	Approx. 90% mortality	9.732	Laughlin et al. 1982
Amphipod (adult), Eohaustorius estuarius	TBTCl (96%)	28.8-29.5	10 days	BCF = 41,200 (no plateau)	0.48	Meador et al. 1993
Amphipod (adult), Eohaustorius washingtonianus	TBTCl (96%)	32.7	10 days	BCF = 60,300 (no plateau)	109	Meador 1997
Amphipod (adult), Rhepoxynius abronius	TBTCl (96%)	32.3	10 days	BCF = 1,700 (no plateau)	660	Meador 1997

Table 6. (Continued)

Species	Chemical[a]	Salinity (g/kg)	Duration	Effect	Concentration (µg/L)[b]	Reference
Grass shrimp, Palaemonetes pugio	TBTO (95%)	9.9-11.2	20 min	No avoidance	30	Pinkney et al. 1985
Grass shrimp, Palaemonetes pugio	TBTO	20	14 days	Telson regeneration retarded; molting retarded	0.1	Khan et al. 1993
Mud crab (larva), Rhithropanopeus harrisii	TBTO	15	15 days	Reduced developmental rate and growth	14.60	Laughlin et al. 1983
Mud crab (larva), Rhithropanopeus harrisii	TBTS	15	15 days	Reduced developmental rate and growth	18.95	Laughlin et al. 1983
Mud crab (larva), Rhithropanopeus harrisii	TBTO	15	15 days	63% mortality	>24.33	Laughlin et al. 1983
Mud crab (larva), Rhithropanopeus harrisii	TBTS	15	15 days	74% mortality	28.43	Laughlin et al. 1983
Mud crab (zoea), Rhitropanopeus harrisii	TBTO	15	20 days	LC50	13.0	Laughlin and French 1989

Mud crab (zoea), Rhithropanopeus harrisii	TBTO	15	40 days	LC50	33.6	Laughlin and French 1989
Mud crab, Rhithropanopeus harrisii	TBTO	15	6 days	BCF=24 for carapace	5.937	Evans and Laughlin 1984
Mud crab, Rhithropanopeus harrisii	TBTO	15	6 days	BCF=6 for hepatopancreas	5.937	Evans and Laughlin 1984
Mud crab, Rhithropanopeus harrisii	TBTO	15	6 days	BCF=0.6 for testes	5.937	Evans and Laughlin 1984
Mud crab, Rhithropanopeus harrisii	TBTO	15	6 days	BCF=41 for gill tissue	5.937	Evans and Laughlin 1984
Mud crab, Rhithropanopeus harrisii	TBTO	15	6 days	BCF=1.5 for chelae muscle	5.937	Evans and Laughlin 1984
Fiddler crab, Uca pugilator	TBTO	25	≤24 days	Retarded limb regeneration and molting	0.5	Weis et al. 1987a
Fiddler crab, Uca pugilator	TBTO	25	3 weeks	Reduced burrowing	0.5	Weis and Perlmutter 1987

Table 6. (Continued)

Species	Chemical[a]	Salinity (g/kg)	Duration	Effect	Concentration (µg/L)[b]	Reference
Fiddler crab, Uca pugilator	TBTO	25	7 days	Limb malformation	0.5	Weis and Kim 1988; Weis et al. 1987a
Blue crab (6-8-day-old embryos), Callinectes sapidus	TBT	28	4 days	EC50 (hatching)	0.047	Lee et al. 1996

Brittle star, Ophioderma brevispina	TBTO	18-22	4 wks	Retarded arm regeneration	-0.1	Walsh et al. 1986a
Atlantic menhaden (juvenile), Brevoortia tyrannus	TBTCl	10	28 days	No effect	0.490	Hall et al. 1988b
Atlantic menhaden (juvenile), Brevoortia tyrannus	TBTO	9-11	-	Avoidance	5.437	Hall et al. 1984
Chinook salmon (adult), Oncorhynchus tshawytscha	TBTO	28	96 hr	BCF=4300 for liver	1.49	Short and Thrower 1986a,1986c
Chinook salmon (adult), Oncorhynchus tshawytscha	TBTO	28	96 hr	BCF=1300 for brain	1.49	Short and Thrower 1986a,1986c
Chinook salmon (adult), Oncorhynchus tshawytscha	TBTO	28	96 hr	BCF=200 for muscle	1.49	Short and Thrower 1986a,1986c
Mummichog (juvenile), Fundulus heteroclitus	TBTO	2	6 wks	Gill pathology	17.2	Pinkney 1988; Pinkney et al. 1989a
Mummichog, Fundulus heteroclitus	TBTO	9.9-11.2	20 min	Avoidance	3.7	Pinkney et al. 1985
Inland silverside (larva), Menidia beryllina	TBTCl	10	28 days	Reduced growth	0.093	Hall et al. 1988b
Mummichog (embryo), Fundulus heteroclitus	TBTO	25	10 days	Teratotogy	30	Weis et al. 1987b
Mummichog (5.3 cm; 1.8 g), Fundulus heteroclitus	TBTO (95%)	15 16-19.5	96 hr 6 wks	LC50 NOEC	17.2 2.000	Pinkney et al. 1989a
Three-spined stickleback (45-60 mm), Gasterosteus aculeatus	TBTO (painted panels)	15-35	7.5 mo	80% mortality (2 months) Histological effects	10 2.5	Holm et al. 1991

Table 6. (Continued)

Species	Chemical[a]	Salinity (g/kg)	Duration	Effect	Concentration (µg/L)[b]	Reference
California grunion (gamete through embryo), Leuresthes tenuis	[c]	-	10 days	Significantly enhanced growth and hatching success	0.14-1.71	Newton et al. 1985
California grunion (gamete through embryo), Leuresthes tenuis	[c]	-	10 days	Significantly enhanced growth and hatching success	0.14-1.72	Newton et al. 1985
California grunion (gamete through embryo), Leuresthes tenuis	[c]	-	10 days	50% reduction in hatching success	74	Newton et al. 1985
California grunion (embryo), Leuresthes tenuis	[c]	-	10 days	No adverse effect on hatching success or growth	0.14-1.72	Newton et al. 1985
California grunion (larva), Leuresthes tenuis	[c]	-	7 days	Survival increased as concentration increased	0.14-1.72	Newton et al. 1985
Striped bass (juvenile), Morone saxatilis	TBTO (95%)	9-11	-	Avoidance	24.9	Hall et al. 1984
Striped bass (juvenile), Morone saxatilis	TBT (painted panels)	13.0-15.0	14	NOEC (serum ion concentrations and enzyme activity)	1.09	Pinkney et al. 1989b
Striped bass (juvenile), Morone saxatilis	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), Citharichthys stigmaeus	TBTO	33-34	96 hr	LC50	18.5	Salazar and Salazar 1989
Stripped mullet (3.2 g); Mugil cephalus	TBTO (96%)	-	8 wks	BCF 3000 (no plateau) BCF 3600 (no plateau)	0.122 0.106	Yamada and Takayanagi 1992

Starry flounder (<l-year-old), Platichthys stellatus	TBTCl (96%)	30.2	10 days	BCF 8,700 (no plateau)	194	Meador 1997

[a] TBTA = tributyltin acetate; TBTCl = 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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