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# AMBIENT AQUATIC LIFE WATER QUALITY CRITERIA

## *Tributyltin* -- DRAFT

DRAFT
1997

# AMBIENT AQUATIC LIFE WATER QUALITY CRITERIA FOR

# TRIBUTYLTIN

CAS Registry Number (See Text)

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## NOTICES

This document has been reviewed by the Environmental Research Laboratories, Duluth, MN and Narragansett, RI, Office of Research and Development and the Health and Ecological Criteria Division, Office of Science and Technology, U.S. Environmental Protection Agency, and approved for publication.

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## FOREWORD

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

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

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

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## Introduction

Organotins are compounds consisting of one to four organic moieties attached to a tin atom via carbon-tin covalent bonds. When there are fewer than four carbon-tin bonds, the organotin compound will be a cation unless the remaining valences of tin are occupied by 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).

Several review papers have been written which cover the production, use, chemistry, toxicity, fate and hazards of TBT in the aquatic environment (Clark et al. 1988; Eisler 1989; Oceans 86 1986; Oceans 87 1987; WHO 1990). The toxicities of organotin compounds are related to the number of organic moieties bonded to the tin atom and to the number of carbon atoms in the organic moieties. Toxicity to aquatic organisms generally increases as the number of organic moieties 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).

Organotins are used in several manufacturing processes, for example, as an anti-yellowing agent in clear plastics and as a catalyst in poly(vinyl chloride) products (Piver 1973). One of the more extensive uses of organotins is as biocides (fungicides, bactericides, insecticides) and as preservatives for wood, textiles, paper, leather and electrical equipment. The use of TBT in antifouling paints on ships, boats, docks and cooling towers probably contributes most significantly to direct release of organotins into the aquatic environment (Clark et al. 1988; Hall and Pinkney 1985; Kinnetic Laboratory 1984).

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 of these organisms on ships' hulls 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. Release of TBT to water occurs during repainting in shipyards when old paint is sand-blasted off and new paint applied. TBT would also be released continuously from the hulls of the painted ships. Antifouling paints in current use contain copper as the primary biocide, whereas the proposed TBT paints would contain both copper and TBT. Interaction between the toxicities of TBT and other ingredients in the paint apparently is negligible (Davidson et al. 1986a).

The solubility of TBT compounds in water is influenced by such factors as the oxidation-reduction potential, pH, temperature, ionic strength, and concentration and composition of the dissolved organic matter (Clark et al. 1988; Corbin 1976). The solubility of tributyltin oxide in water was reported to be 750 $\mu g/L$ at pH of 6.6, 31,000 $\mu g/L$ at pH of 8.1 and 30,000 $\mu g/L$ at pH 2.6 (Maguire et al. 1983). The carbon-tin covalent bond does not hydrolyze in water (Maguire et al. 1983,1984), and the half-life for photolysis due to sunlight is greater than 89 days (Maguire et al. 1985; Seligman et al. 1986). Biodegradation is the major breakdown pathway for TBT in water and sediments with half-lives of several days in water to several weeks in sediments (Clark et al. 1988; Lee et al. 1987; Maguire and Tkacz 1985; Seligman et al. 1986, 1988, 1989; Stang and Seligman 1986). Breakdown products include di- and monobutyltins with some butylmethyltins detected.

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 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 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 1986).

TBT readily sorbs to sediments and suspended solids and can persist there (Cardarelli and Evans 1980). In some instances, most TBT in the water column (70-90%) is associated with the dissolved phase (Valkirs et al. 1986a; Maguire 1986; Johnson et al. 1987). The half-life for degradation of TBT from sediments is reported to be greater than ten months (Maguire and Tkacz 1985). In a modeling and risk assessment study of TBT, 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.

Elevated TBT concentrations in fresh and salt waters, sediments or biota, are primarily associated with harbors and marinas (Cleary and Stebbing 1985; Hall 1988; Hall et al. 1986; Langston et al. 1987; Maguire 1984,1986; Maguire and Tkacz 1985; Maguire et al. 1982; 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; Waldock and Miller 1983; Waldock et al. 1987). 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. Organotin concentrations in the low part per trillion range have been associated with oyster shell malformations (Alzieu et al. 1989; Minchin et al. 1987). Reevaluation of a harbor in the United Kingdom show that since the 1987 restrictions on tributyltin use, oyster culture has returned in the harbor (Dyrynda 1992). In some cases the water surface microlayer contained a much higher concentration of TBT than the water column (Cleary and Stebbing 1987; Hall et al. 1986; Valkirs et al. 1986a). Gucinski (1986) suggested that this enrichment of the surface microlayer might increase the bioavailability of TBT. TBT accumulates in sediments with sorption coefficients which may 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). 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 pens treated with TBT (Short 1987; Short and Thrower 1986a).

Only data generated in toxicity and bioconcentration tests on TBTC (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. 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 requires 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.

## 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. The acute values range from 1.14 μg/L for a hydra, Hydra oligactis, to 24,600 μg/L for a freshwater calm, Elliptio complanatus. 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 tolerate high concentrations of TBT temporarily.

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

All the vertebrate species tested are fish. The most sensitive species is the fathead minnow, Pimephales promelas, which has a SMAV of 2.6 μg/L from a single 96-hr flow-through measured test (Brooke et al. 1986). Rainbow trout, Oncorhynchus mykiss, were tested by four groups with good agreement. 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 flow-through and concentrations were measured. 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 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.

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 23 species of invertebrates and six species of fish (Table 1). The range of acute toxicity to saltwater animals is a factor of about 672. Acute values range from 0.42 μg/L for juveniles of the mysid, Acanthomysis sculpta (Davidson et al. 1986a,1986b) 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). 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, Manuscript). 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 27 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 13 most sensitive genera differ by a factor of less than four. Included within these genera are four species of molluscs and nine species of crustaceans. The saltwater Final Acute Value for TBT was calculated to be 0.7347 μg/L (Table 3), which is greater than the lowest saltwater Species Mean Acute Value of 0.61 μg/L.

## 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 Daphnia 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 the reproductive effects on adult daphnids. The chronic value for Daphnia magna is calculated to be 0.1414 μg/L, 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 set at 0.19 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 D. magna.

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 reduced by 2% at the next lower TBT concentration of 0.92 μg/L, but was higher than in the controls and at 0.45 μg/L and lower 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. Mean length of fry at the end of the test was significantly reduced at concentrations ≥0.45 μg/L. The mean biomass at the end of the test was higher at the 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.

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. Percentage survival of neonates relative to controls was 21% in 0.088 μg/L (nominal concentration of 0.100 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 (27% relative to controls) in 0.224 μg/L; brood size was unaffected in any tested concentration (0.018-0.224 μg/L). Although no statistically significant effects were detected in ≤0.100 μg/L, percentage survival of neonates appears reduced; 76% vs 90% in controls. The chronic value in this test is 0.150 ug/L. Survival of neonates in both tests in the 0.100 μg/L nominal concentration (mean measured concentration = 0.094 μg/L) averaged 42% relative to controls. If this is the best estimate of the upper chronic value, and the 0.056 μg/L treatment from the second test is the best estimate of the lower chronic value, the overall chronic value for the two tests is 0.0725 μg/L. The overall acute-chronic ratio is 27.24 when the acute value of 1.975 μg/L (mean of acute values of 1.4, 2.2 and 2.5 μg/L) is used.

Life-cycle toxicity tests have been 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, 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 0.09 μg/L was higher than in 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 is 0.1308 μg/L, and the acute-chronic ratio is 4.664 (Table 2).

The Final Acute-Chronic Ratio of 14.69 was calculated as the geometric mean of the acute-chronic ratios of 36.60 for *Daphnia magna*, 10.01 for *Pimephales promelas*, 4.664 for *Acanthomysis sculpta* and 27.24 for *Eurytemora affinis*. Division of the freshwater and saltwater Final Acute Values by 14.69 results in Final Chronic Values for freshwater of 0.0625 μg/L and for saltwater of 0.0500 µg/L (Table 3). Both of these Chronic Values are below the experimentally determined chronic values from life-cycle or early life-stage tests.

Toxicity to Aquatic Plants

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 (Table 1). No data are available on the effects of TBT on freshwater vascular plants.

Toxicity tests on TBT have been conducted with five species of saltwater phytoplankton including the 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 (Tables 4 and 6). The 14-day EC50 of 0.06228 µg/L for S. costatum (EG&G Bionomics 1981c) was the lowest value reported, but Thain (1983) reported that a measured concentration of 0.9732 µg/L was algistatic to the same species (Table 4). 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 on tributyltin acetate, tributyltin chloride, tributyltin fluoride, and tributyltin oxide 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).

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). The zebra mussel, Dressena polymorpha, was monitored in cages in a marina and at an uncontaminated site in a lake for 105 days (van Slooten and Tarradellas 1994). The organisms reached steady-state concentrations after 35 days. The BCF/BAF was 180,427 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 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 and guppy demonstrated a plateau BCF in 14 days and BCFs of 501.2 and 460, respectively. Goldfish 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, a snail, 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. 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). Laboratory BCFs for the snail Nucella lapillus (11,000 to 38,000) were also similar to field BAFs (17,000) (Bryan et al. 1987). The soft parts of the Pacific oyster 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 1986; Thain and Waldock 1985). Other tests with the same species (Table 5) resulted in BCFs ranging from 397 to 1,167. One species of saltwater fish was exposed to 0.28 μg/L for 14 days and a plateau BCF of 240 was demonstrated (Tsuda et al. 1990b).

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

Additional data on the lethal and sublethal effects of TBT on aquatic species are presented in Table 6. Two microcosm studies were conducted by Delupis and Miniero (1989, 1991) in which single dose effects were measured on natural assemblages of organisms. In both studies the effects were immediate. Daphnia magna disappeared soon after a 80 μg/L dose of TBT, ostracods increased, and algal species increased immediately then gradually disappeared. In the second study metabolism was monitored by measuring oxygen consumption. Doses of TBT (4.7 and 14.9 μg/L) were administered once and metabolism was reduced by 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.

Wong et al. (1982) exposed a natural assemblage of freshwater algae and several pure cultures of various algal species to TBT in 4-hr exposures. Effects (EC50s) were seen in all cases on the production or reproduction at concentrations ranging from 5 to 20 μg/L which demonstrates a high sensitivity to TBT.

Freshwater rotifers (Brachionus calyciflorus) and 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 was not affected at the sensitive life-stage of hatching until an exposure concentration reached 72 µg/L.

Larvae of the clam, Corbicula fluminea, has a 24-hr EC50 of 1,990 µg/L which is a high concentration relative to most other species of tested freshwater organisms. Another species of clam, Elliptio complanatus, also showed low sensitivity to TBT with a 96-hr LC50 of 24,600 µg/L (Table 1). Various bivalve clam species may have the ability to reduce exposure to TBT temporarily by closing the valves.

The cladoceran, Daphnia magna, has 24-hr EC50s ranging from 3 to 13.6 µg/L (Polster and Halacha 1972; Vighi and Calamari 1985). When a more sensitive endpoint of altered phototaxis was examined in a longer-term exposure of 8 days, the effect concentration (0.45 µg/L) was much lower (Meador 1986). Similarly, rainbow trout (Oncorhynchus 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 10% reduction in growth is 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 at 21 days when exposed to 0.5 µg/L. The frog, Rana temporaria, has a LC50 of 28.2 µg/L for a 5-day exposure to TBT.

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). 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 (RPS; ratio of female to male penis volume) and the six developmental stages of the vas deferens sequence (VDS) (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 through planktonic larval stages, the impacts of TBT are less certain because recruitment is facilitated. Natural pseudohemaphiodism 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 high concentrations of TBT in water, sediment or snails and other biota (Alvarez and Ellis 1990; Bailey and Davies 1988a,1988b; Bryan et al. 1986,1987,; Davies et al. 1987, Durchon 1982; Ellis and Pattisima 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).

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 (Nucella lapillus), file dogwhinkle (N. lima), eastern mud snail [Ilyanassa (Nassarius) obsoleta], a snail (Hinia reticulta), welks (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 (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 as 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. 1989). However, other causes may explain this as oviducts were not blocked and indirect development facilitating recruitment 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 Nucellus 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, Illyanassa obsoleta, collected from the York River, VA near Sarah Creek had no incidence of imposex (Bryan et al. 1989) and contained no detectable TBT, (<0.020 µg/g dry weight). The average TBT concentrations of York River water was 0.0016 µg/L. In contrast, the average TBT concentrations from four locations in Sarah Creek, VA were from 0.010 to 0.023

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

| Species | Experimental Design | TBT Concentration: Water µg/L | TBT Concentration: Snail Tissue µg/g dry | Imposex: RPS % | Imposex: VSD | Imposex: 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 | |
| Welk, *Thais orbita* | Field-Queenscliff, UK | - | 0.365* | 19.55 | - | 100% incidence | Foale 1993 |
| | -Sandringham | - | 0.224* | 12.16 | - | 100% incidence | |
| | -Brighton | - | <0.002* | 7.34 | - | 100% incidence | |
| | -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 | |
| Welk, *Thais clavigera* | Field-32 sites Japan | - | 0.005 | <10 | - | 100% incidence | Horiguchi et al. 1995 |
| | | - | 0.010 | 1-42 | - | 100% incidence | |
| | | - | 0.020 | 30-75 | - | 100% incidence | |
| | | - | 0.030 | 30 | - | 100% incidence | |
| | | - | 0.040 | 75 | - | 100% incidence | |
| | | - | 0.050 | 80 | - | 100% incidence | |
| | | - | 0.060 | 80 | - | 100% incidence | |
| | | - | 0.070 | 40 | - | 100% incidence | |
| | | - | 0.080 | 40 | - | 100% incidence | |

Text Table 1. Continued

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

Text Table 1. Continued

| Species | Experimental Design | TBT Concentration: Water µg/L | TBT Concentration: Snail Tissue µg/g dry | Imposex: RPS % | Imposex: VSD | Imposex: Comments | Reference |
|---|---|---|---|---|---|---|---|
| Atlantic dogwhinkle, *Nucella lapillus* | Transplants, Crooklets Beach to Dart Estuary, UK | 0.022-0.046 | 9.7 | 96.3 | 5.0 | All sterile | Gibbs et al. 1988 |
| 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 | <0.002 | <0.10* | 0.10 | 1.06 | Control, 37.1% imposex | Bailey et al. 1991 |
| | | <0.002 | <0.10* | 0.04 | 0.70 | Solvent control, 24.3% imposex | |
| | | <0.003 | 0.35* | 5.33 | 3.15 | 5.3% reduced growth, 92.3% imposex | |
| | | 0.008 | 1.10* | 20.84 | 3.97 | 11.0% reduced growth, 100% imposex | |
| | | 0.033 | 3.05* | 42.08 | 4.33 | 17.1% reduced growth, 100% imposex | |
| | | 0.125 | 4.85* | 63.40 | 4.25 | 18.9% reduced growth, 100% 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.

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, _Nucella 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.

The effects of TBT on the development of imposex has been studied most in the Atlantic dogwhinkle, _Nucella lapillus_. Bryan et al. (1987a) exposed adult snails for two years to 0.0036 (control), 0.0083, 0.046 and 0.26 μg/L in the laboratory and compared responses to a field control. Imposex was present in laboratory "control" snails exposed to 0.0036 μg/L and extent of penis and vas deferens development increased significantly with increase in TBT exposure; sterility occurred in some snails exposed to 0.26 μg/L. In a similar laboratory experiment that began with snail egg capsules and lasted two years (Gibbs et al. 1988), imposex development was more severe. Field controls spawned and females were normal in <0.0012 μg/L. In the laboratory, one-third of the snails exposed to 0.0036 μg/L were sterile and 160 egg cases were produced. At ≥0.0093 μg/L all females were sterile with only two 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 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 reduce 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/L) before showing affects of imposex. Thais sp. showed high imposex incidence at tissue concentrations as low as 0.005 µg/L and no imposex at other locations with tissue concentrations of 0.108 µg/L. Ocenebra erinacea did not show imposex in a field study at body burdens as high as 0.185 µg/L, 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 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 ≥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 ≥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 ≥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, _Hinmites 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 _Acartia tonsa_ for five days to TBT and observed impaired egg production on days 3, 4 and 5 in 0.1 μg/L and only on day 5 in 0.01 and 0.05 μg/L. For the five days, overall egg production was reduced markedly (25%) only in 0.1 μg/L. Davidson et al. (1986a,1986b), Laughlin et al. (1983,1984b), and Salazar and Salazar (1985a) reported that TBT acts slowly on crustaceans and that behavior might be affected several days before mortality occurs. Survival of larval amphipods, _Gammarus oceanicus_, was significantly reduced after eight weeks of exposure to TBT concentrations ≥0.2816 μg/L (Laughlin et al. 1984b). Hall et al. (1988b) observed no effect of 0.579 µg/L on Gammarus sp. after 24 days. Developmental rates and growth of larval mud crabs, Rhithropanopeus 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 0.0500 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 Acartia 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).

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 0.0500 µg/L

| Species | Experimental Design[a] | Concentration (µg/L) | Response | Reference |
|---|---|---|---|---|
| | | Measured | | |
| Copepod (nauplii-adult), <u>Acartia</u> <u>tonsa</u> | #1: F,M, 9-day duration, ≥10 copepods/replicate, 4 replicates | control<br>0.029<br>0.05-0.5 | 77% survival<br>23% survival[b]<br>0-2% survival[b] | Bushong et al. 1990 |
| | #2: F,M, 6-day duration, ≥10 copepods/replicate, 4 replicates | control<br>0.007-0.012<br>0.023<br>0.048-0.102 | 71% survival<br>32% survival<br>19% survival[b]<br>0-14% survival[b] | |
| | #3: F,M, 6-day duration, ≥10 copepods/replicate, 4 replicates | control<br>0.006-0.010<br>0.024<br>0.051-0.115 | 59% survival<br>44-46% survival<br>30% survival[b]<br>2-35% survival[b] | |
| | | Nominal | | |
| Hard clam (4 hr larvae - metamorphosis), <u>Mercenaria</u> <u>mercenaria</u> | 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<br>0.01-0.5 | 100% Growth (Valve length)<br>~75%-22% Growth (Value length)[b] | Laughlin et al. 1987,1988 |
| | | Nominal | | |
| Pacific oyster (spat), <u>Crassostrea</u> <u>gigas</u> | R,N, 48-day duration, 20 spat/treatment | control<br>0.01-0.05<br>control<br>0.01-0.2<br>0.02-0.2 | Shell thickening<br>100% Growth (Valve length)<br>101% Growth (Value length)<br>0-72% Growth (Value length)[b] | Lawler and Aldrich 1987 |
| | | Nominal | | |
| Pacific oyster (spat), <u>Crassostrea</u> <u>gigas</u> | R,N, 49-day duration, 0.7 to 0.9 g/spat | control<br>0.002<br>0.02-2.0 | No shell thickening<br>Shell thickening proportional to concentration increase | Thain et al. 1987 |

Text Table 2. (Continued)

| Species | Experimental Design[a] | Concentration (µg/L) | Response | Reference |
|---|---|---|---|---|
| | | Measured | | |
| | Field | 0.011-0.015<br>0.018-0.060 | No shell thickening<br>Shell thickening and decreased meat weight | |
| | | Measured | | |
| 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 |
| | | 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<br>0.02-2.0<br>control<br>0.02-2.0 | 100 length<br>76-81% length[b]<br>202% weight gain<br>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.

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 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 0.0500 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 0.01 µg/L to limit unacceptable impacts on Acartia tonsa, Mercenaria mercenaria, Crassostrea gigas and Ostrea edulis observed at 0.02 µg/L. At this criteria concentration, imposex would be expected in Ilyanassa obsoleta, Nucella lapillus and similarly sensitive neogastropods; populations of N. lapillus and similarly sensitive snails with direct development might be impacted and growth of Mercenaria mercenaria might be somewhat 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; Harding et al. 1995; 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; 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), Valkirs et al. (1985), 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; Lee 1985; Mercier et al. 1994; Nosov and Kolosova 1979; Smith 1981c; Stroganov et al. 1972,1977). The 96-hr LC50 of 0.01466 $\mu$g/L reported by Becerra-Huencho (1984) for post larvae of the hard clam, Mercenaria 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 $\mu$g/L (Tables 1 and 6) cast doubt on this LC50 value. 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; Laughlin et al. 1982; Maguire and Tkacz 1985; 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; Rice et al. 1995; Rice and Weeks 1990; Rouleau et al. 1995). Caricchia et al. (1991), Salazar and Chadwick (1991), Salazar and Salazar (1990a, 1990b, 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 (Manuscript) 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 (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).

Summary

The acute toxicity values for thirteen freshwater animal species range from 1.14 μg/L for a hydra (Hydra oligactis) to 24,600 μg/L for a clam (Elliptio complanatus). 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, the remaining species sensitivities varied by a maximum of 11.2 times. Three chronic toxicity tests have been conducted with freshwater animals. Reproduction of Daphnia magna was reduced by 0.2 μg/L, but not by 0.1 μg/L, and the Acute-Chronic Ratio is 30.41. In another test with D. magna reproduction and survival was reduced at 0.34 μg/L but not at 0.19, and the Acute-Chronic Ratio is 44.06. Weight of fathead minnows was reduced by 0.45 μg/L, but not by 0.15 μg/L, and the acute-chronic ratio for this species was 10.01. Bioconcentration of TBT was measured in zebra mussels, Dressena polymorpha, at 180,427 times the water concentration for the soft parts and in rainbow trout, Oncorhynchus mykiss, at 406 times the water concentration for the whole body. Growth of thirteen species of freshwater algae was inhibited by concentrations ranging from 56.1 to 1,782 µg/L.

Acute values for 27 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 4. Larvae and juveniles appear to be more sensitive than adults. A life-cycle toxicity test has been 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. Tributyltin chronically affects certain saltwater copepods, gastropods, and pelecypods at concentrations less than those predicted from "standard" acute and chronic toxicity tests. Survival of the copepod Acartia tonsa was reduced in ≥0.023 µg/L. Growth of larvae or spat of two species of oysters, Crassostrea gigas and Ostrea edulis was reduced in about 0.02 µg/L; some C. gigas larvae died in 0.025 µg/L. Generally concentrations ≤0.01 µg/L have not been demonstrated to affect sensitive life-stages of saltwater organisms. This above data demonstrate that reductions in growth occur in commercially or ecologically important saltwater species at concentrations of TBT less than the Final Chronic Value of 0.0500 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 0.01 µg/L to limit unacceptable impacts on Acartia tonsa, Mercenaria mercenaria, Crassostrea gigas and Ostrea edulis observed at 0.02 µg/L. At this criteria concentration, imposex would be expected in Ilyanassa obsoleta, Nucella lapillus and similarly sensitive neogastropods; populations of N. lapillus and similarly sensitive snails with direct development might be impacted and growth of Mercenaria mercenaria might be somewhat lowered.

## 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 $\mu$g/L more than once every three years on the average and if the one-hour average concentration does not exceed 0.46 $\mu$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 0.010 $\mu$g/L more than once every three years on the average and if the one-hour average concentration does not exceed 0.37 $\mu$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 | TBTC | - | 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 | TBTC | 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 (µg/L)[c] | Species Mean Acute Value (µg/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 |

Table 1. (Continued)

| Species | Method[a] | Chemical[b] | Salinity (g/kg) | LC50 or EC50 (µg/L)[c] | Species Mean Acute Value (µg/L) | Reference |
|---|---|---|---|---|---|---|
| Lugworm (larva), <u>Arenicola</u> <u>cristata</u> | S,U | TBTA | 28 | ~5-10 | ~5.03 | Walsh et al. 1986b |
| Polychaete (juvenile), <u>Neanthes</u> <u>arenaceodentata</u> | S,U | TBTO | 33-34 | 6.812 | - | Salazar and Salazar, Manuscript |
| Polychaete (adult), <u>Neanthes</u> <u>arenaceodentata</u> | S,U | TABTO | 33-34 | 21.41[e] | 6.812 | Salazar and Salazar, Manuscript |
| Blue mussel (larva), <u>Mytilus</u> <u>edulis</u> | R,- | TATO | - | 2.238 | - | Thain 1983 |
| Blue mussel (adult), <u>Mytilus</u> <u>edulis</u> | R,- | TBTO | - | 36.98[e] | - | Thain 1983 |
| Blue mussel (adult), <u>Mytilus</u> <u>edulis</u> | S,U | TBTO | 33-34 | 34.06[e] | 2.238 | Salazar and Salazar, Manuscript |
| Pacific oyster (larva), <u>Crassostrea</u> <u>gigas</u> | R,- | TBTO | - | 1.557 | - | Thain 1983 |
| Pacific oyster (adult), <u>Crassostrea</u> <u>gigas</u> | R,- | TBTO | - | 282.2[e] | 1.557 | Thain 1983 |
| Eastern oyster (embryo), <u>Crassostrea</u> <u>virginica</u> | S,U | TBTO | 22 | 0.8759 | - | EG&G Bionomics 1977 |
| Eastern oyster (embryo), <u>Crassostrea</u> <u>virginica</u> | R,U | TBTC | 18-22 | 1.30 | - | Roberts 1987 |
| Eastern oyster (embryo), <u>Crassostrea</u> <u>virginica</u> | R,U | TBTC | 18-22 | 0.71 | - | Roberts 1987 |
| Eastern oyster, <u>Crassostrea</u> <u>virginica</u> | R,U | TBTC | 18-22 | 3.96[e] | 0.9316 | Roberts 1987 |
| European flat oyster (adult), <u>Ostrea</u> <u>edulis</u> | R,- | TBTO | - | 204.4 | 204.4 | Thain 1983 |
| Hard clam (post larva), <u>Mercenaria</u> <u>mercenaria</u> | S,U | TBTC | - | 0.01466[f] | - | Becerra-Huencho 1984 |
| Hard clam (embryo), <u>Mercenaria</u> <u>mercenaria</u> | R,U | TBTC | 18-22 | 1.13 | - | Roberts 1987 |

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 (larva), *Mercenaria mercenaria* | R,U | TBTC | 18-22 | 1.65 | 1.365 | Roberts 1987 |
| Copepod (juvenile), *Eurytemora affinis* | F,M | TBTC | 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 (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, Manuscript |
| Mysid (subadult), *Metamysidopsis elongata* | S,U | TBTO | 33-34 | 1.946[e] | - | Salazar and Salazar, Manuscript |
| Mysid (adult), *Metamysidopsis elongata* | S,U | TBTO | 33-34 | 2.433[e] | - | Salazar and Salazar, Manuscript |
| Mysid (adult), *Metamysidopsis elongata* | S,U | TBTO | 33-34 | 6.812[e] | <0.9732 | Salazar and Salazar, Manuscript |
| Mysid (<1 day), *Mysidopsis bahia* | F,M | TBTC | 19-22 | 1.1 | - | Goodman et al. 1988 |

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 (5 day), *Mysidopsis bahia* | F,M | TBTC | 19-22 | 2.0 | - | Goodman et al. 1988 |
| Mysid (10 day), *Mysidopsis bahia* | F,M | TBTC | 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 (3-5 mm; 2-5 mg), *Rhepoxynius abronius* | R,M | TBTC | - | 22.8 | - | Meador et al. 1993 |
| Amphipod (3-5 mm; 2-5 mg), *Rhepoxynius abronius* | R,M | TBTC | - | 14.2 | 18.0 | Meador et al. 1993 |
| Amphipod (3-5 mm; 2-5 mg), *Eohaustorius estuarius* | R,M | TBTC | - | 2.0 | - | Meador 1993; Meador et al. 1993 |
| Amphipod (3-5 mm; 2-5 mg), *Eohaustorius estuarius* | R,M | TBTC | - | 1.5 | 1.73 | Meador 1993 |
| 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 |
| 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 |

Table 1. (Continued)

| Species | Method[a] | Chemical[b] | Salinity (g/kg) | LC50 or EC50 (µg/L)[c] | Species Mean Acute Value (µg/L | Reference |
|---|---|---|---|---|---|---|
| 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 |
| 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 1976 |

Table 1. (Continued)

| Species | Method[a] | Chemical[b] | Salinity (g/kg) | LC50 or EC50 (µg/L)[c] | Species Mean Acute Value (µg/L) | Reference |
|---|---|---|---|---|---|---|

| | | | | | | |
|---|---|---|---|---|---|---|
| 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 |

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

[b] TBTC = 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** | | | | | | |
| Copepod, *Eurytemora affinis* | LC | TBTC | 10.3[e] | <0.088 | <0.088 | Hall et al. 1987;1988a |
| Copepod, *Eurytemora affinis* | LC | TBTC | 14.6[e] | 0.100-0.224 | 0.150 | 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; TBTC = 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).

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* | - | 1.975 | <0.088 | >22.44 |
| Copepod, *Eurytemora affinis* | - | 1.975 | 0.150 | 13.17 |
| Mysid, *Acanthomysis sculpta* | - | 0.61[a] | 0.1308 | 4.664 |

[a] Reported by Valkirs et al. (1985).

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 complanatus* | 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 | - |

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] |
|---|---|---|---|---|
| 1 | 1.170 | Hydra, *Hydra littoralis* | 1.201 | - |
| | | Hydra, *Hydra oligactis* | 1.14 | - |
| | | **SALTWATER SPECIES** | | |
| 27 | 204.4 | European flat oyster, *Ostrea edulis* | 204.4 | - |
| 26 | 83.28 | Shore crab, *Hemigrapsus nudus* | 83.28 | - |
| 25 | 34.90 | Mud crab, *Rhithropanopeus harrisii* | 34.90 | - |
| 24 | 24.90 | Grass shrimp, *Palaemonetes pugio* | 20 | - |
| | | Grass shrimp, *Palaemonetes* sp. | >31 | - |
| 23 | 21.34 | Mummichog, *Fundulus heteroclitus* | 21.34 | - |
| 22 | 18.0 | Amphipod, *Rhepoxynius abronius* | 18.0 | - |
| 21 | >14.60 | Amphipod, *Orchestia traskiana* | >14.60 | - |
| 20 | <10 | Amphioxus *Branchiostoma caribaeum* | <10 | - |
| 19 | 9.732 | Shore crab, *Carcinus maenas* | 9.732 | - |
| 18 | 6.812 | Polychaete, *Neanthes arenacedentata* | 6.812 | - |
| 17 | 9.037 | Sheepshead minnow, *Cyprinodon variegatus* | 9.037 | - |

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] |
|---|---|---|---|---|
| 16 | 5.167 | Inland silverside, *Menidia beryllina* | 3.0 | - |
| | | Atlantic silverside, *Menidia menidia* | 8.9 | - |
| 15 | ~5.0 | Lugworm, *Arenicola cristata* | ~5.0 | - |
| 14 | 4.944 | Atlantic manhaden, *Brevoortia tyrannus* | 4.944 | - |
| 13 | 2.238 | Blue mussel, *Mytilus edulis* | 2.238 | - |
| 12 | 1.975 | Copepod, *Eurytemora affinis* | 1.975 | 27.24[e] |
| 11 | 1.911 | Copepod, *Nitocra spinipes* | 1.911 | - |
| 10 | 1.745 | American lobster, *Homarus americanus* | 1.745 | - |
| 9 | 1.73 | Amphipod, *Eohaustorius estuarius* | 1.73 | - |
| 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 | - |
| 2 | <0.9732 | Mysid, *Metamysidopsis elongata* | <0.9732[d] | - |

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] |
|---|---|---|---|---|
| 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.
[e] See text for justification of this value.

Fresh Water

Final Acute Value = 0.9177 µg/L

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

Final Acute-Chronic Ratio = 14.69 (see text)

Final Chronic Value = (0.9177 µg/L)/14.69 = 0.0625 µg/L

Salt Water

Final Acute Value = 0.7347 µg/L

Criterion Maximum Concentration = (0.7347 µg/L)/2 = 0.3674 µg/L

Final Acute-Chronic Ratio = 14.69 (see text)

Final Chronic Value = (0.7347 µg/L)/14.69 = 0.0500 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 ($\mu g/L$)[b] | Reference |
|---|---|---|---|---|---|---|
| **FRESHWATER SPECIES** | | | | | | |
| Alga, *Bumilleriopsis filiformis* | TBTC | - | 14 | No growth | 111.4 | Blanck 1986; Blanck et al. 1984 |
| Alga, *Klebsormidium marinum* | TBTC | - | 14 | No growth | 222.8 | Blanck 1986; Blanck et al. 1984 |
| Alga, *Monodus subterraneus* | TBTC | - | 14 | No growth | 1,782.2 | Blanck 1986; Blanck et al. 1984 |
| Alga, *Raphidonema longiseta* | TBTC | - | 14 | No growth | 56.1 | Blanck 1986; Blanck et al. 1984 |
| Alga, *Tribonema aequale* | TBTC | - | 14 | No growth | 111.4 | Blanck 1986; Blanck et al. 1984 |
| Blue-green alga, *Oscillatoria* sp. | TBTC | - | 14 | No growth | 222.8 | Blanck 1986; Blanck et al. 1984 |
| Blue-green alga, *Synechococcus leopoliensis* | TBTC | - | 14 | No growth | 111.4 | Blanck 1986; Blanck et al. 1984 |
| Green alga, *Chlamydomonas dysosmas* | TBTC | - | 14 | No growth | 111.4 | Blanck 1986; Blanck et al. 1984 |
| Green alga, *Chlorella emersonii* | TBTC | - | 14 | No growth | 445.5 | Blanck 1986; Blanck et al. 1984 |
| Green alga, *Kirchneriella contorta* | TBTC | - | 14 | No growth | 111.4 | Blanck 1986; Blanck et al. 1984 |
| Green alga, *Monoraphidium pusillum* | TBTC | - | 14 | No growth | 111.4 | Blanck 1986; Blanck et al. 1984 |
| Green alga, *Scenedesmus obtusiusculus* | TBTC | - | 14 | No growth | 445.5 | Blanck 1986; Blanck et al. 1984 |
| Green alga, *Scenedesmus quadricauda* | TBTC | - | 12 | Reduced growth (87.6%) | 1 | Fargasova and Kizlink 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<br>87.6%<br>95.9%<br>100% | <br>1<br>10<br>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* | TBTC | - | 14 | No growth | 111.4 | Blanck 1986; Blanck et al. 1984 |
| Green alga, *Selenastrum capricornutum* | TBTC | - | 4 | EC50 | 12.4 | Miana et al. 1993 |
| | | | SALTWATER SPECIES | | | |
| Diatom, *Skeletonema costatum* | TBTO | - | 5 | Algistatic<br>algicidal | 0.9732-17.52<br>>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] TBTC = 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 | 180,427 | 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 | 1,179 | Martin et al. 1989 |
| | | | | | Gall bladder/bile | 331 | |
| | | | | | Kidney | 2,242 | |
| | | | | | Carcass | 1,345 | |
| | | | | | Peritoneal fat | 5,419 | |
| | | | | | Gill | 1,014 | |
| | | | | | Blood | 653 | |
| | | | | | Gut | 487 | |
| | | | | | Muscle | 312 | |
| 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)<br>1.6 (pH = 6.8)<br>1.7 (pH = 7.8) | 14 | Whole body | ~1190<br>~1523<br>~2250 | Tsuda et al. 1990a |
| Goldfish (3.5-4.0 cm; 1.6-2.9 g); *Carassius auratus* | TBTC | 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 (female), *Nucella lapillus* | TBT | - | 0.0038 to 0.268 | 249 to 408 | Soft parts | [illegible],000 to 38,000 | Bryan et al. 1987a |
| Snail (female), *Nucella lapillus* | Field | - | 0.070 | 529 to 634 | Soft parts | 17,000 | Bryan et al. 1987a |
| Snail (18-22 mm), *Nucella lapillus* | TBTC | 35 | 0.0205 | 49 | Soft parts | 30,000 | Bryan et al. 1989b |
| Blue mussel (spat), *Mytilus edulis* | d | 28.5-34.2 | 0.24 | 45 | Soft parts | 6,833[e] | 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* | d | - | 0.452<br>0.204<br>0.204<br>0.079 | 56 | Soft parts | 23,000<br>27,000<br>10,400<br>37,500 | Salazar et al. 1987 |
| Blue mussel (juvenile), *Mytilus edulis* | Field | - | <0.105 | 84 | Soft parts | 5,000-60,000 | Salazar and Salazar, In press |
| Blue mussel (3.0 - 3.5 cm), *Mytilus edulis* | TBTC | 25.1-26.3 | 0.020 | 60 | Muscle and mantle<br>Muscle and mantle | 7,700<br>11,000 | Guolan and Yong 1995 |
| Pacific oyster, *Crassostrea gigas* | TBTO | 28-31.5 | 1.216 | 21 | Soft parts | 1,874[e] | Waldock et al. 1983 |
| Pacific oyster, *Crassostrea gigas* | TBTO | 28-31.5 | 0.1460 | 21 | Soft parts | 6,047[e] | Waldock et al. 1983 |
| Pacific oyster, *Crassostrea gigas* | d | 28.5-34.2 | 0.24 | 45 | Soft parts | 7,292[e] | Thain and Waldock 1985; Thain 1986 |
| Pacific oyster, *Crassostrea gigas* | TBTO | 29-32 | 1.557 | 56 | Soft parts | 2,300 | Waldock and Thain 1983 |

Table 5. (Continued)

| Species | Chemical[a] | Salinity (g/kg) | Concentration in Water (µg/L)[b] | Duration (days) | Tissue | BCF or BAF[c] | Reference |
|---|---|---|---|---|---|---|---|
| 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<br>0.92<br>2.83 | 30 | Soft parts | 2275<br>1369<br>621 | Osada et al. 1993 |
| European flat oyster, *Ostrea edulis* | TBTO | 28-31.5 | 1.216 | 21 | Soft parts | 960[e] | Waldock et al. 1983 |
| European flat oyster, *Ostrea edulis* | TBTO | 28-34.2 | 0.24 | 75 | Soft parts | 875[e] | Waldock et al. 1983 |
| European flat oyster, *Ostrea edulis* | TBTO | 28-34.2 | 2.62 | 75 | Soft parts | 397[e] | Thain 1986 |
| European flat oyster, *Ostrea edulis* | d | 28.5-34.2 | 0.24 | 45 | Soft parts | 1,167[e] | Thain and Waldock 1985; Thain 1986 |
| European flat oyster, *Ostrea edulis* | d | 28.5-34.2 | 2.62 | 45 | Soft parts | 192.3[e] | 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] Test organisms were exposed to leachate from panels coated with antifouling paint containing tributyltin.
[e] 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)<br>Metabolism returned to normal (14.1 days) | 4.7 | Miniero and Delupis 1991 |
| | | | | Metabolism reduced (1 day)<br>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<br>(production)<br>(reproduction) | 20<br>5 | Wong et al. 1982 |
| Green alga, *Ankistrodesmus falcatus* | TBTO (97%) | - | 7 days<br>14 days<br>21 days<br>28 days | BCF = 300<br>BCF = 253<br>BCF = 448<br>BCF = 467 | 5.2<br>4.7<br>2.1<br>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* | TBTC | - | 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* | TBTC | - | 8 days | Altered phototaxis | 0.45 | Meador 1986 |
| Cladoceran (14-d-old), *Daphnia magna* | TBTC | 150 | 7 days | Altered behavior | 1 | Bodar et al. 1990 |
| | | | | Reproductive failure | 1 | |
| | | | | Digestive storage cells reduced | 5 | |
| Cladoceran (<24-h old), *Daphnia magna* | TBTC | 312.8 | 48 hr | EC50 (mobility) | 9.8 | Miana et al. 1993 |
| Fairy shrimp (cysts), *Streptocephalus texanus* | TBTC | 250 | 24 hr | EC50 (hatching) | 15 | Crisinel et al. 1994 |
| Rainbow trout (yearling), *Oncorhynchus mykiss* | TBTO | - | 24 hr<br>48 hr | LC50 | 25.2<br>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* | TBTC | 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* | TBTC | 96-105 | 110 days | NOEC (mortality and growth)<br>LOEC (mortality and growth) | 0.040<br>0.200 | de Vries et al. 1991 |

Table 6. (Continued)

| Species | Chemical[a] | Hardness (mg/L as $CaCO_3$) | Duration | Effect | Concentration ($\mu$g/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 | NOEC | 1.0 | Wester and Canton 1991 |
| | | | 3 mo | NOEC | 0.32 | |
| Frog (embryo, larva), *Rana temporaria* | TBTO | - | 5 days | LC40 | 28.4 | Laughlin and Linden 1982 |
| | TBTF | - | | LC50 | 28.2 | |
| | TBTO | - | | Loss of body water | 28.4 | |
| | TBTF | - | | Loss of body water | 28.2 | |

Table 6. (Continued)

| Species | Chemical[a] | Salinity (g/kg) | Duration | Effect | Concentration (µg/L)[b] | Reference |
|---|---|---|---|---|---|---|
| | | | | SALTWATER SPECIES | | |
| Natural microbial populations | TBTC | 2 and 17 | 1 hr | Significant decrease in metabolism of nutrient substrates | 4.454 | Jonas et al. 1984 |
| Natural microbial populations | TBTC | 2 and 17 | 1 hr (incubated 10 days) | 50% mortality | 89.07 | Jonas et al. 1984 |
| Microcosm (seagrass bed) | TBT (>95%) | 21.5-28.9 | 6 wks | Fate of TBT<br>Sediments 81-88%<br>Plants 9-17%<br>Animals 2-4% | 0.2-20 | Levine et al. 1990 |
| Microcosm (seagrass bed) | TBTC | - | 6 wks | Reduced plant material loss:<br>loss of amphipod *Cymadusa compta* | 22.21 | Kelly et al. 1990a |
| Periphyton communities | TBTC | - | 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 |
| Diatom, *Phaeodoctylum tricornutum* | TBTO | - | 72 hr | No effect on growth | 1.460-5.839 | Salazar 1985 |

Table 6. (Continued)

| Species | Chemical[a] | Salinity (g/kg) | Duration | Effect | Concentration (µg/L)[b] | Reference |
|---|---|---|---|---|---|---|
| Diatom, *Nitzschia* sp. | TBTO | - | 8 days | EC50 | 1.19 | Dojmi et al. 1987 |
| 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* | TBTC | 30 | 72 hr | EC50 (population growth) | 0.3207 | Walsh et al. 1985;1987 |
| Diatom, *Skeletonema costatum* | TBTC | 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* | TBTC | 30.5 | 96 hr | NOEC | 1 | Reader and Pelletier 1992 |
| Diatom, *Minutocellus polymorphus* | TBTO | - | 48 hr | EC50 | ~340 | Walsh et al. 1988 |
| Diatom, *Minutocellus polymorphus* | TCTC | - | 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 |

Table 6. (Continued)

| Species | Chemical[a] | Salinity (g/kg) | Duration | Effect | Concentration (µg/L)[b] | Reference |
|---|---|---|---|---|---|---|
| Alga, _Pavlova lutheri_ | TBTO | 34-40 | 12-26 days | Population growth | 1.0 | Beaumont and Newman 1986 |
| Alga, _Pavlova lutheri_ | TBTO | - | 16 days | NOEC<br>LOEC | 5.36<br>21.46 | St. 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)<br>EC50 (growth) | 11.256<br>13.629 | Brix et al. 1994a |
| Polychaete worm (juvenile), _Neanthes arenaceodentata_ | TBTC (96%) | 30 | 10 wks | NOEC (survival)<br>LOEC (survival) | 0.100<br>0.500 | Moore et al. 1991 |
| Rotifer (neonates), _Brachionus plicatilis_ | TBT | 15 | 30 min | Induction of the stress protein gene SP58 | 20-30 | Cochrane et al. 1991 |
| Hydroid, _Campanularia flexuosa_ | TBTF | 35 | 11 days | Colony growth stimulation; no growth at 1.0 µg/L | 0.01 | Stebbing 1981 |
| Sand dollar (sperm), _Dendraster excentricus_ | TBT | 32-33 | 80 min | EC50 (mortality) | 0.465 | Brix et al. 1994b |
| Starfish (79 g), _Leptasterias polaris_ | TBTC | 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_ | TBTC | 35 | 6 months | Imposex induced | ≥0.012 | Stroben et al. 1992b |
| Dogwhinkle (subadult), _Nucella lapillus_ | TBTC | 35 | 22 | BCF = ~20,000 | 0.019 | Bryan et al. 1993 |
| Blue mussel (larva), _Mytilus edulis_ | TBTO | - | 24 hr | No effect on sister chromatid exchange | 1.0 | Dixon and Prosser 1986 |

Table 6. (Continued)

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

Table 6. (Continued)

| Species | Chemical[a] | Salinity (g/kg) | Duration | Effect | Concentration ($\mu g/L$)[b] | Reference |
|---|---|---|---|---|---|---|
| Blue mussel (juveniles and adults), _Mytilus_ sp. | TBT (field) | - | 84 days | BCF | 3,000-100,000 | Salazar 1992 |
| Blue mussel (3.0-3.5 cm), _Mytilus edulis_ | TBT | - | 2 days | Reduced ability to survive anoxia | 1 | Wang et al. 1992 |
| Blue mussel (4 cm), _Mytilus edulis_ | TBTC (>97%) | - | 2.5 days | Increased respiration 0.15 µg/g tissue<br>Reduced food absorption efficiency 10 µg/g | - | Widdows and Page 1993 |
| Blue mussel (8-d-old larvae), _Mytilus edulis_ | TBT | - | 33 days | NOEC (growth)<br>LOEC (growth) | 0.006<br>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 |

Table 6. (Continued)

| Species | Chemical[a] | Salinity (g/kg) | Duration | Effect | Concentration (µg/L)[b] | Reference |
|---|---|---|---|---|---|---|
| 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 |
| Pacific oyster (larva), *Crassostrea gigas* | TBTF | 18-21 | 21 days | Reduced number of normally developed larvae | 0.02346 | Springborn Bionomics 1984a |
| Pacific oyster (larva), *Crassostrea gigas* | TBTF | 18-21 | 15 days | 100% mortality | 0.04692 | Springborn Bionomics 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<br>Normal growth rate | 0.040<br>0.010 | Stephanson 1991 |
| Pacific oyster (spat), *Crassostrea gigas* | TBTO | 35 | 4 wks | 79% reduced growth | 0.005 | Nell and Chvojka 1992 |
| Pacific oyster (fertilized eggs), *Crassostrea gigas* | TBTO | - | 24 hr | LC50<br>Delayed development | 7.0<br>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 |

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* | TBTC | 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 ($\mu g/L$)[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 | $\geq$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; $\geq$1.0 $\mu g/L$ 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* | TBTC | 18-22 | 48 hr | Delayed development | 0.77 | Roberts 1987 |
| Common Pacific Littleneck (adult), *Protothaca stamina* | TBTO | 33-34 | 96 hr | 100% survival | $\geq$2.920 | Salazar and Salazar, Manuscript |
| 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 (nauplii), *Acartia tonsa* | TBTC | 10-12 | 9 days | Reduced survival | ≥0.029 | Bushong et al. 1990 |
| Copepod (nauplii), *Acartia tonsa* | TBTC | 10-12 | 6 days | Reduced survival; no effect 0.012 µg/L | 0.023 | Bushong et al. 1990 |
| Copepod (nauplii), *Acartia tonsa* | TBTC | 10-12 | 6 days | Reduced survival; no effect 0.010 µg/L | 0.024 | Bushong et al. 1990 |
| Copepod (adult), *Acartia tonsa* | TBTO | 28 | 5 days | Reduced egg production | 0.010 | Johansen and Mohlenberg 1987 |
| 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. | TBTC | 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 |
| 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 |

Table 6. (Continued)

| Species | Chemical[a] | Salinity (g/kg) | Duration | Effect | Concentration ($\mu g/L$)[b] | Reference |
|---|---|---|---|---|---|---|
| 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), FL *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 regneration and molting | 0.5 | Weis et al. 1987a |
| Fiddler crab, *Uca pugilator* | TBTO | 25 | 3 weeks | Reduced burrowing | 0.5 | Weis and Perlmutter 1987 |
| 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* | TBTC | 10 | 28 days | No effect | 0.490 | Hall et al. 1988b |

Table 6. (Continued)

| Species | Chemical[a] | Salinity (g/kg) | Duration | Effect | Concentration (µg/L)[b] | Reference |
|---|---|---|---|---|---|---|
| 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. 1989 |
| Mummichog, *Fundulus heteroclitus* | TBTO | 9.9-11.2 | 20 min | Avoidance | 3.7 | Pinkney et al. 1985 |
| Inland silverside (larva), *Menidia beryllina* | TBTC | 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<br>16-19.5 | 96 hr<br>6 wks | LC50<br>NOEC | 17.2<br>2.000 | Pinkney et al. 1989 |
| Three-spined stickleback (45-60 mm), *Gasterosteus aculeatus* | TBTO (painted panels) | 15-35 | 7.5 mo | 80% mortality (2 months)<br>Histological effects | 10<br>2.5 | Holm et al. 1991 |
| 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 |

Table 6. (Continued)

| Species | Chemical[a] | Salinity (g/kg) | Duration | Effect | Concentration (µg/L)[b] | Reference |
|---|---|---|---|---|---|---|
| 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. 1989 |
| Striped bass (juvenile), Morone saxatilis | TBT (painted panels) | 1.1-3.0<br>1.9-3.0<br>12.2-14.5 | 6 days<br>7 days<br>7 days | NOEC 0.067; LOEC 0.766<br>NOEC 0.444; LOEC 1.498<br>LOEC >0.514 | - | Pinkney et al. 1990 |
| Speckled sanddab (adult), Citharichthys stigmaeus | TBTO | 33-34 | 96 hr | LC50 | 18.5 | Salazar and Salazar, Manuscript |
| Stripped mullet (3.2 g); Mugil cephalus | TBTO (96%) | - | 8 wks | BCF 3000 (no plateau)<br>BCF 3600 (no plateau) | 0.122<br>0.106 | Yamada and Takayanagi 1992 |
| 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 |

[a] TBTA = tributyltin acetate; TBTC = 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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Table 6. (Continued)

| Species | Chemical[a] | Salinity (g/kg) | Duration | Effect | Concentration ($\mu$g/L)[b] | Reference |
|---|---|---|---|---|---|---|
| California grunion (embryo), <u>Leuresthes</u> <u>tenuis</u> | c | - | 10 days | No adverse effect on hatching success or growth | 0.14-1.72 | Newton et al. 1985 |
| California grunion (larva), <u>Leuresthes</u> <u>tenuis</u> | c | - | 7 days | Survival increased as concentration increased | 0.14-1.72 | Newton et al. 1985 |
| Striped bass (juvenile), <u>Morone</u> <u>saxatilis</u> | TBTO (95%) | 9-11 | - | Avoidance | 24.9 | Hall et al. 1984 |
| Striped bass (juvenile), <u>Morone</u> <u>saxatilis</u> | TBT (painted panels) | 13.0-15.0 | 14 | NOEC (serum ion concentrations and enzyme activity) | 1.09 | Pinkney et al. 1989 |
| Striped bass (juvenile), <u>Morone</u> <u>saxatilis</u> | TBT (painted panels) | 1.1-3.0<br>1.9-3.0<br>12.2-14.5 | 6 days<br>7 days<br>7 days | NOEC 0.067; LOEC 0.766<br>NOEC 0.444; LOEC 1.498<br>LOEC >0.514 | - | Pinkney et al. 1990 |
| Speckled sanddab (adult), <u>Citharichthys</u> <u>stigmaeus</u> | TBTO | 33-34 | 96 hr | LC50 | 18.5 | Salazar and Salazar, Manuscript |
| Stripped mullet (3.2 g); <u>Mugil</u> <u>cephalus</u> | TBTO (96%) | - | 8 wks | BCF 3000 (no plateau)<br>BCF 3600 (no plateau) | 0.122<br>0.106 | Yamada and Takayanagi 1992 |
| Fouling communities | c | 33-36 | 2 months | Reduced species and diversity; no effect at 0.04 $\mu$g/L | 0.1 | Henderson 1986 |
| Fouling communities | c | - | 126 days | No effect | 0.204 | Salazar et al. 1987 |

[a] TBTA = tributyltin acetate; TBTC = 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.

Table 6. (Continued)

| Species | Chemical[a] | Salinity (g/kg) | Duration | Effect | Concentration (µg/L)[b] | Reference |
|---|---|---|---|---|---|---|
| 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. 1989 |
| Mummichog, Fundulus heteroclitus | TBTO | 9.9-11.2 | 20 min | Avoidance | 3.7 | Pinkney et al. 1985 |
| Inland silverside (larva), Menidia beryllina | TBTC | 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<br>16-19.5 | 96 hr<br>6 wks | LC50<br>NOEC | 17.2<br>2.000 | Pinkney et al. 1989 |
| Three-spined stickleback (45-60 mm), Gasterosteus aculeatus | TBTO (painted panels) | 15-35 | 7.5 mo | 80% mortality (2 months)<br>Histological effects | 10<br>2.5 | Holm et al. 1991 |
| 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 |