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A review of organotin regulatory strategies, pending actions, related costs and benefits

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Abstract

Achieving consensus on equitable and effective national and global regulation(s) for the use of organotins as biocides in antifouling boat bottom paints has proven to be very complex and difficult for a variety of reasons as discussed in this paper. There appears to be broad agreement among stakeholders about the effectiveness of tributyltin (TBT) in antifouling paints. A draft Assembly Resolution prepared by the Marine Environmental Protection Committee (MEPC) of the International Maritime Organization (IMO) to propose a global ban on the use of organotins in antifouling paints was approved by the IMO at its 21st regular session (November 1999). In approving the Resolution, the Assembly agreed that a legally binding instrument (global convention — an international treaty) be developed by the Marine Environmental Protection Committee that should ensure by 1 January 2003, a ban on the application of tributyltin (TBT)-based antifouling paints; and 1 January 2008 as the last date for having TBT-based antifouling paint on a vessel. The Assembly also agreed that a diplomatic conference be held in 2001 to consider adoption of the international legal instrument. Monitoring, policing, enforcement, fines and record-keeping are yet to be defined. In addition, the MEPC has also proposed that IMO promotes the use of environmentally-safe anti-fouling technologies to replace TBT. Existing national regulations in the US and Europe have: (1) restricted the use of TBT in antifouling boat bottom paints by vessel size (less than 25 m in length), thus eliminating TBT from the smaller and recreational vessels that exist in shallow coastal waters where the impacted oysters species grow; (2) restricted the release rates of TBT from co-polymer paints; and (3) eliminated the use of free TBT in paints. The present movement toward a global ban suggests that the above regulatory approach has *not* been sufficient in some countries. Advocates of the ban cite international findings of: (1) higher levels of TBT in surface waters of ports and open waters; (2) imposex still occurring and affecting a larger number of snail species; (3) TBT bioaccumulation in selected fisheries; and (4) the availability of 'comparable' alternatives (to TBT) with less environmental impact. The global ban has been absent of a policy debate on the: (1) lack of 'acceptable and approved' alternatives in many nations; (2) appreciation of market forces in nations without TBT regulations; (3) full consideration of the economic benefits from the use of TBT; (4) 'acceptance' of environmental impacts in marinas, ports and harbors; and (5) realization of the 'real' time period required by ships for antifoulant protection (is 5–7 years necessary or desirable?). Estimates of fuel savings range from $500 million to one billion. In assessing the

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environmental impact from TBT, there are two sources: the shipyard painting vessels and the painted vessel itself. Today vessels can be painted with regulated or banned antifouling materials by boatyards in a country that does not have TBT regulations and subsequently travel in international and regulated national waters and thus bringing the impact back to the country which was trying to prevent it. Worse, local and national regulations for TBT have proven to be the antithesis of the popular environmental cliché — 'Think Globally and Act Locally.' Legislative policies enacted by 'regulated' countries to regulate the use of TBT to protect (their) *local* marine resources have subsequently had far reaching environmental and economic impacts which have in essence transferred TBT contamination to those countries least able to deal with it. Market forces are selective for cheap labor and cheap environments. 'Unregulated' countries have unknowingly accepted the environmental and human health risks to gain the economic benefits from painting TBT on ships. Unfortunately, these countries may not have the funding or environmental expertise available for the monitoring, research and technology development essential to use these modern high technology compounds. Therefore, they end up with more contamination because they do not have the necessary regulatory structure to prevent it. In the US coastal zone, federal and state regulations have had a significant impact on reducing TBT levels, generally to well below the provisional water quality standard of 10 ng/l, and in bivalve tissues. Current environmental and marine and estuarine water concentrations are well below predicted acute TBT toxicity levels. Estimation of chronic toxicity effects using mean water TBT concentrations indicate that current levels would be protective of 95% of species. Analysis of allowable daily intake/oral reference dose values from market basket surveys and the NOAA National Status and Trends data suggest that there is no significant human health risk from consuming seafood contaminated with TBT. Most of the data that exceeded these values were from areas of high TBT input from ports, harbors and marinas (commercial shipping, shipyards and drydock facilities) and sites of previous contamination. In the US, at this time, TBT environmental data and lack of acceptable alternatives does *not* justify a global ban for TBT. Three significant aspects of the regulatory discussion should *not* be forgotten: (1) none of the available alternatives to TBT-based antifouling paints has been approved on a global basis or in the US by the USEPA, the VOC levels are above current regulatory levels and in the past such reviews have taken up to 54 months to complete; (2) studies in Ireland have found that the use of TBT has greatly reduced the threat and risk of introduction of invasive (exotic) marine species in foreign waters; and (3) a biofouled ship can transport on its bottom approximately 2 000 000 marine organisms which is significant when compared to the small numbers transported in ballast waters. Alternatives to TBT are available, but not proven and accepted on a global basis. Unfortunately in the less than 1000 days remaining before the proposed IMO ban, an international independent process is not available to expedite the IMO recommendation to evaluate and select alternatives to TBT. The cost (to shipowners) for this failure has been estimated to range from $500 million–$1 billion annually. A third party, neutral, independent, international Marine Coatings Board has been proposed to supplement the national regulatory process by providing the international standardized scientific data and information of the highest quality. The cost of the Marine Coating Board to evaluate available alternatives has been estimated to be $10 million/year or 1–2% of the estimated annual direct costs to shipowners of *not* having comparable antifouling marine coating alternatives to TBT. In ship operating coasts, this is less than $1/day per vessel in global commerce with a total ROI in the first 37 days of 2008.

Keywords: Tributyltin; Biofouling; Antifouling; Shipping; Fuel savings; Marine coating; Regulation; Policy; Environmental benefits; Economic benefits; Marine R&D; Toxicity; Invasive organisms; Ballast waters; The US Antifouling Paint Control Act of 1988; International Marine Organisation; Marine Environmental Protection Committee; International conventions; Imposex; *Nucella lapillus*; *Crassostrea gigas*; Marine Coating Board

1. Introduction

The regulatory policies and practices of developed countries on the use of organotin compounds as biocides in antifouling boat bottom paints have been extensively reviewed by Abel (1996), Bosselmann (1996), Champ and Wade (1996), Stewart (1996). These reviews discuss actions by the United Kingdom, United States, France, Switzerland, Germany, Japan, and other nations, as well as by the Commission of European Communities (CEC), and international conventions which govern the use of organotin compounds for biocides in antifouling boat bottom paints. After the implementation of national restrictions by the above countries in the mid to late 1980s, there has been about a decade of assessing the effectiveness of the initial regulations without subsequent additional or more restrictive regulations. Austria and Switzerland banned the use of TBT even though they are land locked. Japan banned the use of TBT in 1990. New Zealand's restriction of the use of TBT-based antifoulant paints in 1989 increased the use and marketing of copper-based marine coatings in the south Pacific as alternatives to TBT as being 'environmentally friendly' (de Mora, 1996a).

The purpose of this paper is to review the impact of regulatory strategies, policies, economic and environmental costs and benefits from the use of TBT. The impacts are related to the effectiveness of regulations in reducing local environmental contamination, as well as preventing the shift of organotin-related environmental hazards with subsequent economic loss of shipyard business to non-regulated countries; and to estimate the impacts on the shipping industry from increased fuel and operating costs. This paper is a synthesis of data, information and perspectives from many sources. It is a summary of documents from MEPC/IMO sessions (30–43) and reflects an overview of regulatory actions to date. It also includes a preliminary review of the science used to make regulatory decisions and recommends the creation of a Marine Coatings Board to conduct an independent international calibration of available alternative non-TBT marine coatings. This would provide environmental, economic, and operational data and information to support the regulatory process and the marketplace in selecting future marine antifoulants.

2. Organotin regulatory strategies

The first use of organotin-based antifouling boat bottom paints began in the early 1970s. In 1974, oyster growers first reported the occurrence of abnormal shell growth in *Crassostrea gigas*, the pacific oyster along the east coast of England (Key et al., 1976). However, it was not until the mid 1980s, that researchers in France and the United Kingdom began to suggest that the use of TBT in antifouling paints was adversely impacting a number of marine species other than the fouling organisms. This economically important species is *Crassostrea gigas*, the Pacific oyster, which is farmed in coastal waters of England and France (Waldock, 1986; Waldock et al., 1987a,b; Thain et al., 1987; Alzieu, 1991; His, 1996 and references therein). Since the Pacific oyster is from Japan, in the UK, France, and the US, it is an 'exotic' — non-native — (foreign) invasive species whose cultivation and growth outside of Japan, is at the displacement of native species. It is the only species of oyster that has been found to demonstrate abnormal growth from exposure to TBT. The difficulty in delineating cause-and-effect relationships and the effects on untargeted species attracted international concern (Stebbing, 1985, 1996). See Champ and Seligman (1996a), de Mora (1996a) for an overview of organotin environmental fate and effects and the updated literature cited in this paper. See Milne (1993) for a review of the history of the development and chemistry of self polishing antifoulings.

At the 6th International Ocean Disposal Symposium (21–25 April 1986) held at the Assilomar Conference Center in Pacific Grove, California, Edward D. Goldberg, of the Scripps Oceanographic Institute, the keynote speaker, pointed out that 'TBT was perhaps the most toxic substance ever deliberately introduced to the marine environment by mankind' (Goldberg, keynote address, unpublished manuscript, also see Goldberg, 1986).

The following sections present summaries of regulatory strategies developed by nations and international regulatory bodies in response to regulating the use of organotin compounds and in particular tributyltin (TBT) as a biocide in antifouling marine coatings. Fig. 1 is a map of the world with an overlay of the legislative position on antifoulings by country (courtesy of International Paint, reproduced from http://www.international-marine.com/).

2.1. United States

In the US, regulatory actions for TBT stem

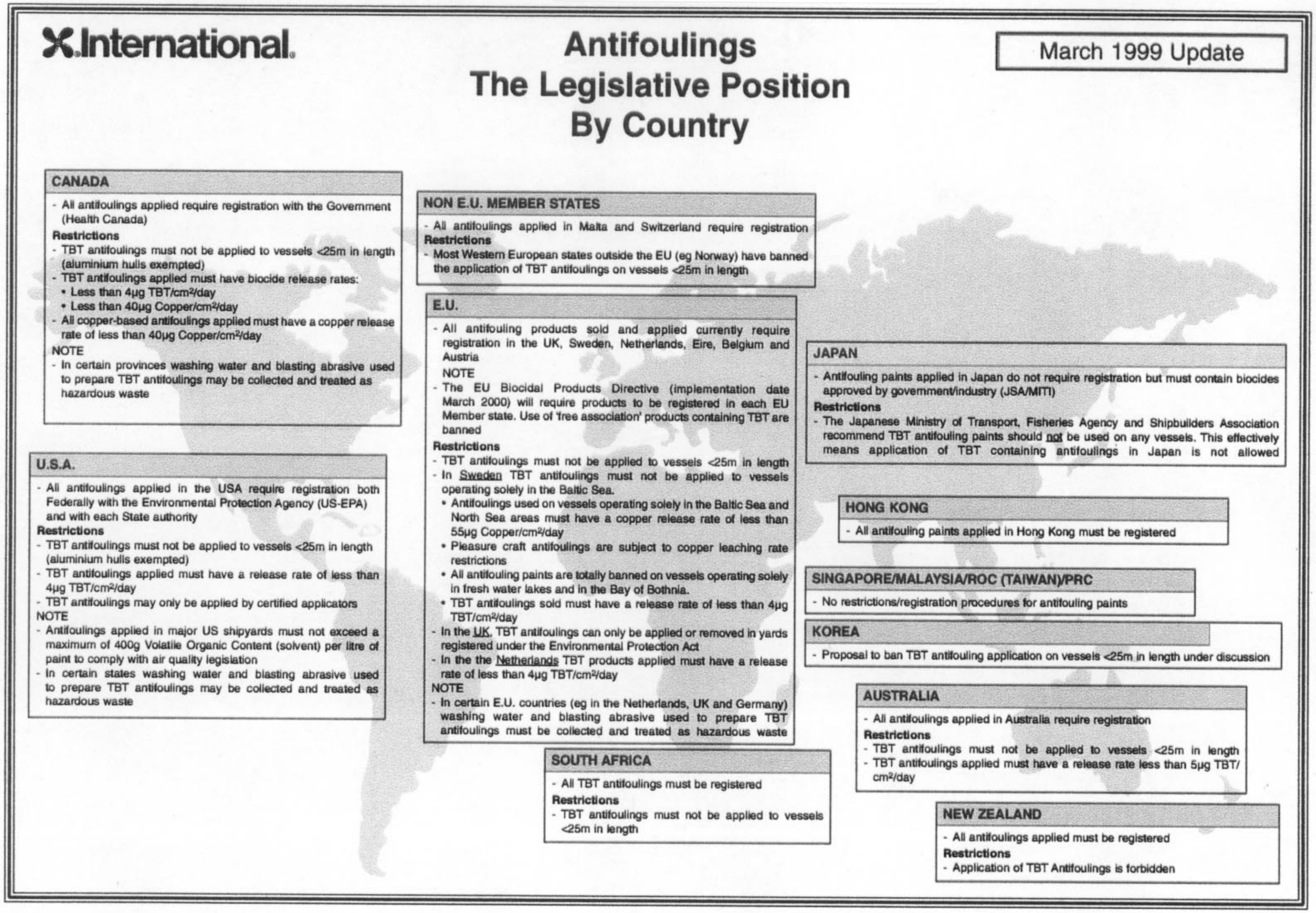

Fig. 1. The legislative position on antifoulings by country.

from the US Navy's issuing an environmental assessment: Fleetwide Use of Organotin Antifouling Paint (NAVSEA, 1984). The Navy assessment addressed the British (MAFF) studies by Thain (1983), Waldock and Miller (1983), Waldock et al. (1983), Alzieu and Portman (1984), and the French studies by Alzieu et al. (1981–1982, 1986), His and Robert (1983–1985) on the impact of TBT from free association paints on Pacific oysters in Europe (Champ, 1986).

The Navy's conclusion (US Navy, 1986) in their recommendation for the use of TBT was based on the conclusion that the impacts found in Europe were related to excessive use of free-association TBT-based paints on *small* recreation boats, which were primarily used in shallow coastal estuarine waters where oysters were grown (Champ and Lowenstein, 1987; Champ and Pugh, 1987). The US Navy's decision was based on the following: (1) navy ships were mostly seagoing vessels and spent only minimal periods of time in harbors or shallow coastal waters; (2) the Navy was proposing to use TBT-based co-polymer paints with low release rates, so that the impact on non-target organisms would be very limited; and (3) the cost benefits from the use of these coatings was estimated to be from $100 to $130 million annually in fuel avoidance (savings) costs and millions in annual maintenance costs (NAVSEA, 1984, 1986; Bailey, 1986; Eastin, 1987; Ricketts, 1987; Schatzberg, 1987).

The determination of the US Navy to utilize TBT-based antifouling paints was probably a valid scientific conclusion based upon the three reasons given above. Nevertheless, Virginia Senator Trible was concerned that the Navy would be able to use TBT and it would impact oysters in the lower Chesapeake Bay. He inserted language in the 1986 Report of the Continuing Resolution for the FY86's Federal Budget requiring the Navy to have approval from the EPA to use TBT (Champ and Wade, 1996). The Navy did not fully appreciate the sensitivity of the TBT issue in coastal states and to members of Congress, and that EPA's regulatory process could prevent them from using TBT. The Navy's position was based on advice from its legislative advisors in Congress, that the EPA would be able to easily give the Navy a variance for the use of TBT, due to the reasons referred to earlier. To the Navy's surprise, it subsequently learned that once EPA initiated the special review process that the EPA would not be able to make any decisions until the process was completed. This could take years.

The US Environmental Protection Agency (EPA) on 8 January 1986, announced the initiation of a special review of all registered pesticide products containing TBT compounds used as biocides in antifouling paints. By mid-1986, TBT had joined the list of the 'Chemical of the Month' at the EPA.

At about the same time, US academic researchers and state water quality boards or state natural resource agencies in a few key coastal states (Virginia, North Carolina, California, Oregon, and Washington) began to closely follow the organotin issues in France and the UK, and investigated their coastal waters for similar effects. The only published paper finding deformed oysters in the US is by Wolniakowski et al. (1987) for a specimen found in Coos Bay, Oregon. For additional information on TBT concentrations in Chesapeake Bay, and other US and Canadian waters, see papers published in the Proceedings of the International Organotin Symposium of the Oceans 1986 Conference (1986); Proceedings of the International Organotin Symposium of Oceans 1987 Conference (1987); Proceedings of the National Organotin Symposium of the Oceans 1988 Conference (1988); Proceedings of the National Organotin Symposium of the Oceans 1989 Conference (1989) and Proceedings of the Third International Organotin Symposia, 1990 Conference (1990). In addition, see the following citations and references cited therein: Maguire (1984, 1987, 1991, 1996a,b, 1998), Maguire and Hale (1981), Maguire et al. (1982, 1985, 1986), Grovhoug et al. (1996), de Mora and Pelletier (1997), Seligman et al. (1996a) and Seligman et al., (1996b) and US EPA (1987).

2.2. Commonwealth of Virginia (state of Virginia)

By mid-1987, most coastal states were planning or had implemented restrictions on the use of organotins. Virginia was among the first to become concerned and implemented a regulatory strategy developed by the Virginia Water Control Board after which the subsequent US federal law was modelled. One of the areas that states could regulate organotin usage was linked to the ambient water quality concentrations of organotins through state environmental quality standards. Studies initiated in 1984 by Huggett and others at the Virginia Institute of Marine Science for the lower Chesapeake Bay and were finding high concentrations of TBT near drydocks and shipyards. See: Huggett et al. (1992, 1996) and references cited therein.

During 1986, a series of excellent articles were written by Bruce Reid (Reid, 1986) and published by the Daily Press/The Times-Herald, a local newspaper in Newport News, Virginia. Reid was the first in the US to report on the impact of TBT on oysters in France and the UK and on the dangers and public health risk to yard workers applying TBT-based antifouling paints. He also reported on a variety of health problems that shipyard workers reported after they started welding and performing other work on the hulls of vessels painted with TBT. The workers symptoms included chronic skin inflammation, respiratory problems, headaches, stomach aches, burning eyes, dizziness, fatigues and frequent colds and flu. One article covered the lawsuit being considered by Charleston shipyard workers, due to a wide range of respiratory problems and constant headaches and coughing related to applying TBT-based paints. The possible human health risks to shipyard workers that were identified subsequently ranged from dermatitis to cancer. Shipyards in the Virginia area (that painted naval ships, cruise ships, and cargo vessels) supported the regulation of TBT because it would protect their workers.

Subsequently, after hearings in the US Congress, key US congressmen from coastal states believing that the EPA regulatory process would be too slow, proposed the 'Organotin Antifouling Paint Control Act of 1988' (OAPCA) which was signed into United States law by President Reagan on 16 June 1988. Coded in this law, P.L. 100-333, (33 USC 2401) are the United States federal laws and regulations concerning the use (and subsequent disposal) of organotin compounds as additives or biocides in antifouling boat bottom paints. It should be noted that the concern for TBT by leading US researchers and the interest at the state level helped congress to quickly draft US national legislation (Champ and Wade, 1996).

Virginia initially accepted the EPA advisory allowable level of 10 ng/l for salt water. However, immediately upon passage of the OAPCA, the Virginia Water Control Board indicated that it thought that the level should be reduced to 1.0 ng/l (Commonwealth of Virginia, 1988) and was subsequently followed by the state of California with a level of 6.0 ng/l. Virginia also passed a state law that set the release rate at not greater than 4 $\mu g/cm^2$ per day (US Congress, 1987a,b).

The Commonwealth of Virginia also was the only US state to set the National Pollution Discharge Elimination Standard (NPDES) permit levels for TBT at 50 ng/l from shipyards and drydocks in state waters. It gave shipyards a 5-year compliance period, which ended in September 1999 to meet this standard. Hull wash down is a 30-h operation using 400 000 l+ of washwater, resulting in TBT levels in wash down waste water ranging from 15 000 to 485 000 ng/l. In 1998, it became apparent that shipyards were not able to comply with regulations using Best Management Practices (BMP) and Best Available Technology (BAT). This led Virginia legislators at state and federal levels to develop a cooperative R&D project, which was initiated during the summer of 1999. The project is supported by the US EPA, The Chesapeake Bay Foundation, and local shipyards in the Norfolk, Virginia area through the Center for Applied Ship Repair and Maintenance (CASRM) at the Old Dominion University to develop TBT wastewater treatment technologies for shipyards and drydocks to meet these standards (see Messing et al., 1997; Champ et al., 1999; Fox et al., 1999).

The impact of regulations in Virginia has been a continued reduction in TBT levels in the marine environment since 1987 and control of point source discharges from shipyards now at less than 200 ng/l levels. The Department of Environmental Quality of Virginia is concerned about the lack of control on the largest non-point source of TBT to coastal waters, since TBT leaching from vessel hulls in ports is not regulated and it believes that international regulation is required to reduce these levels (Johnson, 1999).

The interests of the Commonwealth of Virginia are: reducing environmental and public health risks; protecting the marine environment; promoting economic development; competitiveness of Virginia shipyards, drydocks and ports (Virginia is home to the second-largest port facility on the east coast and the Norfolk area is the largest naval port in the world); and creating jobs. Oyster production in Chesapeake Bay in the state of Virginia is a significant marine resource and economic base for local economies, but it had been declining since the early 1980s. Concern with the difficulty in delineating cause-and-effect relationships and the reported effects on untargeted species attracted Virginia marine scientists.

2.3. The US Antifouling Paint Control Act of 1988

Organotins are the only chemical compound regulated by law in the United States in which environmental legislation has been enacted solely for the chemical by name — The Organotin Antifouling Paint Control Act of 1988 (US Congress, 1988). The purpose of the Act was 'to protect the aquatic environment by reducing immediately the quantities of organotin entering the waters of the United States.' In the Act, there are two permanent sections, the 25-m size requirement, and the prohibition of retail sale of TBT antifouling paint additives. The release rate portion of the bill had a duration time period that would be in effect until a final decision of the administrator of the EPA regarding continued registration of TBT as an ingredient in antifouling paints takes effect.

The prohibitions in the Act are: 'No person in any state may apply to a vessel that is less than 25 m in length, an antifouling paint containing organotin' with the following exceptions: '(1) the aluminum hull of a vessel that is less that 25 m in length; and (2) the outboard motor or lower drive unit of a vessel that is less than 25 m in length'. No person in any state may: (1) sell or deliver to, or purchase or receive from, another person, an antifouling paint containing organotin; or (2) apply to a vessel an antifouling paint containing organotin; unless the antifouling paint is certified by the administrator (of EPA) as being a qualified antifouling paint containing organotin; and (3) sell or deliver to, or purchase or receive from, another person at retail any substance containing organotin for the purpose of adding such substance to paint to create an antifouling paint.

A key certification was that the EPA administrator shall certify each antifouling paint containing organotin that the administrator has determined has a release rate of not more than 4.0 $\mu g/cm^2$ per day on the basis of the information submitted to the EPA in response to its data call in notices. This data is provided by registrants, which is cost effective, but it has the concern of the 'fox guarding the chicken house'. It also creates a public perception that this data could have been manipulated to support special interests or it would not have been provided.

The administrator of the US EPA, in consultation with the Under Secretary of Commerce for Oceans and Atmosphere (NOAA), was required to monitor the concentrations of organotin in the water column, sediments, and aquatic organisms of representative estuaries and near-coastal waters in the United States. The secretary of the Navy was to provide for periodic (not less than quarterly) monitoring of waters serving as the home port for any Navy vessel coated with an antifouling paint containing organotin compounds to determine the concentrations of organotins in the water column, sediments, and aquatic organisms of such waters. These monitoring programs were to remain in effect for 10 years or until the last US Navy ships coated with TBT paint had been removed from service.

Although the OAPCA and subsequent US EPA regulations allowed use of TBT coatings by large vessels, the US Navy in 1989 decided not to use organotin coatings because of environmental concerns and the uncertain regulatory future at state and regional levels. Following the Navy's decision to not use organotins, the regulatory action was perceived as the elimination of 'the problem'. This meant that the decision eliminated the need for the paint manufacturers (now with a reduced market) and the Navy (the previous major US federal agency source) to fund any TBT R&D in the US. Exceptions were: (1) the in-house Navy monitoring studies from 1984 to 1987 in San Diego Bay, Pearl Harbor, the Norfolk region and 12 other harbors, and Navy dry dock release rates studies that were conducted between 1989 and 1995; (2) the monitoring program that was required by the EPA of the paint manufacturers as part of the TBT permit process; (3) the NOAA National Status and Trends Program added the analysis of TBT to their coastal monitoring program (sediment and oyster tissue) to establish a base line for commercially representative and important populations; and (4) analysis of TBT in samples collected by the EPA EMAP program.

2.4. France

France was the first country to regulate the use of organotin antifouling paints in an attempt to reduce environmental concentrations. On 19 January 1982, the French Ministry of Environment announced a temporary 2 year ban on TBT paint containing more than 3% wt. organotin for the protection of hulls of boats of less than 25 t, for both the Atlantic coasts and the English Channel. The decree of 16 September 1982 extended the ban to the whole coastal area and to all organotin paints, beginning on 1 October 1982. These regulations also only allow the application of antifouling paints containing organotin to hulls of all boats and marine craft having an overall length of greater than 25 m. Hulls made of aluminum or aluminum alloys were exempted from the ban. This extension was effective through the 12 February 1987 and banned the application of antifouling paints containing organotin on vessels less than 25 m in length (Alzieu, 1991) and see Alzieu this volume.

2.5. United Kingdom

The first regulatory action in the UK to reduce the environmental impact of organotin compounds from antifouling paint was announced by the Environment Minister in Parliament on 24 July 1985. The action consisted of the following steps: (1) develop regulations to control the retail sale of the most damaging organotin-containing paints (beginning 1 January 1986, they intended to ban the use of 'free association' organotin-based paints by small boat owners, and to set the maximum levels for the organotin content of 'copolymer' paints); (2) establish a notification scheme for all new antifouling agents; (3) develop guidelines for the cleaning and painting of boats coated with antifoulants; (4) propose the establishment of a provisional ambient environmental quality target (EQT) for the concentration of tributyltin in water (20 ng/l was proposed as the UK's EQT); and (5) coordinate and further develop organotin monitoring and research programs so that the government could assess the effectiveness of these regulatory actions at a later date.

The first legislation to control the retail sale of organotin-based antifoulant paints was the Control of Pollution (anti-fouling paints) Regulations of 1985, which came into force on 13 January 1986. These regulations were developed under sections 100 and 104(1) of the Control of Pollution Act of 1974. They prohibited the retail sale of antifouling paints containing organotin compounds if: (1) the total concentration of tin in dried copolymer paints exceeded 7.5% wt. of tin; or (2) the total concentration of tin in other non-copolymer (free association) paints exceeded 2.5% wt. of tin [the Control of Pollution (antifouling paints) Regulations (UK DOE, 1986a,b,c)]. These regulatory actions were enacted with the provision that they would be reviewed with the interim results of the comprehensive scientific studies that were being carried out by both government and non-government laboratories, which included studies on the distribution, fate and effects of TBT in the environment and laboratory toxicity studies.

The DOE subsequently lowered the TBT water quality standard from 20.0 to 2.0 ng/l (Abel, 1996). These new regulations, introduced in January 1987, reduced the maximum allowable tin content of copolymer paints from 7.5 to 5.5% through the Control of Pollution Act (COPA) of 1986, which amended the Control of Pollution

Act of 1985 (the Anti-fouling Paints Regulations, of 1985), UK DOE (1987). These prohibited the retail sale and the supply for retail sale of anti-fouling paints containing a triorganotin compound as well as the wholesale and retail sale of anti-fouling treatments containing such a compound. The ban also did not make any exceptions to accommodate vessels with aluminum hulls, outboard drives, parts or fittings, as have US regulatory strategies.

These regulations came into force on 28 May 1987 [the Control of Pollution (anti-fouling paints and treatments) Regulations, 1987 — Statutory Instruments No. 783 1987]. It also should be noted that the control of pesticide regulatory actions in the UK, shifted from the DOE to the Ministry of Agriculture, Fisheries and Food (MAFF) on 1 July 1987 through powers conferred to MAFF by sections 16(2) and 24(3) of the UK Food and Environmental Protection Act of 1985 and Regulation 5 of the Control of Pesticides Regulations 1986, as reflected in the Statutory Instruments No. 15 10. For a more complete discussion, see Abel (1996).

The UK Government also enacted the Food and Environment Protection Act (FEPA) to ensure that in the future all antifouling agents of any kind would be screened in the same way as other pesticides under provision of Part III. This was coordinated with the Control of Pesticides Regulations of 1986, which provided for the statutory screening of antifouling paints beginning on 1 July 1987. These regulations prohibit the advertisement, sale, supply, storage or use of any pesticide — including antifouling paints and treatments — unless approved by ministers.

2.6. Switzerland, Austria and Germany

Both Switzerland and Austria (which have no direct access to the ocean) have banned all use of TBT in antifouling paints in freshwater environments. In the Federal Republic of Germany, the following regulations for organotin compounds are in force:

- ban on its use for boats less than 25-m long;
- ban on retail sale;
- ban on its use on structures for mariculture;
- TBT limit of 3.8% (wt.) in copolymeric paints; and
- regulation for the safe disposal of antifouling paints after removal (MEPC 30/20/2-IMO, 1990).

2.7. Japan

Monitoring studies in Japan in the late 1980s found that a ‘biologically significant’ amount of organotin compounds derived from antifouling paints had been released to the marine environment with high residues in fish which ranged from 0.06 to 0.75 ng/l TBT; and 0.03 to 2.6 ng/l TPT (triphenyltin), giving some concern for future human health affects. Also bird tissues were found to range from 0.03 to 0.05 ng/l TPT (MEPC 30/WP.1). In 1990, given these findings and the results of laboratory and field studies, seven TPT compounds (January), and 13 TBT compounds (September) were designated as Class II Specified Chemical Substances. Subsequently, the production, import, and use of these compounds have come under the domestic law concerning the examination and regulation of manufacture of chemical substances. Japanese government ministries have introduced domestic countermeasures to prohibit the application of TPT antifouling paints on all vessels including boats, ships, and marine structures. Regarding TBT antifouling paints, the following restrictions came into force in July 1990 (MEPC 30/WP.1, IMO, 1990):

- TBT antifouling paints shall not be applied to non-aluminum hulled vessels engaged in domestic voyages as well as on non-aluminum hulled vessels engaged in international voyages with a dry-docking interval of approximately 1 year; and
- TBT antifouling paints shall not be applied to hulls, other than shell plating between the load line and the bilge keel, of vessels engaged in international voyages with a dry-docking interval of longer than 1 year. Shell plating between the load line and bilge keel of such vessels may be painted with antifouling paints

containing a low percentage of TBT compounds.

Since regulations in 1990, levels in Japan have declined. They also found a high incidence of imposex in over 100 species of sea snails. They also reported that in 1995, TBT and TPT concentrations in all fish and shellfish tissues were below the provisional ADI. Their report is among the first to assess the impact of TBT in the deep sea and in particular TBT levels in squid livers. Squid livers from the open ocean off Japan were found to accumulate TBT to 48 000 times ambient concentrations, suggesting that TBT bound to particulate matter through sinking is the source and pathway to the deep ocean.

2.8. Commission of the European communities

On 1 February 1988, the Commission of the European Communities proposed an amendment for Council Directive (76/769/EEC) restricting the marketing and use of certain dangerous substances and preparations [COM (88) 7 Final-Brussels]. The proposal lists 'organostannic compounds' and restricts their use as substances and constituents of preparations intended for use to prevent the fouling by micro-organisms, plants or animals of: (a) the hulls of boats of an overall length, as defined by ISO 8666, of less than 25 m; and (b) cages, floats, nets and any other appliances or equipment used for fish or shellfish farming (see Davies and McKie, 1987; Davies et al., 1986, 1987), and may be sold only to professional users in packaging of a capacity of not less than 20 l.

2.9. The Paris Commission

The Paris Commission deals with land-based sources of pollution to the north-east Atlantic ocean under the auspices of the Paris Convention. The Convention recommended in 1987 that contracting parties should take effective action to eliminate pollution by TBT of the inshore areas within the Convention. One of the key recommendations was that restrictions should be considered on the use of organotins on sea-going vessels. This recommendation was debated in 1988 and the Commission concluded that for economic reasons a ban on sea-going vessels was not achievable. However, contracting parties agreed 'to develop procedures and technologies aimed at a reduction of the amount of organotins released from boat yards and dry docks due to sand-blasting, dust, paint chips, over spray, etc., and to implement them in the near future' (MEPC 30/IN17.5-IMO, 1990).

2.10. The Barcelona Convention

In 1989, the contracting parties to the Barcelona Convention (for protection of the Mediterranean Sea against pollution) approved a restriction on large vessels. At that time, they also agreed to develop a code of practice to minimize the contamination in the vicinity of boat yards and dry docks to reduce contamination from removal of spent antifouling paints and application of fresh ones. For the Mediterranean Sea, comprehensive assessments of organotin compounds have been prepared by United Nations organizations: the United Nations Environmental Program (UNEP) and Food & Agriculture Organization (FAO) in cooperation with World Health Organization (WHO) and The International Atomic Energy Agency (IAEA) to support the Mediterranean Action Plan (MEPC 29/15/1), MEPC 29/INF.19). The data and information from these assessments have led to a set of recommendations on organotin compounds which was adopted by the sixth ordinary meeting of the contracting parties of the Barcelona Convention:

1. 'as from 1 July 1991 not to allow the use in the marine environment of preparations containing organotin compounds intended for the prevention of fouling by micro-organisms, plants or animals;
2. on hulls of boats having an overall length [as defined by the International Standards Organization (ISO) Standards No. 8666] of less than 25 m; and
3. on all structures, equipment or apparatus used in mariculture.'

In addition, 'Contracting parties not having access to substitute products for organotin compounds by 1 July 1991 would be free to make an exception for a period not exceeding 2 years, after having so informed the Secretariat'. A recommendation was also made 'that a code of practice be developed to minimize the contamination of the marine environment in the vicinity of boat-yards, dry docks, etc., where ships are cleaned of old antifouling paint and subsequently repainted' (MEPC 29/22, IMO, 1990).

2.11. International Maritime Organization

The Marine Environmental Protection Committee (MEPQ) of the International Maritime Organization (IMO), has for several years reviewed the position of organotin compounds in its lists of hazardous substances and collected information on the effects of organotin compounds on the marine environment and human health. Concern had been expressed within the Consultative Meeting of Contracting Parties to the London Dumping Convention (now referred to as the London Convention). The MEPC meets twice a year at the IMO in London and each session is given a number. The 43rd session of the MEPC meeting was on 28 June–1 July, 1999. Summary documents and press briefings for recent sessions are posted on the IMO website. For MEPC 43, the URL is http://www.imo.org/imo/meetings/mepc/43/mepc43.htm. For these meetings, each country or organization can submit position papers, or information documents that are distributed in advance of the meeting. These documents are currently not available on the IMO website, however the US Coast Guard (the Secretariat for the US delegation) has posted MEPC 43 meeting agenda and is in the process of posting these documents on its website at URL: http://www.uscg.mil/hq/g-m/mso4/imomepc-43.htm.

2.12. Historical perspective

At its 29th session (on 27 April, 1990), the MEPC reviewed the actions taken by the consultative meetings of the contracting parties to the London Convention. The MEPC for some years has reviewed the position of organotin compounds in its lists of hazardous substances and collected information on the effects of organotin compounds on the marine environment and human health. Particular concern had been raised within the MEPC of the potential hazards caused by disposal at sea of dredged material from marinas, dockyards, etc., containing high levels of organotin compounds (MEPC 29/15, MEPC 29/22, IMO, 1990).

Subsequently, the Third International Organotin Symposium, of which IMO was a co-sponsor, was held in Monaco (17–20 April 1990). A special policy and regulatory session was chaired by the author of this paper, in which a conceptual list of regulatory requirements (Champ, unpublished document) was presented to the IMO for consideration in developing global regulations (Stewart, 1996):

1. Implement no- or low-cost regulatory requirements.
2. Implement fee schedules. Biocide producer pays all registration fees. Benefited user pays user benefit fee as an environmental degradation fee.
3. Create an environmental degradation fund from user benefit fees to support regional research, monitoring, and mediation activities. To be coordinated by a national research review panel.
4. Implement limited cost ($>10\%$) bureaucratic and administrative management structures to manage these funds and activities.
5. Require all international vessels (as part of the ship's registration papers) to have certified and duly recorded, the following specific data related to the use of organotin compounds in antifouling paints: the specific type, composition, release rate, and quantity of organotin utilized.

In this session at Monaco, the author also identified the following comprehensive range of regulatory options that could be considered for regulating the use of organotin compounds in antifouling boat bottom paints:

1. Total ban on the use of organotin compounds in antifoulant paints.
2. Regulate the use of organotin compounds by the length of vessels, such as prohibition on vessels of less than 25 m in length with approval on all aluminum hull vessels. Ban on non-commercial or recreational vessels-any length.
3. Limit the amount of organotins (on a percentage basis) in a specific paint formulation.
4. Limit the release rate of organotins from antifouling paints to the adjacent water column.
5. Regulate the application/removal of antifouling paints, which utilize organotins to trained and certified applicators.
6. Regulate the removal, containment, clean up, and disposal of antifouling paints which contain organotins which are removed from vessels in dry dock facilities.
7. Regulate the discharge rates of organotins in discharge waters from dry dock facilities by standard prevention practices and clean up procedures.
8. Regulate the dockage time of large vessels (25 m) that utilize organotin-based antifouling paints to specific time periods with limited excess at anchor time in harbors.
9. Foreign vessels utilizing organotin based antifouling paints in harbors are required to pay an environmental degradation fee (US$1200/day or $50/h) for anchoring time in estuaries or ports (airport users tax).
10. Self-regulatory public information strategies for small boat owners — who had painted their boat with organotin-based antifouling paints — within the last 5 years.

Participants at Monaco felt that some of the above suggestions were not applicable to their own country. They might be either: (1) impracticable (e.g. environmental charges for use of the paints, or as a restriction on the amount of time spent in waterways); and or (2) not relevant to IMO (such as a ban on the use of organotin on vessels of less than 25 m).

The following MEPC Resolution [MEPC.-29(30)] was adopted considering all of the above suggestions on 16 November 1990 (IMO, 1990):

1. to recommend that governments adopt and promote effective measures within their jurisdictions to control the potential for adverse impacts to the marine environment associated with the use of tributyltin compounds in antifouling paints, and as an interim measure specifically consider actions as follows:
    - eliminate the use of antifouling paints containing tributyltin compounds on non-aluminum hulled vessels of less than 25 m in length;
    - eliminate the use of antifouling paints containing tributyltin compounds which have an average release rate of more than 4 $\mu g/cm^2$ per day;
    - to develop sound management practice guidance applicable to ship maintenance and construction facilities to eliminate the introduction of tributyltin compounds into the marine environment as a result of painting, paint removal, cleaning, sandblasting, or waste disposal operations, or run-off from such facilities;
    - to encourage development of alternatives to antifouling paints containing tributyltin compounds, giving due regard to any potential environmental hazards which might be posed by such alternative formulations; and
    - to engage in monitoring to evaluate the effectiveness of control measures adopted and provide for sharing such data with other interested parties.
2. to consider appropriate ways towards the possible total prohibition in the future of the use of tributyltin compounds in antifouling paints for ships.

At the 30th session of the Marine Environmental Pollution Committee (MEPC) of the International Maritime Organization (IMO), the Japanese delegation indicated that it felt that the above interim measures were insufficient and that 'a total ban on the use of TBT antifouling paints on all vessels including vessels engaged in international voyages should be introduced as soon as possible as an international agreement'.

In reviewing the papers submitted to the MEPC correspondence group set up by MEPC 38/14, the Japanese submission (MEPC 41/INF.3) 'calls for the worldwide ban on every use of organotin-based antifouling paints for ship bottoms' and reports that since 1990, the use of organotin compounds have been practically prohibited by government regulation and voluntary restriction by the industry; but that the international traffic of large ships in Japanese waters is their main source of TBT pollution today. They compared monitoring data from harbors with high large vessel density to those with low vessel density (without normalizing the data for dilution volumes, water retention times, mixing, etc.) and determined that the high incidence of ocean-going vessels was causing the higher levels of TBT in these ports and harbors.

2.13. Status of the proposed IMO Organotin Convention

In 1998, at MEPC 42, several countries (Belgium, Denmark, France, Germany, Norway, the Netherlands, Sweden, and the UK) joined Japan in requesting a global ban and proposed that the MEPC recommend a 10 year period for phasing out (total ban) of the use of TBT in antifouling boat bottom paints on ships worldwide (MEPC 42/22, annex 5). It was proposed that the legal instrument to be developed by the IMO should be a free standing convention, legally binding, global in scope, effective, and should be such as to ensure expeditious entry-into-force, and furthermore agreed that the instrument should include a mechanism for addressing antifouling systems other than organotin-based systems.

At its 43rd session (28 June–2 July 1999) the MEPC agreed to use the framework and the basic text contained in document MEPC 43/3/2 as a basis for developing the legal instrument. (MEPC 43/21). The Committee also reviewed documents by the Marshall islands (MEPC 43/3/6) and joint documents by BIMCO, INTERCARGO, ICS, INTERTANKO, OCIMF and SIGTTO (MEPC 43/3/9) related to their concern on the timing of the phase-out dates (2003 and 2008) and indicated that these were 'tentative target' dates which would be finalized at the diplomatic conference which considers the legal instrument.

Also at its 43rd session the MEPC held a roll-call vote to establish whether the committee was satisfied that sufficient progress had been made in preparing for a diplomatic conference (with 35 delegations in favor, 12 against, and 15 abstaining). MEPC agreed to request the IMO Council meeting in November 1999, for approval of 'the holding of a 1-week diplomatic conference on antifouling systems to be held in the 2000–2001 biennium to adopt a legal instrument to regulate the use of shipboard antifouling systems, in particular to phase out those containing organotins such as tributyltin (TBT)'. It was also recommended that a review of all antifoulants is inappropriate and that consideration would be limited in the treaty to specific proposals made by parties that request international action on a specific antifouling system or biocide. The MEPC Working Group, (formed at MEPC 42), proposed a 10 year period to implement the ban with 1 January 2003 being the last date for the application of TBT-based antifouling paints and 1 January 2008 being the last date for TBT-based marine coatings to be on a vessel. Discussion text (Document MEPC 43/3/21) for the proposed organotin convention was submitted by the US is downloadable from the USCG Website at URL: http://www.uscg.mil/hq/g-m/mso4/imomepc-43.htm.The summary report of the MEPC 43 (167 pages) is available and can be downloaded. In addition, the MEPC has also proposed that IMO promote the use of environmentally safe antifouling technologies to replace TBT. Following the general assembly meeting in November 1999, an excellent summary and the DRAFT text of the legal instrument is presented in the summary report of the MEPC Antifouling Paints Working Group (MEPC, 44/3), this document can be found on the USCG web site at: http://www.uscg.mil/hq/g-m/mso4/imomepc44.htm.

The summary of MEPC from the 43rd session (MEPC 44/3, 10 November 1999), following presentations and discussions of the Working Group for Harmful Effects on the Use of Antifouling Paints for Ships identified several remaining issues for developing the Treaty. These are: identification, review, selection and listing of 'restricted' antifouling systems, notification systems, determination of leaching rates, enforcement either by the port States or flag States. The issue of port States and flag States relative to this Treaty is a separate and major issue at IMO and is not discussed in this paper.

The most subtle change over the past 24 months of discussion is the shift from a ban on TBT to a ban on organotins in general to now more of a focus on 'restricted antifouling systems.' Marine coating experts may feel that the 'regulation by general categories' is too expansive, and highly restrictive to the development of alternatives to TBT. Because perfectly acceptable alternatives to TBT (as we known TBT now), may be in the future either a modified TBT or other new or developed organotins (Alex Milne, personal communication). In part, these changes have occurred as member States become involved and informed in the debate and realize who will be policing and enforcing the global ban (and the difficulty of such, given the potential for the development of an illegal marketplace). IMO is not a policing organization.

An international ISO Working Group has been charged with developing an international standard method of measuring leaching rates of biocides from antifouling system (MEPC 43/3/1). The majority of the MEPC Working Group considered that there was not a compelling need for ISO to continue their work on determining the leaching rate of tin-based biocides. However, it should be noted that there is not a quick, inexpensive or standardized non-destructive method for detecting the presence of TBT on a vessel in the water or a method for measuring precise release rates from a vessel hull (as you would need for port inspections) in the water. The current most sensitive and standardized method – Grignard Derivatization (Unger et al., 1996) for the analysis of TBT requires from 2 to 3 days (which is long after the ship has left port). In the U.S., only a few EPA or NOAA certified laboratories currently can analyze at the low ng/L (1–10 parts per trillion) detection level at a rate of around 5 water samples per week, each costing \$500 to % \$1000 USD. These concerns for inspection, policing, and enforcement of a ban are further discussed in Champ et al. (1999).

A second issue not discussed at these meetings is who will be liable for the new additional costs of dredging, treatment and disposal of TBT contaminated bottom sediment after TBT has been banned. Will it be port and harbor authorities, shipyards and drydocks, ship owners or the paint manufacturers? For further discussion, see section 12 of this paper and Champ (1999c).

IMO will not create a list of approved systems or review all antifoulants. Therefore, in the future, for an antifouling system to be banned under the treaty, a contracting party (member nation) to the Treaty must introduce a proposal to restrict a specific antifoulant (i.e. innocent until proven guilty) as described in a two step process in Section 4 (MEPC 45/4). This proposal must be evaluated and supported by an experts group appointed by IMO – as defined in Section 5 (MEPC 45/4), which now includes non-parties, IGO's and NGO's.

This expert group is unfortunately far short of the proposed independent and neutral international Marine Coatings Board (MCB) proposed in 1988 by Champ (1999b) and discussed in 14 of this paper, and in editorial comments by Abel this volume). The expert working group (without independent funding for international standardized studies) would be reviewing submissions (similar to the present process which is not comprehensive or internationally standardized and in the past has been mostly data provided by the paint companies). In addition, the remaining less than 1000 days for shipowners to select an alternative to TBT (till 1.1.2003), a smooth transition process requires comparable data and information on available alternatives to prevent a Catch 22' like TBT from occurring again.

A second problem identified at the November (1999) meeting is related to whether a total removal of all traces of TBT from a ship's hull will be required. The Working Group is debating whether a total removal of TBT will be required, or whether an overcoat painted on the hull (with or without sealants) is sufficient. Total removal of TBT results in longer time periods in shipyards and greater amounts of both solid and liquid TBT contaminated wastes for treatment, discharge and or disposal in shipyards.

The Working Group for Harmful Effects on the Use of Antifouling Paints for Ships of MEPC will have two full meetings (October 2–6, 2000 and April 2001) with the IMO Marine Environment Protection Committee to continue these discussions and deliberations for the development of the final draft language before the Treaty Diplomatic Conference which is proposed for October/December 2001.

3. Comments on the scientific basis for the regulation of TBT

It is interesting to note, that the 'movement' to regulate TBT-based antifouling paints during the 1980s was initially based on 'correlation' and 'generality' type science (see Salazar and Champ, 1988). Peruse the bioassay discussions in White and Champ (1984), and see Evans et al. (1996), Evans (1997, 1999a); Evans and Nicholson (this volume); for a discussion on imposex. The Salazar and Champ (1988) paper was a preliminary review of the science that was prepared for an Oceans 1988 conference proceedings to stimulate discussions. However, it was published about the same time that the OAPCA was passed in the US and interest in TBT and support for further research declined (Champ and Seligman, 1996b). Fortunately as evidenced in Table 1, this was not true on a global basis. Some of these concerns have been revisited and are discussed in a collection of papers reprinted and submitted by the paint industry to the MEPC by the Organotin Environmental Program Association (ORTEP, 1996, 1997, 1998; ORTEP, 1999). Many of these points were discussed at the 1998 Annual Meeting of the American Chemical Society in Dallas (Rouhi, 1998) and in Champ (1998). In addition, papers were presented at the Oceans 1999 Conference in Seattle (September, 1999) that discussed the science being used in the regulatory process (see Brancato and MacLellan, 1999; Cardwell et al., 1999b; Damodaran et al., 1999; Evans, 1999c; Evans and Nicholson, 1999; Evans and Smith, 1999; MacLellan et al., 1999; Toll et al., 1999). Several of these papers delineate problems with data quality and quantity, protocols and question the emphasis of the data and the information that is being utilized as a basis for proposed additional regulation. *However, these points are moot if comparable and environmentally friendly alternatives to TBT are available and acceptable (Champ, 1998, 2000).*

This issue of the science being used in the regulatory process is not a 'red herring'; it is a red flag the scientific community needs to address collectively. The regulatory community needs a firm scientific basis for policy and decision-making (Champ, 1999a). Scientists by their nature are always in the 'question or continue the debate mode'. Unless we can define a cause-and-effect relationship to the *n*th degree, we may not feel that we have enough data to be conclusive (and we fail to support the regulatory process). Thus, the regulatory debate continues without us on it. In the policy world, after the scientists have identified the problem, the regulators determine its relevance. Something that we do not appreciate is that regulators make adjustments for uncertainty in assessing environmental impacts and/or establishing cause and effects by using application (correction) factors in setting standards or exposure levels. They do this to be conservative and are on the side of environmental protection. We are too precise and need over 100% proof (because of risk of not being 100% 'correct as in perfect science) and do not appreciate that these application factors reduce that risk. Policy and decision-makers interpret public interest and assess importance, because it is perceived to be their responsibility. Scientists tend to communicate mostly with other scientists and often create confusion by taking both points of view for the sake of debate and learning. All of this factors into the gut reaction or trust of the policy and decision-maker who seeks a 'yes or no' answer with a scale defined for each instead of a 'maybe'.

Table 1
Listing of journal publications (first author and title) associated with monitoring, bioaccumulation, and effects/impact/imposex/toxicity of TBT, from the enactment of national regulations to the present

Recent citations Monitoring papers	Title — subject area
Alzieu et al. (1989), Thompson et al. (1985)	Monitoring and assessment of butyltins in Atlantic coastal waters
de Mora et al. (1989), Maguire and Tkacz (1987)	Tributyl tin and total tin in marine sediments: profiles and the apparent rate of TBT degradation.
King et al. (1989)	Tributyl tin levels for sea water, sediment, and selected marines in coastal Northland and Auckland, New Zealand.
Kram et al. (1989)	Adsorption and desorption of tributyltin in sediments of San Diego Bay and Pearl Harbor.
Lee et al. (1989)	Importance of microalgae in the biodegradation of tributyltin in estuarine waters.
Seligman et al. (1988)	Evidence for rapid degradation of tributyltin in a marina.
Seligman et al. (1989)	Distribution and fate of tributyltin in the United States marine environment.
Stewart and de Mora (1990)	A review of the degradation of tri(*n*-butyl) tin in the marine environment
Alzieu et al. (1991)	Organotin compounds in the Mediterranean: a continuing cause for concern.
Cleary (1991)	Organotin in the marine surface microlayer and sub-surface waters of south-west England: relation to toxicity thresholds and the UK environmental quality standard.
Evans and Huggett (1991)	Statistical modeling of intensive TBT monitoring data in two tidal creeks of the Chesapeake Bay.
Ritsema et al. (1991)	Butyltins in marine waters of the Netherlands in 1988 and 1989: concentrations and effects.
Valkirs et al. (1991)	Long-term monitoring of tributyltin in San Diego Bay California.
Waite et al. (1991)	Reductions in TBT concentrations in UK estuaries following legislation in 1986 and 1987.
Waite et al. (1996)	Changes in concentrations of organotins in water and sediment in England and Wales following legislation.
Chau et al. (1992a)	Determination of butyltin species in sewage and sludge.
Chau et al. (1992b)	Occurrence of butyltin species in sewage and sludge in Canada.
Dowson et al. (1992)	Organotin distribution in sediments and waters of selected east coast estuaries in the UK.
Hardy and Cleary (1992)	Surface microlayer contamination and toxicity in the German Bight.
Stang et al. (1992)	Evidence for rapid, non-biological degradation of tributyltin in fine-grained sediments.
Stewart and de Mora (1992)	Elevated tri(*n*-butyl)tin concentrations in shellfish and sediments from Suva Harbor, Fiji.
Cortez et al. (1993)	Survey of butyltin contamination in Portuguese coastal environments.
Dowson et al. (1993b)	Depositional profiles and relationships between organotin compounds in freshwater and estuarine sediment cores.
Dirkx et al. (1993)	Determination of methyl- and butyltin compounds in waters of Antwerp harbor.
Foale (1993)	An evaluation of the potential of gastropod imposex as an indicator of tributyltin pollution in Port Phillip Bay, Victoria.
Yonezawa et al. (1993)	Distributions of butyltins in the surface sediment of Ise Bay, Japan.
CEFIC (1994)	Results of TBT monitoring studies.
Macauley et al. (1994)	Annual statistical summary: EMAP — estuaries Louisianian Providence 1993.
Ritsema (1994)	Dissolved butyltins in marine waters of the Netherlands 3 years after the ban.
Abd-Allah (1995)	Water and biota from Alexandria harbors.
de Mora et al. (1995)	Sources and rate of degradation of tri(*n*-butyl)tin in marine sediments near Auckland, New Zealand
Gomez-Ariza et al. (1995)	Acid extraction treatment of sediment samples for organotin speciation; occurrence of butyltin and phenyltin compounds on the Cadiz coast, south-west Spain.
Ko et al. (1995)	Tributyltin contamination of marine sediments of Hong Kong.
Michel and Averty (1995)	Tributyltin contamination in the Rade De Brest.
Minchin et al. (1995)	Marine TBT antifouling contamination in Ireland, following legislation in 1987.
Stronkhorst et al. (1995)	TBT contamination and toxicity of harbor sediments in the Netherlands.
Batley (1996)	The distribution and fate of tributyltin in the marine environment

Table 1 (*Continued*)

Recent citations **Monitoring papers**	Title — subject area
Dowson et al. (1996)	Persistence and degradation pathways of tributyltin in freshwater and estuarine sediments.
Grovhoug et al. (1996)	Tributyltin concentrations in water, sediment, and bivalve tissues from San Diego Bay and Hawaiian harbors.
Huggett et al. (1996)	Tributyltin concentrations in waters of the Chesapeake Bay.
Kalbfus et al. (1996)	Analysis of butyltin species in water, sediment and environmental matrices.
Maguire (1996a)	Tributyltin in Canadian waters.
Maguire (1996b)	The occurrence, fate and toxicity of tributyltin and its degradation products in fresh water environments.
Minchin et al. (1996)	Biological indicators used to map organotin contamination in Cork harbor, Ireland.
Russell et al. (1996)	Comparison of trends in tributyltin concentrations among three monitoring programs in the United States.
Stronkhorst (1996)	TBT contamination and toxicity of sediments. The present status of TBT-copolymer antifouling paints.
Tong et al. (1996)	Tributyltin distribution in the coastal environment of Peninsular Malaysia.
Ariese et al. (1997)	Monitoring Loswal Northwest dumping location 1996.
de Mora and Phillips (1997)	Tributyltin (TBT) pollution in riverine sediments following a spill from a timber treatment facility in Henderson, New Zealand.
Chau et al. (1997a)	Occurrence of organotin compounds in the Canadian aquatic environment 5 years after the regulation of antifouling uses of tributyltin.
Chau et al. (1997b)	Occurrence of butyltin compounds in mussels in Canada.
Colin et al. (1997)	Organo-Tin Concentrations in Brest Naval Port, in 1993 and 1994. Ecorade: The Bay of Brest: its state of environmental health.
Kan-Atireklap et al. (1997)	Contamination by butyltin compounds in sediments from Thailand.
Saint-Louis et al. (1997)	Tributyltin and its degradationi n the St. Lawrence Estuary (Canada).
Smeenk (1997)	Strandings of sperm whales *Physeter macrocephalus* in the North Sea: history and patterns.
Stewart and Thompson (1997)	Vertical distribution of butyltin residues in sediments of British Columbia Harbors.
Hashimoto et al. (1998)	Concentration and distribution of butyltin compounds in a heavy tanker route in the strait of Malacca and in Tokyo Bay.
Oh (1998)	Studies on TBT contamination in marine environment of Korea.
Ritsema et al. (1998)	Determination of butyltins in harbour sediment and water by aqueous phase ethylation GC-ICF-MS and hydrode generation GC AAS.
Thompson et al. (1998)	Recent studies of residual in coastal British Columbia sediments.
Yang et al. (1998)	Occurrence of butyltin compounds in beluga whales *(Delphinapterus leucas)*.
Michel and Averty (1999)	Contamination of French coastal waters by organotin compounds: 1997.
Rees et al. (1999)	Surveys of the Epibenthos of the Crouch Estuary (UK) in relation to TBT contamination.
Rilov et al. (1999)	Unregulated use of TBT-based antifouling paints and TBT pollution in Israel.
Hwang et al. (1999)	Tributyltin compounds in mussels, oysters, and sediments of Chinhae Bay.
Waldock et al. (1999)	Surveys of the benthic infauna of the Crouch Estuary (UK) in relation to TBT contamination.
Murray et al. (In Press)	Sediment quality in dredged material disposed to sea from England and Wales. CATS 4: Conference on the characterization and treatment of sediments.
Bioaccumulation papers	
Batley et al. (1989)	Accumulation of tributyltin by the Sydney rock oyster, *Saccostrea commercialis.*
Rice et al. (1989)	Uptake and catabolism of tributyltin by blue crabs fed TBT contaminated prey.
Langston and Burt (1991)	Bioavailability and effects of sediment-bound TBT in deposit-feeding clams, *Scrobicularia plana.*
Salazar and Salazar (1991)	Assessing site specific effects of TBT contamination with mussel growth rates.
Tas and Opperhuizen (1991)	Analysis of triphenyltin in fish.

Table 1 (*Continued*)

Recent citations **Monitoring papers**	**Title — subject area**
Wade et al. (1991)	Oysters as biomonitors of butyltins in the Gulf of Mexico.
Garcia-Romero et al. (1993)	Butyltin concentrations in oysters from the Gulf of Mexico from 1989 to 1991.
Iwata et al. (1995)	High accumulation of toxic butyltins in marine mammals from Japanese coastal waters.
Kannan et al. (1995)	Butyltins in muscle and liver of fish collected from certain Asian and Oceanian countries.
Kannan et al. (1996a)	Accumulation pattern of compounds in dolphin, tuna and shark collected from Italian coastal waters.
Kannan et al. (1996b)	Sources and accumulation of butyltin compounds in Ganges River dolphin, *Platanista gangetica*
Kannan and Falandysz (1997)	Butyltin residues in sediment, fish, fish-eating birds, harbor porpoise and human tissues from the Polish coast of the Baltic Sea.
Ariese et al. (1998)	Butyltin and phenyltin compounds in liver and blubber samples of sperm whales *(Physeter macrocephalus)* stranded in the Netherlands and Denmark.
Kannan and Falandysz (1998)	Butyltin residues in sediment, fish, fish-eating birds, harbor porpoise and human tissues from the polish coast of the Baltic Sea.
Kannan et al. (1998)	Butyltin residues in southern sea otters (*Enhydra lutris nereis*) found dead along California coastal waters.
Law et al. (1998)	Organotin compounds in liver tissue of harbor porpoises (*Phocoena phocoena*) and Grey Seals (*Halichoerus grypus*) from the coastal waters of England and Wales.
Salazar and Salazar (1998)	Using caged bivalves as part of an exposure-dose-response to support and integrated risk assessment strategy.
Shim et al. (1998a)	Tributyltin and triphenyltin residues in Pacific oyster (*Crassostrea gigas*) and rock shell (*Thais clavigera*) from the Chinhae Bay System, Korea.
Shim et al. (1998b)	Accumulation of tributyl- and triphenyltin compounds in Pacific oyster, *Crassostrea gigas*, from the Chinhae Bay System, Korea.
Tanabe et al. (1998)	Butyltin contamination in marine mammals from north Pacific and Asian waters.
Karman and Falandysz (1999)	Response to the comment on: butyltin residues in sediment, fish, fish-eating birds, harbor porpoise and human tissues from the Polish coast of the Baltic Sea.
Law et al. (in press)	Butyltin compounds in liver tissue of pelagic marine mammals stranded on the coasts of England and Wales.
Saint-Jean et al. (1999)	Butyltin concentrations in sediments and blue mussels (*Mytilus edulis)* of the southern Gulf of St. Lawrence, Canada.
St-Louis et al. (in press)	Recent butyltin contamination in beluga whales (*Delphinapterus leucas*) from the St. Lawrence Estuary and Northern Quebec, Canada.
Effects−imposex−toxicity	
Minchin et al. (1987)	Possible effects of organotins on scallop recruitment
Davies et al. (1988)	Effects of tributyltin compounds from antifoulants on pacific oysters *Crassostrea gigas* in Scottish Sea Lochs, UK
Gibbs et al. (1988)	Sex change in the female dog-whelk *Nucella lapillus*, induced by tributyltin from antifouling paints.
Martin et al. (1989)	Acute toxicity, uptake, depuration and tissue distribution of tri-*n*-butyltin in rainbow trout, *Salmo gairdneri*.
Bailey and Davis (1991)	Continuing impact of TBT, previously used in mariculture, on dogwhelk (N*ucella lapillus* L.) populations in a Scottish sea loch.
Davies and Bailey (1991)	The impact tributyltin from large vessels on dogwhelk (*Nucella lapillus*) populations around Scottish oil ports.
Evans et al. (1991)	Recovery of dogwhelks, *Nucella lapillus* (L.) suffering from imposex.
Gibbs et al. (1991)	TBT-induced imposex in the dogwhelk, *Nucella lapillus*: geographical uniformity of the response and effects.
Lee (1991)	Metabolism of tributyltin by marine animals and possible linkages to effects.
Moore et al. (1991)	Chronic toxicity of tributyltin to the marine polychaete worm, *Neanthes arenaceodentata*.
Spooner et al. (1991)	The effect of tributyltin upon steroid titres in the female dogwhelk, *Nucella lapillus*, and the development of imposex.
Dyrynda (1992)	Incidence of abnormal shell thickening in the Pacific oyster *Crassostrea gigas* in Poole Harbour (UK), subsequent to the 1987 TBT restrictions.
Stewart et al. (1992)	Imposex in New Zealand neogastropods.
Douglas et al. (1993)	Assessments of imposex in the dogwhelk (*Nucella lapillus*) and tributyltin along the north-east of England.

Table 1 (*Continued*)

Recent citations Monitoring papers	**Title — subject area**
Fent and Stegeman (1993)	Effects of tributyltin in vivo on hepatic cytochrome P450 forms in marine fish.
Meador (1993)	The effect of laboratory holding on toxicity response of marine infaunal amphipods to cadmium and tributyltin.
Meador et al. (1993)	Differential sensitivity of marine infaunal amphipods tributyltin
Widdows and Page (1993)	Effects of tributyltin and dibutyltin on the physiological energetics of the mussel, *Mytilus edulis*.
Evans et al. (1994)	Recovery of dogwhelk populations on the Isle of Cumbrae, Scotland following legislation limiting the use of TBT as an antifoulant.
Oehlmann et al. (1994)	New perspectives of sensitivity of littorinids to TBT pollution.
Ten Hallers-Tjabbes et al. 1994	Imposex in whelks (*Buccinum undatum*) from the open North Sea: relation to shipping traffic intensities.
Bauer et al. (1995)	TBT effects on the female genital system of *Littorina littorea:* a possible indicator of tributyltin pollution.
Cadée et al. (1995)	Why the whelk *(Buccinum undatum)* has become extinct in the Dutch Wadden Sea.
Evans et al. (1995)	Tributyltin pollution: a diminishing problem following legislation limiting the use of TBT-based antifouling paints.
Guolan and Young (1995)	Effects of tributyltin chloride on marine bivalve mussels.
Horiguchi et al. (1995)	Imposex in Japanese gastropods (Neogastropoda and Mesogastropoda): effects of tributyltin and triphenyl from antifouling paints.
Minchin (1995)	Recovery of a population of the flame shell, *Lima hians,* in an Irish bay previously contaminated with TBT.
Svavarsson and Skarphédinsdóttir (1995)	Imposex in the dog-whelk *Nucella lapillus* (L) in Icelandic waters.
Ten Hallers-Tjabbes and Boon (1995)	Whelks *(Buccinum undatum* L.) and dogwhelks *(Nucella lapillus* L.) and TBT — a cause for confusion.
Tester and Ellis (1995)	TBT controls and the recovery of whelks from imposex.
Champ and Seligman (1996a)	Organotin: environmental fate and effects.
Evans et al. (1996)	Widespread recovery of dogwhelks, *Nucella lapillus* (L.) from tributyltin contamination in the North Sea and Clyde Sea.
Gibbs and Bryan (1996a)	Reproductive failure in the gastropod *Nucella lapillus* associated with imposex caused by tributyltin pollution: a review.
Gibbs and Bryan (1996b)	TBT-induced imposex in neogastropod snails: masculinization to mass extinction.
His (1996)	Embryogenesis and larval development in *Crassostrea gigas:* experimental data and field observations on the effect of tributyltin compounds.
Huet et al. (1996)	Survival of *Nucella lapillus* in a tributyltin-polluted area in west Brittany: a further example of a male genital defect (dumpton syndrome) favoring survival.
Minchin et al. (1996)	Biological indicators used to map organotin contamination in Cork harbor, Ireland.
Moore et al. (1996)	Surveys of dogwhelks *Nucella lapillus* in the vicinity of Sullom Voe, Shetland, August 1995.
Oehlmann et al. (1996)	Tributyltin (TBT) effects on *Ocinebrina aciculata* (Gastropoda: Muricidae): imposex development, sterilization, sex-change and population decline.
Smith (1996)	Selective decline in imposex levels in the dogwhelk *Lepsiella scobina* following a ban on the use of TBT antifoulants in New Zealand.
Ten Hallers-Tjabbes et al. (1996)	The decline of the North Sea whelk *(Buccinum undatum L.)* between 1970 and 1990: a natural or a human-induced event?
Tester et al. (1996)	Neogastropod imposex for monitoring recovery from marine TBT contamination.
Bauer et al. (1997)	The use of *Littorina littorea* for tributyltin (TBT) effect monitoring-results from the Berman TBT survey 1994/1995 and laboratory experiments.
Gibbs et al. (1997)	Evidence of the differential sensitivity of neogastropods to tributyltin (TBT) pollution, with notes on a species *(Columbella rustica)* lacking the imposex response.

Table 1 (*Continued*)

Recent citations Monitoring papers	**Title — subject area**
Meador (1997)	Comparative toxicokinetics of tributyltin in five marine species and its utility in predicting bioaccumulation and acute toxicity.
Meador et al. (1997)	Toxicity of sediment-associated tributyltin to infaunal invertebrates: species comparison and the role of organic carbon
Mensink et al. (1997a)	Bioaccumulation of organotin compounds and imposex occurrence in a marine food chain (eastern Scheldt, the Netherlands).
Mensink et al. (1997b)	Tributyltin causes imposex in the common whelk, *Buccinum undatum:* mechanism and occurrence.
Minchin and Minchin (1997)	Dispersal of TBT from a fishing port determined using the dogwhelk *Nucella lapillus* as an indicator.
Minchin et al. (1997)	Biological indicators used to map organotin contamination from a fishing port, Killybeg, Ireland.
Prouse and Ellis (1997)	A baseline survey of dogwhelk *(Nucella lapillus)* imposex in eastern Canada (1995) and interpretation in terms of tributyltin (TBT) contamination.
Swennen et al. (1997)	Imposex in sublittoral and littoral gastropods from the Gulf of Thailand and Strait of Malacca in relation to shipping.
Evans (1997)	Assessments of tributyltin contamination from 1986 until 1997. The misues of imposex as a biological indicator of TBT pollution.
Atkins (1998)	Assessment of the risks to health and to the environment of tin organic compounds in antifouling paint and of the effects of further restrictions on marketing and use.
Day et al. (1998)	Toxicity of tributyltin to four species of freshwater benthic invertebrates using spiked sediment bioassays.
Evans (1999b)	TBT or not TBT?: that is the question.
Folsvik et al. (1998)	Quantification of organotin compounds and determination of imposex in populations of dogwhelks (*Nucella lapillus)* from Norway.
Matthiessen and Gibbs (1998)	Critical appraisal of the evidence for tributyltin-mediated endocrine disruption in molluscs.
Morgan et al. (1998)	Imposex in *Nucella lapillus* from TBT contamination in south and southwest Wales: a continuing problem around ports.
Nicholson et al. (1998)	The value of imposex in the dogwhelk *Nucella lapillus* and the common whelk *Buccinum undatum* as indicators of TBT contamination.
Oehlmann et al. (1998)	Imposex in *Nucella lapillus* and intersex in *Littorina littorea*: interspecific comparison of two TBT-induced effects and their geographical uniformity.
Valkirs et al. (1998)	Use of tributyltin by commercial sources and the US Navy: fate-and-effects assessment and management of impacts on the marine environment.
Meador and Rice (in press)	Impaired growth in the polychaete *Armandia brevis* exposed to tributyltin in sediment.
Poloczanska and Ansell (1999)	Imposex in the whelks *Buccinum undatum* and *Neptunea antiqua* from the west coast of Scotland.
Tanguy et al. (1999)	Effects of an organic pollutant (tributyltin) on genetic structure in the Pacific oyster *Crassostrea gigas.*
Meador (2000)	Predicting the fate and effects of tributyltin in marine systems

It is interesting to note that the general public as well as policy and decision-makers have a greater degree of conservatism (or orders of magnitude) in protecting the marine environment than they require for terrestrial environments. We lack a settlement process or a closure to the scientific debate or a process that integrates and interprets scientific opinion (and asks the 'so what' question). Part of the problem is the time it takes to get scientific peer reviewed papers in journals, which can be 18 months. In addition, consensus is a very difficult process for those making tradeoffs if they have a stake in what is traded off. We also are restricted by our disciplines in which biologists have suffered from having not enough chemical data and chemists from having not enough biological data. In cases like TBT, where effects occur at the 1 ppt level (ng/l), which is the equivalent to 1 s in 31 000 years, a great depth of understanding specific to the chemistry and bioavailability of organotins and subsequent biological uptake and effects is required to appreciate the uncertainty or significance of specific data or this problem would have been solved before now.

In the 1980s international scientific conferences have been used to delineate and discuss multidisciplinary issues associated with the TBT problem (see Proceedings National and International Organotin Symposia, 1986; Proceedings National and International Organotin Symposia, 1987; Proceedings National and International Organotin Symposia, 1988; Proceedings National and International Organotin Symposia, 1989; and Proceedings National and International Organotin Symposia, 1990). The importance of the regulation of TBT merits such an effort predicated upon the economic impact of the global ban on the shipping industry, and is exacerbated by consideration that none of the available alternatives have global approval. The paint companies and the shipping industry could sponsor a neutral, independent international study of the top scientists in the world in this area to conduct a scientific peer review of what we know and have a group prepare a formal risk assessment integrating the data and information from all the interested parties. This effort could start with the EPA Integrated Risk Information System (IRIS). http://ww.epa.gov.jgov/iris (US EPA, 1997) and update the Navy risk assessment conducted in 1997 (US NAVY, 1997).

It is interesting to note that the shipping industry (which is highly fragmented and relatively uninvolved in the current TBT debate), part of their disinterest is that antifouling costs are a small percentage of the operating costs. They also believe that if they are going to be regulated for marine coatings, the regulators should be responsible for providing extensive test data on available alternatives at no cost, in exchange for the burden of being regulated.

3.1. The scientific controversy

Early recognition of the environmental impact of TBT was a simple correlation of the presence of high numbers of boats painted with TBT-base antifouling paints in an estuarine area where deformed oysters were first found and not validated scientific studies (see de Mora, 1996b). TBT levels in surface waters or the water column were not measured until after the correlation was first reported at International Council for Exploration of the Seas (ICES) and in the scientific literature. This was due to several reasons that included the difficulty in analyzing for TBT at the then limit of detection and lack of standardized laboratory analytical protocols, or standard reference materials SRMs). Because TBT's action level was near its detection level, life cycle biologists were the first to investigate the observed impact on oysters. The environmental impact evidence was largely circumstantial yet in the US, it was appealing to scientific reason: TBT was a man-made chemical, it followed the pattern of DDT, the environmental movement of the 1970s was waning, marine environmental and ecosystem research funding was drying up, and it stirred the pot.

Early concern was expressed that most of the evidence the regulatory process considered to be significant came from bivalve mollusks: (1) it was believed that mollusks were more sensitive than other animal groups to TBT; (2) many bivalves have a cosmopolitan distribution and are commonly maintained in the laboratory; (3) filter-feeding bivalves may be more susceptible to TBT due to their feeding strategy; and (4) many bivalves have an economic importance in the commercial shellfish industry (Champ and Seligman, 1996b; Champ and Seligman, 1996c). A significant and subtle distinction that needs to be kept in mind is the difference between the environmental impact of TBT on the shellfish industry and the environmental impact of TBT on natural shellfish populations. The point is that the effects on cultured shellfish do not necessarily demonstrate similar ecological effects in a typical natural situation. A second point is related to public definition of 'acceptable' land use. It is difficult to appreciate being interested in culturing shellfish in areas adjacent to marinas and shipyards given their history of being defined as 'polluted' due to acute and chronic contamination problems (Champ, 1983). Ports, harbors, and marinas are publicly approved marine land uses. These facilities are usually located in highly protected areas with low flushing rates, long water mass retention times, oil spills, high levels of contaminants, and high silt loads which are not optimum conditions for culturing filter-feeding bivalves.

In Europe, the critical evidence for the initial regulations in the mid 1980s, was associated with shell thickening in oysters *(Crassostrea gigas)* and imposex in dog-whelks *(Nucella lapillus).* In the US, the early critical evidence was associated with laboratory studies that reportedly demonstrated unacceptable effects on growth and development in oysters (*C. gigas, Ostrea edulis)* and clams *(Mercenaria)* (Champ, 1986). All of this evidence was based on only four species, a similar number of laboratory tests and field observations, generally unsupported by chemical measurements and not published in peer reviewed journals. In general, the laboratory studies utilized questionable methodology and field studies lacked the necessary scientific rigor. The regulatory process and need for regulatory data and information drove everything (Champ and Bleil, 1988).

In the US, regulatory offices do not have funds for independent research and monitoring of regulated chemicals. Instead, they solicit data and information through data-call-in notices (DCIs), as part of the permit application process from product manufacturers and other interested parties. They also review research findings if they are available. In the mid 1980s, the information requirements of the regulatory process for TBT monopolized many research resources in an attempt to get the information needed for policy and decision-making. In essence, for TBT, there was an abundance of scientific information that was not quantitative or good science in predicting environmental effects. This forced the regulatory process to be more conservative because of the abnormally high scientific uncertainty in the data (Salazar and Champ, 1988). A thorough independent, neutral international scientific-peer reviewed debate on TBT has not occurred. The same situation exists today, but many of the scientists that have recently questioned the data being used in the regulatory process, do not have funding through a third party independent process. Therefore, their questions and concerns maybe perceived as pro TBT and not pro good science because some of their funding for the studies published in ORTEP (1997, 1998); ORTEP (1999) were provided by chemical manufacturers and paint companies that manufacture TBT. In addition, researchers needing more funding to conduct their studies have promoted or 'marketed' the findings of some preliminary studies. All of this is better stated in Sindermann (1982). The problem is that the policy and decision-maker in the regulatory process is forced to sift through the scientific controversy, not rigorous science.

4. Bioaccumulation of TBT from sediments

The results of what may become a classic regulatory text book debate and case study, are summarized in a US EPA Region 10 Technical Memorandum entitled: 'Topics Related to the Tributyltin Study at the Harbor Island Superfund Site, Seattle, Washington' (Keeley, 1999, personal communication). During EPA Superfund remedial investigations at the Harbor Island Site (Weston, 1994), TBT had been previously identified as a contaminant of potential concern due to elevated concentrations in the marine sediment (higher concentrations ranged from 10 to 50 ppm dry wt. TBT).

Because there are no established U.S. federal or state sediment quality guidelines or standards for evaluating TBT concentrations in sediment, the US EPA formed an interagency working group to identify and evaluate approaches to deriving an effects-based sediment cleanup concentration for use at Superfund sites in Puget Sound, Washington. Most of the available literature presented toxicity of TBT for water, and only two studies (covering four species) evaluated toxicity associated with sediment concentrations of TBT (US EPA, 1996a). The working group also proposed the calculation of an apparent effects threshold (AET) value, which could be used as a sediment criterion for TBT using available chemical (bulk sediment) and biological (sediment toxicity, benthic infauna) data from Puget Sound. The working group found that: (1) existing Puget Sound data did not support a clear identification of an AET value for TBT; (2) a maximum no-effect concentration could often not be established because, in several cases, the highest sediment TBT concentration was associated with no biological effects and was also the highest concentration measured among all the stations sampled; (3) good correlations were not found between bulk TBT sediment concentrations and laboratory toxicity and in situ benthic community responses; and (4) based on an evaluation of available information, ‘bulk sediment concentrations of TBT were a poor predictor of bioavailable TBT’ (US EPA, 1996a). Furthermore, the working group recommended, based on a general understanding of chemical partitioning and the lack of observed relationships between bulk sediment TBT and adverse ecological effects, that when TBT is a contaminant of concern in sediment, that pore water concentrations of TBT should be measured, and toxicity testing or bioaccumulation testing (in situ or laboratory) be conducted to confirm the ecological significance of concentrations measured in pore water. The working group did not provide recommendations for specific bioaccumulation test species, because it was believed that additional work needed to identify the most appropriate species (ESI, 1999a).

In a series of subsequent TBT-related studies, a consortium of Harbor Island waterfront property owners (the Port of Seattle, Lockheed Martin Corporation and Todd Shipyards Corporation) funded a study to evaluate the bioavailability of and the potential effects associated with TBT in sediments at the Superfund site. The overall purpose of this study was to develop a site-specific, effects-based TBT tissue trigger concentration that could be used to determine the need for remediation of TBT-contaminated sediments. In this study, effects considered relevant for the development of a site-specific tissue trigger value were mortality, reduced growth, and reproductive impairment. The normal TBT effects cited in the literature, such as bivalve shell thickening or induction of (early stage) imposex or intersex in meso- and neogastropods, were not appropriate in this evaluation; because (1) these biological responses do not have established connection to population- level effects; and (2) there is a lack of suitable habitat at the site for the species (oysters, mesogastropods, and neogastropods) typically affected by shell thickening, imposex and intersex. The study site is a deep (−30 to −60 foot MLLW), industrialized channel of subtidal sediments within the Duwamish River Estuary. Very little intertidal habitat is available, due to extensive channelization and dredging of the waterway, and no commercial or recreational shellfish beds occur. In addition, gastropods typically are not a large component of the benthic community at the site, and mesogastropods and neogastropods are very limited in abundance (ESI 1999a). The study was performed in accordance with a sampling and analysis plan (SAP), prepared by ESI (1998) that was reviewed and commented on by all reviewers prior to its approval by the US EPA, and resultant data from the TBT study were determined to be of high quality by EPA (ESI, 1999b).

The evaluation of TBT sediments from the Harbor Island sediments was conducted in two studies. First, a literature review was conducted to identify global paired tissue residue and effects data for marine invertebrates and fish (ESI 1999a).

The tissue residue data were used to estimate a site-specific, effects-based tissue trigger concentration for TBT (ESI 1999a). Second, sediment samples were collected throughout the study site for chemical and biological testing (ESI 1999b). TBT concentrations were measured in bulk sediments and pore water samples; a subset of sediment samples collected was used for bioaccumulation testing. With approval from all involved agencies and consistent with national guidance, bioaccumulation testing was conducted to determine site-specific exposures to two marine invertebrate species: (1) a bivalve (*Maconia nasuta*); and (2) a polychaete (*Nephtys caecoides*). No approved marine sediment toxicity bioassay protocols for test species that have demonstrated sensitivity to TBT were available (US EPA, 1996a), so no toxicity testing was conducted. The resulting tissue TBT concentrations were then compared to the effects-based trigger concentration derived from the literature (ESI, 1999b; Keeley, personal communication).

Results of this study were that the survival of the laboratory test organisms was high, and the lipid content of the organisms exposed to test sediments was similar to controls, which suggests to many reviewers of the project that the organisms were in good physiological health during the exposure period. A site-specific tissue trigger (3 mg/kg dry wt. TBT) was estimated (Meador, 2000) for the study site for evaluating bioaccumulation data from the study area, and for the 20 stations sampled and tested at the site, none of the tissue samples from the bioaccumulation tests exceeded the tissue trigger value of 3 mg/kg dry wt. TBT. Thus, no cleanup of TBT sediments was recommended. The value of 3 mg/kg dry wt. TBT, which was derived from paired tissue residue effects data in the literature, is estimated to be the tissue residue associated with reduced growth in a number of invertebrate species. The level is however, very similar to the overall geometric mean of paired effect/no-effect data and the estimate of a sublethal effects level based on a multi-species acute-to-chronic effects ratio for the study area. The development of tissue residue effects thresholds is part of EPA's overall strategy for management of specific contaminants in sediments in the US rivers and estuaries. The lack of TBT bioaccumulation from sediments in these studies is not understood, creating more unanswered questions and confusion in the data and suggests that further studies are needed prior to the development of a protocol for estimating TBT tissue level triggers for regulatory use. Results from the study also found that TBT tissue concentrations were most strongly correlated with dw-sediment and carbon-normalized sediment TBT concentrations, and there were weak correlations with filtered and unfiltered pore water TBT concentrations. If there is no relationship between levels in sediments and bioaccumulation levels in tissues, then the TBT in the sediments has been shown to not be bioavailable. For the determination of ocean dumping for dredged materials, the decision has to do with whether a species has accumulated more than 3 mg/kg dry wt. TBT.

After completing the Harbour Island TBT bioaccumulation studies, the US EPA (1999) prepared a technical memorandum to address topics of interest identified by EPA and other agency reviewers on issues related to the findings presented in the above study (ESI 1999c). Several scientists reviewing the results of the Harbour Island studies had a difference of opinion in the interpretation of the results. Some reviewers of ESI (1999b) indicated that the measured TBT bioaccumulation in test organisms for this project was less than they would have expected from the measured sediment and pore water TBT concentrations in site samples. This concern was based in part on a comparison of the bioaccumulation test results with studies reported in the literature and with other similar studies performed in the general Harbor Island area. Some reviewers suggested that several test parameters (e.g. species selection, exposure regime of tests, organism health) might have influenced the results.

Salazar and Salazar (1999a,b, in preparation) in reviewing the Puget Sound bioaccumulation studies believe that the major lesson learned from this study and their separately-conducted caged bivalve bioaccumulation studies are that lab tests do not predict nature very well, or adequately consider equilibrium and energetics. They have listed the following specific lessons learned from their research on TBT uptake by mussels: that (1) lab tests generally over-estimate toxicity; (2) lab tests generally under-estimate bioaccumulation; (3) bivalves are sensitive test species; (4) exposure period should be determined by equilibrium; (5) growth rate affects bioaccumulation potential; (6) quantifying health is important in data interpretation; and (7) tissue chemistry can be used to predict effects. Salazar and Salazar (1987, 1989, 1996) and Salazar et al. (1987) have found that survival and growth effects of TBT were over-estimated, based on laboratory tests and mesocosm studies. They placed caged mussels at the seawater intake to test tanks and found that growth rates were approximately four times faster outside the test tanks compared to growth in the control tanks.

In the Harbour Island studies, the issue is the interpretation of the tissue chemistry data. Salazar and Salazar (1996a,b, in preparation) believe that even though the EPA followed all state and national guidance and accepted state-of-the-art protocols, laboratory exposures have underestimated bioaccumulation levels due to poor animal health from test conditions. Meador (personal communication) suggests that *Macoma* in these tests were probably ventilating clean overlying water, reducing its exposure to TBT. Generally speaking, bivalves are extremely sensitive to food and flow rate and growth rates seldom if ever achieve the growth rates of animals in nature. Laughlin (1996) reported that BCF is related to growth rate and that the highest growth rates were associated with the highest BCFs. Laughlin referred to this as the concentration dependence of TBT accumulation. Widdows et al. (1990) found that the operative mechanism is that growth rate is also related to filtration rate. Laughlin (1996) measured BCFs of only approximately 5000 compared to an average of approximately 30 000 from Salazar (1989), Salazar and Salazar (1996) transplanted mussels, suggesting that Laughlin's animals may have been under severe stress. The 28 day exposure bioaccumulation tests in the Puget Sound Studies with the marine bivalve *Macoma nasuta* (which is a facultative feeder-both filter feeding and deposit feeder) did not reach steady state, when the test was extended to 45 days, and the results may have reflected test conditions in which *Macoma* may have been stressed. Originally, EPA proposed modifying the test procedure in accordance with Test Sediment Renewal (EPA Guidance Manual on Bedded Sediment Bioaccumulation Tests, EPA/600/R-93/183) which recommends complete sediment renewal for tests longer than 28 days. Bruce Boese (EPA Newport Laboratory, and an author of the manual) suggests that the primary reason for performing sediment renewal was to give the animals more 'food'. For the Harbor Island tests, it was decided to add 0.5 cm of sediment to the test chambers every 7–10 days for the entire test. Questioned in the study was also the use of lipid content at the beginning and end of the test, as a means of evaluating potential stress on the test organisms. Boese (personal communication) felt that the lipid content of *Macoma* does not give any information about the health of the animal, and that gain or loss of lipids is primarily related to reproduction.

Laboratory bioassays have become an environmental test industry and big business in making regulatory decisions. Their simplicity, cost and reproducibility are very attractive to regulatory policy and decision-makers. However, their scientific value or merit has been repetitively questioned. White and Champ (1983) addressed this issue of 'The Great Bioassay Hoax' and Salazar (1986) asked similar questions regarding the application of traditional laboratory toxicity tests to assessments of TBT. Salazar and Salazar have raised these questions to a higher level of sophistication but the old problems still remain. Scientists in the bioassay testing business hesitate to challenge an accepted regulatory test, because of a lack of a replacement, and the process to get one accepted, but still need to strive to develop standardized tests that validate and represent what an organism actually experiences in the environment.

Salazar and Salazar (1999a,b, in preparation) also feel that the other interesting issue here is that they believe that the *Macoma* bioaccumulation test may be flawed for the following reasons: (1) since the ASTM protocols do not require any effects measurements, one can never be sure of the health of the test organisms; (2) the largest and slowest-growing animals generally have the lowest tissue concentrations in transplant studies; and (3) people tend to forget that *Macoma* is a facultative deposit feeder, and can either filter- or deposit-feed. Recent summary papers have reported that many benthic invertebrates are quite plastic in their feeding mode and readily shift back and forth from filter- to deposit-feeding depending on local environmental conditions and available food and can select between clean and filtered seawater and highly contaminated sediment.

Langston and Burt (in preparation) found that concentrations in tissues of *Scrobicularia plana* (a deposit feeding clam) in the UK reached equilibrium in tissues after 40 days of exposure. They also reported that sediments are an important vector for TBT uptake in deposit-feeding clams. They also concluded that it is particulate rather than desorbed TBT, which is most significant. Laughlin (1996) reports that bioaccumulation factors appear to be high, but field studies, in particular, have not necessarily carefully characterized the route of uptake (water or food).

Salazar and Salazar (Personal communication, 1999) have found numerous examples where bivalves have been the most sensitive test species. Their predicted tissue burden for effects in mussels is an order of magnitude lower than that for amphipods based on the work of Meador (1997 and references cited therein) and others. Theory suggests that tissue concentrations for effects should be relatively constant across species and that appears to be true for particular endpoints like growth. The problem is that it is relatively difficult to measure growth rate in an amphipod. The difference in sensitivity is due to the growth rate endpoint in bivalves and the mortality endpoint in amphipods which theory suggests is about an order of magnitude different (McCarty, 1991; McCarty and Mackay, 1993). An additional problem with most laboratory tests is that they were not originally selected and standardized by equilibrium kinetics and steady state.

Amphipod tests are routinely conducted for only 10 days, even though Meador (1997, 2000) has found that it takes approximately 45 days to reach chemical equilibrium or steady state. This may explain why there appears to be a disconnection between sediment chemistry, laboratory toxicity tests, and benthic community assemblages using the sediment quality triad. This has led to suggestions of using tissue chemistry to predict effects (McCarty, 1991; McCarty and Mackay, 1993). Subsequently, Salazar and Salazar (1991, 1998); Salazar and Salazar (submitted) developed the exposure–dose–response triad that relies on tissue chemistry to make the link between the various effects endpoints. This relates to Salazar's point of growth rate affecting bioaccumulation potential. Sick and dying animals do not accumulate much TBT, which is why it is essential to confirm the health of the test animals.

With TBT data, they have been able to predict where effects will occur based on where the relationship between water or sediment and tissue TBT begins to change. This was first demonstrated in a graph published in Salazar and Salazar (1996) that plotted the relationship between water and tissue TBT. They found that grouping the data above 105 ng/l gave one regression and at 105 ng/l or lower that it gave a very different regression. The Salazars recently replotted the Langston and Burt (1991) data and found exactly the same relationship, which Langston concurred. With Langston and Burt's data, they found effects in *Scrobicularia* to occur between 0.1 and 0.3 μg/g TBT dry wt. in sediment, which agrees with Meador's data for effects on the polychaete *Armandia brevis* (Meador and Rice, in press). The Salazars summarized their findings in a paper presented at the SETAC (1999) meeting in Philadelphia. This paper is being expanded to emphasize the significance of field data over laboratory data in predicting effects and will be submitted to the *Journal of Marine Environmental Research*. They concluded that these data sets: (1) support their hypothesis that one can predict the concentrations where effects will begin to occur based on the relationship between external concentrations and tissue burdens; (2) demonstrate that the concept may work for both water and tissue; and (3) suggest that tissue burdens associated with effects (acute 10X > chronic) are relatively constant across marine organisms.

5. Monitoring and research trends

In reviewing a manuscript by Law and Evers (submitted) entitled 'The environmental distribution and effects of tributyltin — an update to mid-1999' it occurred to me that perhaps several hundred papers had been published about TBT in the decade post most national regulations (1998). This caused me to put together Table 1, which is a listing of journal papers from my files, and omissions are apologized for. The table has been organized to focus on the 'so what' question. Each paper is listed first by author and title. They are grouped first as monitoring papers, next bioaccumulation papers, and then the imposex/effects/impacts papers. In reviewing the titles of this list of papers, it occurred to me that we still have not got to the science of TBT. Has the scientific community answered the question — should TBT be banned? Have we provided regulators with the kind of data and/or information that is needed to make the correct or best decision given public interests? Has a public environmental problem with a $1 billion annual benefit been addressed with the appropriate funded level of scientific studies? If we have imposex some fixed distance adjacent to each port or waterway is this an acceptable land use decision?

6. The decade following 'national' regulations — decline in environmental concentrations

After national regulation in the late 1980s, the number of studies on TBT and number of papers from the regulated countries appeared to decline. The August 1994 issue of the *Marine Pollution Bulletin* (MPB) had an article entitled 'TBT on the way out'. It reports that the MEPC/IMO Resolution of 1990 has proven successful. The article cites the European Council of Chemical Manufactures' Association (CEFIC) findings that in all regions surveyed (in Japan, UK, US and Germany) that TBT in water and in marine organisms has been reduced and that high levels were only found in some harbors and in the vicinity of some shipyards and docks.

In the February 1995 issue of MPB, a paper on TBT pollution in coastal areas of Ambon Island (eastern Indonesia), then one in Irish waters, and in August one on TBT in Sydney Rock Oyster from the Hawkesbury River Estuary, NSW, Australia. In 1996, three papers were published on TBT from (Icelandic waters, New Zealand, and the northeastern Mediterranean), and researchers were finding occurrence and accumulation of butyltin compounds in fish from certain Asian and Oceanian countries (see Kannan et al., 1995, 1996a,b). In 1998, a paper was published in the MPB on TBT occurrence in waters off the Polish coast of the Baltic Sea (Poland and Eastern Europe) (Kannan and Falandysz, 1997). A discussion of this data is presented in ORTEP (1997), Green et al. (1997) with a reply in Kannan and Falandysz (1999).

In reviewing the literature in the decade following the adoptions of national regulations, three conclusions are readily apparent: (1) unfortunately during the period following national regulations, there was a transfer of the painting of TBT on ocean going vessels from the major regulated countries to less and non-regulated countries; and (2) in reviewing Table 1, it is apparent that the occurrence of imposex in dogwhelks dominated the literature in the early and mid 1990s; and (3) we have still failed to provide a sound scientific basis for the regulation of antifouling marine coatings.

6.1. In the US — a decade later

In the United States, since the passage of the Antifouling Paint Control Act of 1988, the environmental concentrations of organotin compounds have declined (Seligman et al., 1990; US EPA, 1991; Valkirs et al., 1991; Wade et al., 1991; Huggett et al., 1992). Three national and regional monitoring programs in the US have sampled for TBT since the passage of OAPCA in 1988. These are the US National Oceanic & Atmospheric Administration's (NOAA) National Status and Trends Monitoring (NS&T) Program, which was created in 1984 (see O'Connor, 1998). Overviews are presented in the Proceedings of the Coastal

Zone 93 and the special issue of MPB Vol. 37 No. 1 (O'Connor and Pearce, 1998) and the papers therein. A second TBT monitoring program was the US Navy long-term monitoring program associated with Navy home ports and harbors (see US Navy and EPA, 1997). The third monitoring program is the consortium of tributyltin manufactures (ORTEPA) long-term monitoring program contracted to Parametrix Inc., with results published in Cardwell et al. (1999a) ORTEPA (1997, 1998). The results of these three national monitoring programs have been compared by Russell et al. (1998) who found that all of these programs have found declining environmental concentrations of TBT over time since the enactment of OAPCA in 1988. The water concentrations have declined 56–71%, sediment 47–55% decline, and bivalve tissues 40–82% within a few years. Mean TBT concentrations in water are generally below the current US EPA marine chronic water quality criterion of 10 ng/l (Russell et al., 1996).

Studies have found that mean TBT surface water concentrations have significantly decreased in San Diego Bay, following legislative restriction on the use of organotin antifouling paints in California. Regression analysis of the San Diego data suggests that surface water concentrations would decrease by 50% in 8–24 months. It was found that sediment TBT concentrations in San Diego Bay did not reflect recent decreases in water column values and were variable among stations over time, and that tissue concentrations in *Mytilus edulis* have generally declined in San Diego Bay since February 1988 (significantly since April and July 1990), Valkirs et al. (1991).

Similar findings have been reported for the Chesapeake Bay by Huggett et al. (1992) for the Hampton, Virginia area of the bay. Surface water samples analyzed after the passage of the Organotin Antifouling Paint Control Act (OAPCA) of 1988 in marinas and yacht clubs indicated that TBT concentrations had significantly decreased when compared to results of earlier studies by Hall (1986, 1988), Hall et al. (1986, 1987), Huggetts (1986, 1987), Huggett et al., (1986); US EPA Chesapeake Bay Program (1987).

It is interesting to note that in the FY 97 Defense Appropriations Bill, congressman Bateman from Virginia (in September 1996) inserted some language requiring the Navy to reassess the discharge levels of TBT from drydocks. In so doing he revitalized the TBT debate in the US. Discussions with his staff indicated that this interest is due to the major shipyards in his region of the state having an economic interest to again apply TBT antifouling paints, even though it was Virginia shipyards that were originally supportive of the regulations back in 1988. Currently Virginia shipyards paint 10 or so cruise ships a year with organotins. It is interesting to note that ships can go up Chesapeake Bay (through the State of Virginia) to Shipyards in the Port of Baltimore and be painted with TBT (with considerable cost savings) because the state of Maryland does not have a discharge limit on TBT from Shipyard wastewaters that would require the expense of treatment of TBT in discharges.

In addition, EPA noted that the use of copper is coming under increasing regulatory pressure with some coastal states restricting the amount of copper that may be discharged into local harbors during hull cleaning and washing. These regulations may impact the US Navy's use of copper in antifoulant paints and leave the Navy without alternatives that meet their requirements. The Navy has funded the development of in-the-water cleaning systems for copper that also collect all waste and wastewater for treatment (Bohlander and Montemarano, 1997). It also should be noted that both Holland and Sweden have recently introduced regulations on antifouling paints for pleasure vessels containing copper effective 1 September 1999. Canada has set the release rates of copper in antifouling paints at 40 mg/cm^2 per day. Copper is a potential toxin to marine organisms (Lewis and Cave, 1982; Goldberg, 1992). It should also be noted that the US Department of Defense and the US Environmental Protection Agency have been working on the Uniform National Discharge Standards (UNDS) which will regulate the amount of biocidal discharges from antifouling coatings into the sea by December 2000, with the current release rates under consideration for copper less than the 40 mg/cm^2 per day. (see UNDS website: http://206.5.146.100/n45/doc/unds/SITEMAP/ITEMAP.HTML).

The EPA report to congress (US EPA, 1996a) is a summary of the status of development of alternatives to TBT. The driving force is to develop an alternative to TBT, which could compete in the US$500 million/year total antifoulant paint market (C&E News, Oct 14, 1996). The TBT copolymer used in deep-ocean going vessels represents between 65 and 70% of this market. The goal is to develop a non-toxic (no effect on non-target organisms) antifoulant, which effectively inhibits the formation of biofilms and prevents biofouling. The major finding of the EPA, 1996 Report (which has not been updated) was that 'an alternative antifoulant as effective as TBT self polishing copolymer paints has not been found'. They also reported that the principal alternatives today to TBT antifouling paints are copper-based. However, hulls treated with copper-based paints were reported to foul within 15–18 months due to formation of a 'green layer' on the surface of the hull. The green layer is the reaction of copper to seawater, which results in the formation of a coating of insoluble cupric salts, preventing the release of copper from the paint underneath. Once the green layer is present, the antifoulant protection is no longer effective. Underwater hull scrubbing is required to remove the green layer and attached fouling organisms and with frequent scrubbings, the period of protection can be extended for up to 30–36 months depending on water temperatures. Revised estimates on fuel savings from the use of TBT by the Navy ranged from 18 to 22% of the total fuel consumption (US EPA, 1996b).

6.2. In the UK — a decade later

During 1985, the UK government took action under the Control of Pollution Act of 1974 to regulate the use of TBT antifouling paints on small vessels and set an environmental quality target (EQT) concentration for TBT at 20 ng/m^3 per day (see Abel, 1996 for further regulatory discussion of these deliberations). Subsequent studies by UK researchers during the next summer (Cleary and Stebbing, 1985; Waldock et al., 1987a) found that in the past several years, organotin concentrations increased in the spring with the launching of small boats and yachts, usually followed by a secondary peak in later summer or autumn associated with repainting or hosing off activities and that concentrations declined during the winter. These studies suggested that the EQT needed to be reduced by a factor of 10 to achieve environmental protection. As a result of these studies in February 1987, the UK government announced its intention to introduce further controls under the Control of Pollution Act. This included complete bans on retail sale of TBT antifouling paint formulations and on the sale of products containing TBT used to treat fish farm cages.

Studies subsequent to this second regulatory action carried out by researchers at MAFF have found significant concentrations of TBT in harbors and at anchorages in a study that focused on large vessel contributions. They also found that dry-docking practices and illegal use have result in discharges of hazardous concentrations of TBT (Waldock et al., 1988). Reductions in concentrations or organotins in estuarine surface water and sediment concentrations in England and Wales followed the 1987 legislation (see Waldock et al., 1987a; Waite et al., 1991, 1996; Dowson et al., 1993a). However, surface water TBT concentrations in many areas exceeded the new EQT of 2 ng/l, and studies in new marinas suggested that the higher than expected concentrations may have resulted from illegal use of TBT by boat owners. Dry docks in these studies were also singled out as a major source of TBT to estuaries.

6.3. Global environmental concentrations — a decade later

TBT concentrations in water, sediment, and biota have generally declined. Evans (1999b) has an excellent summary paper on the concentrations and environmental effects as a measure of the effectiveness of national regulations. TBT concentrations in surface marine waters have declined in Arcachon Bay, France (Alzieu et al., 1986, 1989) and in the UK (Cleary, 1991; Waite et al., 1991, 1996; Dowson et al., 1992, 1993a,b;

Dowson et al., 1994) the USA (Valkirs et al., 1991; Huggett et al., 1992, 1996; Uhler er al., 1993) and in the Gulf of Mexico from Wade et al., 1991; Garcia-Romera er al., 1993; Champ and Wade, 1996) and Australia (Batley et al., 1992). Tissue concentrations in molluscs have declined (Valkirs et al., 1991; Wade et al., 1991; Waite et al., 1991, 1996; CEFIC, 1994; Champ and Wade, 1996).

Exceptions to this general decline of TBT in bottom sediments have been reported as hot spots associated with ship channels, ports, harbors, and marinas in Galveston Bay (Wade et al., 1991), Hong Kong (Ko et al., 1995), the Netherlands (Ritsema et al., 1998), Iceland (Svavarsson and Skarphédinsdóttir, 1995) and in Israel (Rilov et al., 1999).

Oyster culture has recovered in France (Alzieu, 1991; Alzieu, 1996; Alzieu et al., 1986, 1989). In southern England, Waite et al. (1991, 1996), Dyrynda (1992) reported improved oyster culture. For Australia, Batley et al. (1992) have reported improvements in oysters. Minchin et al. (1987) have reported improvements for scallops and Minchin (1995) for flame shells in Ireland.

The literature has also reported widespread decline in imposex and population recovery for dogwhelks (*Nucella* spp.): England (Evans et al., 1991; Douglas et al., 1993; Gibbs and Bryan, 1996a,b) Scotland (Evans et al., 1994, 1996; Nicholson et al., 1998) Ireland (Minchin et al., 1995) Norway (Evans et al., 1996) and Canada (Tester and Ellis, 1995; Tester et al., 1996).

6.4. Exceptions to the declining TBT data

A set of environmental data from a study of surface microlayer does not share the similarities of the decline in organotin concentrations following national regulation. These studies were part of the 1990 Bremerhaven Workshop on Biological Effects of Contaminants and measured TBT concentrations from the German Bight to the North Sea (Stebbing and Dethlefsen, 1992). Hardy and Cleary (1992) found a zone of surface water with contaminant levels exceeding UK water quality standards (EQS) extended from 100 to 200 km offshore. Surface microlayer TBT concentrations (> 20 ng/l) were 10 times higher than needed to induce imposex in dog whelks. A high occurrence of fish egg and larval fish abnormalities were found in this region by Dethlefsen et al. (1985). It was concluded that these high levels of microlayer contaminants could pose a threat to fisheries recruitment in the North Sea (Hardy and Cleary, 1992). It was concluded that this is the first time toxic concentrations of any contaminant have been found in the open ocean with the implication that this type of pollution is from ocean going ships and may be occurring in oceans throughout the world (Coghlan, 1990).

For an example of the uncertainty in the data and information, see Salazar and Salazar (1998) transplant studies using mussel (in situ field bioassays) in San Diego Bay. In this paper, the authors have re-evaluated growth from in situ exposure tests to water column background levels of TBT and found that the predicted tissue concentration for probable effects on mussel growth should be lowered from 7.5 to 4 μg TBT/g tissue dry wt., suggesting that the predicted ecological risk assessment prepared by the Navy (1997) for TBT 'probably underestimated the risk'.

The Japanese submission to the NIEPC 42 Correspondence Group (NIEPC 414NF.3) reports that since 1990, the use of organotin compounds has been practically prohibited by government regulation and the voluntary restriction by the industry. Nevertheless, the main source of high levels of TBT in Japanese waters today is considered to be international ships. The Japanese study correlated marine ship traffic (number of ships) to TBT levels in waters and sediments in waterways, ports and harbors (without normalizing the data for dilution volumes, water retention times, mixing, etc.) and determined that the high incidence (counts) of ocean-going vessels was the source of TBT. Imposex was found in over 100 species of sea snails. They also report that in 1995, TBT and TPT concentrations in all fish and shellfish tissues were below the provisional ADI. Their report is among the first to assess the impact of TBT in the deep sea and in particular TBT levels in squid livers. Squid livers from the open ocean off Japan were found to accumulate TBT 48 000 times ambient concentrations, suggesting that TBT bound to particulate matter through sinking is the source and pathway to the deep ocean. It is interesting to note the absence of any references in the literature to shell thickening in oysters in Japan, the original home of *Crassostrea gigas*.

7. Impact of current regulatory policies and practices

The impact of current regulatory policies and practices can be assessed in the following ways: (1) loss of military benefits; (2) loss of economic benefits; (3) loss of operating benefits; (4) loss of individual ship costs and benefits; (5) loss of environmental benefits from use of TBT; and (6) the subsequent shift of TBT application and contamination to non-regulated countries.

7.1. Military benefits

Over the past 200 years, naval fleets with superior hull antifoulings have often proved more effective in combat. Some examples include:

- Nelson's victory of the French fleet at Trafalgar in 1805. The British fleet was 'copper bottomed' and foul-free; the French fleet has heavily fouled and hence less maneuverable.
- In World War II, US naval antifouling technology was more effective at controlling fouling than that used by the Japanese. This advantage provided the US fleet with a significant fuel efficiency and subsequent operating range over the Japanese fleet.
- During the Falklands War of 1982, the cruise liner, Queen Elizabeth II, was converted to troopship status in a few days. Thanks to her organotin copolymer antifouling bottom paint, she required virtually no hull coating work prior to dispatch to the Falklands, and arrived there ahead of schedule.

The economic benefits to navies using TBT copolymers have not only increased combat performance but also include the following:

- extended in-service deployment periods of 5–7 years between drydockings vs. 24–30 months at present. Improved ship's operating readiness, which is a critical factor in a time of national emergency, enabling ships to be available on short notice for deployment with clean, foul-free hulls, without requiring drydocking to remove fouling or to repaint hulls;
- increased operating range, which is important in distant tropical waters such as the Indian Ocean, Persian Gulf, and South Pacific;
- maintenance of top vessel speed capabilities and lower fuel consumption during extended high-speed operations, such as the 40 knots needed for the launching of aircraft from aircraft carriers;
- elimination of costly and time-consuming underwater hull cleaning to remove fouling during deployment. Copolymers 'polish' and 'smooth', providing the possibility of reducing underwater hull noise; and
- application to underwater advanced sonar and electronic communication and defense systems.

In 1985, the US Navy calculated that if the entire fleet (600 ships) were to be painted with TBT antifoulant paints, the fuel avoidance costs (extra consumption) would exceed $130 million annually (calculated with fuel costing approximately $18/barrel) (NAVSEA, 1986). Because of improved copper-based antifouling coatings, more recent estimates have reduced this cost avoidance estimate. However, today's $40 price for a barrel of oil drives this cost to over $\frac{1}{4}$ Billion USD. Also, additional costs that are difficult to estimate because they vary significantly for different oceans are costs from operational activities for fouling reduction such as increased underwater cleaning and dry dock costs for repainting every 18–30 months for non-organotin-based paints.

The use of tributyltin antifouling paints on commercial ships, fishing vessels and private boats in the United States could add another $300–$400 million (or 2 billion gallons) in fuel savings annually. Moreover, these estimated cost savings do not include the savings from decreased wear on propulsion machinery and down time for hull scraping, cleaning, and painting resulting from the use of organotin paints.

8. Economic benefits

In the US, the major manufacturers of organotin antifouling paints (M&T Chemicals, Inc. and International Paint Company) predicted that the US regulation of organotin compounds in antifouling paints would have the following additional negative impacts:

- deep sea vessels would go to foreign ship yards for painting;
- higher antifoulant protection costs to vessel owners;
- higher transportation costs;
- domestic vessels would have a dramatic increase in operating costs;
- severe hardship to US shipyards (125 000 workers);
- maintenance and repair declining now;
- TBT ban would push many ship yards over the edge, and foreign vessels, and ship yards would capture market;
- more than 70% of world's fleet uses organotin copolymers;
- national defense and military preparedness;
- extended drydock intervals; and
- TBT painted hulls would still be in US waters (modified from Gibbons, 1986; Ludgate, 1987).

Fouling creates roughness on vessel hulls due to the growth of aquatic plants and animals. This roughness increases turbulent flow and drag, reducing vessel speed per unit of energy consumption (Milne, 1990a). A 10-μm increase in average hull roughness creates between 0.3 and 1.0% increase in fuel consumption. Fuel is the largest single cost in operating a ship. For bulk carriers, fuel costs can be 50% of the total vessel operating costs. In 1985–1986, the fuel bill for the Queen Elizabeth II was $17 million.

At the 30th session of the Marine Environmental Protection Committee (MEPC) of the International Maritime Organization (IMO), A. Milne of COURTAULDS NCT, presented a paper entitled 'Cost/Benefit Analysis of SPC Organo-Tin antifoulings' (Milne 1990b). In his study, he considered the vessel as an industrial plant, and time in drydock and associated delays constituted expensive 'down time' for loss of vessel revenue. His analysis was framed around the following categories: direct fuel savings (1976–1986) extension of drydocking interval, improved plant utilization, capital savings, and antifoulings and the environment. The results of his study are presented below:

- the marine transport industry burns 184 million tonnes of fuel per annum; at \$100/t, the fuel bill is US\$$18.4 \times 10^9$.
- The cost of not having fouling protection was approximately 40% or 72 million tonnes of oil per year. It should be noted that this is greater than 60% of the 1990 North Sea oil production.
- The cost of fouling failure for oil tankers was estimated to be \$500 000 per 200 000 dry wt. tonnage (DWT) vessel per annum, estimating this failure to occur beyond 14 months.
- Self-polishing antifouling copolymers of TBT introduced in 1974 were estimated to provide the world fleet with an improvement in fuel efficiency of 2% with a 'very conservative' estimate of 2% savings from fouling for a total of 4% in power and fuel equivalent to 7.2 million tonnes of fuel or \$$0.7 \times 10^9$ saved annually.

In terms of antifouling performance in the 1970s, Milne reported that the drydocking intervals were: industry demand was 30 months, achieved was 18 months, and guaranteed was 12 months. The self-polishing antifouling copolymers of TBT (for a sample of over 4000 vessels/annum) by 1986 had shifted the mean docking interval to 27 months. The tonnage docked per annum was estimated to be 280×10^6 DWT. The mean cost was estimated at \$10/DWT. The calculated savings in drydock fees were \$$20.2 \times 10^6$/year. His calculations for improved plant utilization were \$$409 \times 10^6$/year. Capital savings were estimated to \$$500 \times 10^6$/year. The sum of these gave an estimate of \$$2449 \times 10^6$/year in total savings to the world commercial fleet (over 6000 tankers).

In addition the use of organotin based antifoulings provided the following environmental benefits: a reduction of 23 million tonnes/year of green house gasses and a reduction of 580 000 t/year in acid rain. Milne (1990b) concluded that it was based on the above figures that the environmental impact of the continued use of organotins in antifouling paints needed to be assessed.

The Organotin Environmental Programme (ORTEP) Association in the Netherlands and the Marine Painting Forum in the UK summarized a number of technical papers presented to the IMO MEPC Committee meeting in November 1990 (MEPC 30) organized by the European Chemical Industry Council (CEFIC, 1992). This document revised the Milne's calculated cost savings. Using current fuel prices and operating practices, they added an estimate of $1 billion more dollars in cost savings due to indirect savings giving a total estimate of $2.7 billion/year of 'significant' economic benefits to the marine industry from the use of TBT copolymer antifouling paints. Milne (1996) included in his estimate costs for greenhouse gases and emissions.

The reader is encouraged to read the paper by Abbott et al. (this volume) which has a unique approach to estimating the above costs. The added fuel and operational costs for ship owners are significant to them. But the total costs which includes an estimate of the external costs from the use of less comparable (to TBT) alternative antifouling paints may be very significant to the general public and to the debate. These costs include impact from green house gases, sulfur emissions, invasive species, etc.

Recently a draft report has been released for review (Haas and Johnson, 2000) on 'encouraging superior alternative antifouling for recreational boats' from the University of California Sea Grant Program. The purpose of the study was to foster the development and use of superior alternatives to metal-based (primarily copper) antifouling coatings for recreational boats. This study was funded by several programs in the State of California and reflects public interest in California in shifting recreational boat owners from Copper-based antifouling systems to more environmental friendly alternatives. The report is an in depth and balanced review of the problem and has recommendations about the factors to be included in the decision-making process that are very relative to the TBT debate.

9. Operating benefits

In 1998, a study was funded by the ORTEP Association to estimate operating cost benefit estimates from the use of TBT in antifouling paints for deep-sea vessels (Damodaran et al., 1998). They conducted a comparative analysis of the costs of TBT self-polishing copolymer (SPC) antifouling paints and their alternatives. The evaluation included antifouling paint costs, dry-docking rates, clean hull fuel consumption, and fuel consumption penalties as a result of hull fouling and found that TBT SPC paints offer significant cost savings to the shipping industry, because their 5-year dry-docking interval reduces dry-docking costs and revenues lost while the ship is in dry-dock for cleaning and repainting. In addition, they found no data indicating that tin-free paints can match the performance in terms of efficiency as TBT SPC marine coatings. They also found that tin-free SPCs were 95–146% more expensive, and copper ablatives were 156–401% more expensive than TBT SPC due to higher dry-docking costs, revenues lost, paint costs, and in the case of copper ablatives, fuel costs (Damodaran et al., 1998). The study estimated annualized additional costs to the worldwide fleet of bulkers, container vessels, and very large crude carriers to be on the order of $500 million, if a 30-month tin-free SPC is substituted for a 60-month TBT SPC. If a 30-month copper ablative coating were substituted for the 60-month TBT SPC, the additional costs would be on the order of $1 billion/year. These estimates do not include environmental costs (Milne, 1990a,b), paint application, and hull surface preparation and waste disposal. In 1996, TBT SPC was reported to be used on 70% of the world fleet of approximately 27 000 ships (CEFIC, 1996).

10. Individual ship costs and benefits

A global ban on TBT without acceptable alternatives could: place shipowners at an undefined economic risk; double antifouling protection costs; increase fuel costs; increase yard service costs; increase ship operating costs; and decrease ship operating life time. Shipowners are faced with the problem of: finding comparable alternatives to TBT; testing and evaluation of comparable alternatives; and getting regulatory approval of alternatives comparable to TBT.

Recently, Bohlman (1999) of Sea–Land Corporation drafted a summary of sea–land experiences over the past 10 years with tin-free type antifouling hull coatings and reported that the suggested phase out dates proposed by MEPC 42 were not achievable. He reported that sea–land had not found tin-free types of antifoulants to be effective for more than 3 years, and that in most cases all vessels required regular cleaning after 2–3 years. After 3 years they repainted the nine ships that had been painted with tin-free paints with TBT. Bohlman (1999) also reported that regular cleaning costs of approximately \$6000–\$10 000 per cleaning, which was necessary about every 6 months, once the antifouling loses effectiveness. Thus, the typical annual cost for cleaning is approximately \$15 000–\$20 000/ship. The tin-free antifoulants that sea–land used lost effectiveness after approximately 2.5 years incurring an additional cleaning cost of 2.5 × \$18 000/vessel or approximately \$45 000/vessel if tin-free antifoulings were used instead of TBT when drydocking vessels on a 5-year cycle. He also reported for sea–land that the fouling between cleanings beginning after approximately 2.5 years with tin-free antifouling would cause a 3% increase in fuel consumption. This would result in an annual additional fuel cost of approximately \$60 000–\$90 000/vessel based on the average annual fuel consumption figures for their ships of \$2–\$3 million. In total, sea–land estimates its total additional costs from the use of tin-free paints over a typical 5-year drydock cycle to range from \$200 000 to \$270 000/vessel. Bohlman (1999) concluded that until reliable alternatives are proven, the uncertain benefits do not outweigh the costs and recommended that IMO delay the ban until alternatives have been proven to be effective.

11. Environmental benefits from the use of TBT in antifouling paints

Recent research has suggested that hull biofouling will be likely to play a much greater role in introduction of invasive (exotic) species following a global ban on the use of TBT in antifouling paints. The 10th International Congress on Marine Corrosion and Fouling (February, 1999) in Melbourne, Australia included two special sessions on invasive species transported on vessel hulls. Stephan Gollasch, from the Institute for Marine Sciences in Germany gave a keynote address on the importance of ship hull fouling as a vector of species introductions into the North Sea. Dan Minchin presented a paper on data and information from Ireland and Mary Sue Brancato presented data from the US (see also Brancato and MacLellan, 1999). Historically invasive species from the hulls of ships has been mostly an exotic marine algae and plants problem due to the speed and size of ships and poor water quality in ports.

Minchin estimated that 1.8 million marine organisms could exist on the hull of a severely biofouled vessel (Minchin, personal communication). However, after the introduction and use of TBT in the early 1970s, fouling on hulls was not considered a significant source problem for invasive species, because in general hulls were cleaner. Considering the coincidence of global climate fluctuations and the proposed global ban on the use of TBT, invasion of species via the biofouling community on fouled hulls of ships may eventually constitute a greater threat then those in ballast water (Minchin and Sheehan, 1999).

Minchin is also concerned that there is a correlation between ship hull hitchhikers and water temperature changes. Ships pass through rapid water temperature fluctuations while entering harbors and channels and ports from the open ocean. These sudden temperature swings may initiate spawning triggering invasive species introduction in ports and port channels. Populations could easily become established in the invaded US port because the US Clean Water Act has greatly cleaned up (reduced pollution) US ports over the years. In the past, the level of contamination in most ports has reduced the probability of the invading organism becoming established. With the movement to clean up ports and harbors worldwide, the risk of introduction has greatly increased. Minchin believes that the IMO must have available replacements that are as effective as TBT, in providing the same degree of protection to coastal waters from invasive species as TBT has for the past three decades. To ban it, we would face serious introduction of invasive species in the temperate environments. Their environmental impacts include changes in biodiversity, food webs, trophic levels competition, and the introduction of disease organisms and parasites.

It has been estimated that over 6000 species have been introduced in the US. The introduction of the lamprey eel and zebra mussel in the Great Lakes are examples of major invasive species. The zebra mussel has had detrimental effects on lakeside piers, industrial facilities and public beaches. Another example the European Green Crab (*Carcinus maenas)* has the potential to impact the $20 million crab industry in the state of Washington alone (Brancato, 1999). Additional examples of invasive species are the toxic Japanese dinoflagellates and the northern Pacific sea star, which have infested New Zealand and Australia. The American comb jellyfish has greatly impacted the anchovy industry in the Black Sea.

In his keynote address Stephan Gollasch reported on historical studies of invasive species in the North and Baltic Seas and compared vectors of introduction including ballast water and hull fouling from 200 ships. In the 1992–1995 time frame, Gollasch reported that most of the non-native species with the highest potential for establishment were from fouled hulls, with 53% of the marine exotic species found in the North Sea introduced by shipping and 98% of the hulls sampled revealed non-native species (Reise et al., 1999). Of the species connectable to shipping, 66% were introduced from the hull, 34% from ballast tanks. Gollasch, the second author of the Reise et al. (1999) paper is also a scientific advisor and member of the German delegation for the ballast water working group at the IMO's MEPC 43. He believes that IMO should consider the hull fouling dilemma in its assessment of the ban of TBT and balance the risk of introduction of invasive species harming local ecosystems with the environmental risks of TBT on non-target species in their decision-making process. He further said that a ban of TBT is; from the environmental perspective; absolutely necessary in order to protect the environment from unwanted negative effects of TBT due to its accumulation in non-target organisms. He has found that most of the species of high concern are transported in ballast water including cholera bacteria and phytoplankton algae causing harmful algal blooms, but he believes the risk of species introduction from ships hulls is increasing by the ban on TBT without having an environmentally sound and effective alternative method and without TBT it could be even worse (Gollasch, 1999, personnel communication).

12. Potential liability of the shipping industry, shipyards, drydocks and paint manufactures

If TBT is banned by an international treaty as proposed by MEPC 42, the future cost of removal of dredged material from harbors and waterways will probably increase significantly. An example of how regulation can increase disposal of dredged material costs is seen in the two alternatives available to the port of NY/NJ for immediate disposal of dredge spoils. The Mud Dump Site (located 3 miles offshore in the open waters at the mouth of the harbor) has been operational for many decades and has been the traditional disposal area and can accept Category I dredged materials. Category II and III contaminated 'spoils' have to be disposed of at an upland hazardous waste disposal facility, however, from 1977 to 1991, 90% of all NY/NJ dredge spoils were tested and classified Category I and only 1–2% were Category III. However, in 1991 the US EPA replaced the existing tests in the NY region and added new bioassay testing which altered Category I, II, and III determinations.

For Category II and III dredged material, the currently available alternative is upland disposal at a hazardous materials storage facility and none are available in the near vicinity. Howland Hook Terminal in Staten Island shipped 150 000 $yard^3$ of sediment via barge and rail to Utah at a cost of \$17 million or over \$110/$yard^3$. Traditional fees for dumping dredge materials at the Mud Dump Site are in the area of \$10/$yard^3$.

If TBT ('as perhaps the most toxic substance ever deliberately introduced to the marine environment by mankind') is banned by an international convention (it will be the first chemical by name to have its own convention or treaty) it could then be considered equal or more hazardous than Category III compounds. As such it might greatly increase the cost of disposal of dredged materials from most ports and harbors that are contaminated with TBT, because of its persistence and its universal distribution in bottom sediments of ports and ship channels.

An additional concern for the paint companies, shipyards and shipping industry may be that in the future that they have to bear the liability for cost contained dredging. It may be that the liability for the additional or special costs of dredging and disposal of TBT contaminated dredged materials from ports and ship channels might revert back in the courts to sources such as have the costs of health settlements from smoking in the courts. The impact of TBT contamination in port sediments on future shipping and port development is significant. For example, plans to dredge the river Tyne in Newcastle (UK) may be abandoned, because of extremely high TBT concentrations in river sediments, and the concern that organotins will desorb from particles on agitation during dredging and disposal of dredge material at sea (Hartl, personal communication). Approval for dredging is pending on the outcome of a survey being conducted by CEFAS, Burnham-on Crouch.

13. Shift of application to non-regulated countries

A consideration that should not be omitted here involves the forces (economics and regulations) that drive international maritime companies to look for cheap labor and cheap environmental laws in non-regulated countries for painting their vessels with organotin antifouling paints. The length exclusion (> 25 m) allows for the use of organotin compounds by large ocean going vessels and gives the worlds maritime fleet significant economic benefits. The regulatory logic for this exclusion is that since they spend most of their time at sea (except when anchored in estuaries awaiting port space or goods, etc., and or at the loading dock). Therefore, they should not contribute significantly to the critical environmental concentrations of organotin compounds in estuaries, or near coastal waters where sensitive species of mollusks reside.

Environmental scientists in non-regulated countries have begun to find deformities in oysters similar to those in Europe. They are aware that there has been a large increase in the number of vessels being painted with organotin based antifouling paints in local shipyards in their respective countries. The impact of *not* painting ships with TBT on the Hampton Roads economy has been estimated to be a loss of \$340.2 million and 2160 jobs (Godfrey, 1999, personal communication).

US Navy studies at Pearl Harbor, Hawaii conducted during painting and release of ships from dry docks, found that with appropriate environmental management practices, drydock effluents could be maintained at low nanogram per litre levels. The costs for this environmental protection were reported to be high. For simulation of effectiveness of improved dockyard practice see Harris et al. (1991). In essence, economics and regulation in the developed countries have shifted an environmental problem to the countries least able to address them.

14. Summary and conclusions

The history of organotin antifouling coating regulatory strategies (as reviewed in preceding sections) is an excellent example of how well intentioned public policy and regulatory strategies responding to concerns perceived by the public to be urgent often fall short of achieving long-term goals. The long-term economic and environmental public goals should be that vessels (regardless of length) need effective antifouling coating technologies and that this effectiveness not impact non-target organisms. The regulation of antifouling coatings is a genuine public policy concern because their selection influences the cost reflected in the price of vessel shipped common goods, food, energy, etc.

Most effective antifouling coatings today contain toxic additive substances known as biocides. Organotin compounds have been found to be the most effective biocides developed to date. However, as engineered today, they are too effective because they also endanger non-target organisms. The challenge is essentially a matter of designing a means of reducing or controlling the scope of their effectiveness or replacing them. As scientists in the TBT debate, we are trying to provide proof of cause and effect relationships to the *n*th degree for a chemical that is at the edge of our understanding. In this debate we lose sight of the level of proof needed for regulatory decision-making. Simply stated if an environmentally friendly or non-toxic alternative is available or can be developed then there is *no* further need for regulatory debate on the question of the science in the TBT debate. Perhaps the proposed ban is an attempt by the regulators to get the coatings and shipping industries interested in available alternatives (i.e. using regulatory pressure for 'redefining antifouling coatings'). If it takes a convention, it means that the alternatives are not as good or ready and/or that there is not an appropriate means of evaluating them in the time period proposed. The US EPA has used this strategy for years; perhaps this is global outreach.

The shift from high release rate paints such as free association to copolymer-based paints (and the development of self-polishing copolymer paints) to lower the concentrations of organotins in the environment was a step in the right direction. However, there are other additional technological advances that should be explored in developing an economically and environmentally sound regulatory strategy. For a regulatory strategy to be supportive of the creation of high technology chemicals and products, it must include the promotion of continued research and development to push these compounds to additional refinements that enhance environmental attributes and improve competitiveness in the global market place. It is easy for market dominance or regulation to have the unintended consequence of inhibiting economic development of products and technologies.

The current organotin regulatory strategies have several major shortcomings. First, national regulations may unfortunately focus on short-term national self-interests and may not represent a 'think globally, act locally' philosophy. The principal regulatory approach is to reduce organotin concentration in the local environment by reducing the concentration in the paint (or in the release rate) and in the concentrations discharged to the environment from shipyards. Setting regulatory environmental concentrations (water quality standards) to protect local coastal waters, nations are, in effect, encouraging shipping companies to take their antifouling repainting business abroad at the economic loss of domestic shipyards. US, European and Japanese shipyards cannot effectively compete in the non-environmentally-regulated marketplace, if, in addition to high labor and operational costs, they must also shoulder the expense of waste treatment and disposal of antifouling residues from removal of spent antifouling paints to achieve a regulated discharge (environmental water quality standard) level to protect local waters. Consequently, large vessel owners can enjoy the double cost benefit of being able to have their vessels painted by cheap labor without having to be responsible for environmental degradation and human health hazards (externalities) in non-regulated countries.

The ultimate long-term solution to the antifouling coatings problem is to come up with effective regulatory strategies that promote the development of new and advanced antifouling coating and technologies that are 'environmental friendly' as alternatives to biocides; i.e. which are not toxic to non-target organisms and are inexpensive to treat or degrade in shipyard waste treatment systems. The strategy needs to also cover costs for public education and environmental monitoring. In the UK, the small brochure: 'Don't Foul Things Up' was extremely effective in reducing contamination from small boats. The challenge to the scientific community and the coatings industry is to 'redefine antifoulings coatings' to eliminate the need for and use of biocides altogether, (see Swain, 1999 for further discussion). In its 1996 report to congress, the US EPA, identified many new alternatives to TBT, however, many were associated with some form of copper. In the 5 years since, paint companies have intensively investigated the development of alternatives to TBT. In Japan, for example, the Japanese Ship Research Association organized a committee (SR209) comprised of representatives from universities, national institutes of ship owners, paint manufactures, and biocide manufactures, which met over the past 3 years to review alternatives to TBT. They have nominated 17 alternatives as safer than organotin compounds for use as antifoulant coatings (Mikami, 1999, personal communication). What is lacking is an international standardized-comparative test and evaluation mechanism of the available alternatives by a neutral third party to expedite their use by the shipping industry.

A more equitable and independent process than current approaches would be to promote standardized international comparative testing and evaluation of environmental friendly alternatives. An internationally and independent standardized process could complement the regulatory approach in providing the best scientific data and information for intercomparison of all available antifouling marine products, coatings, technologies and systems to regulators and shipowners. The creation of a Marine Coatings Board (MCB) would combine the regulatory processes and the forces of the marketplace to work together to develop the most suitable alternatives and get them in the marketplace in the shortest-time period (Champ, 1999a, 2000). It would integrate requirements of regulatory bodies, shipowners and operators, coating manufacturers and others; develop comparable and standardized international test protocols; support the regulatory acceptance

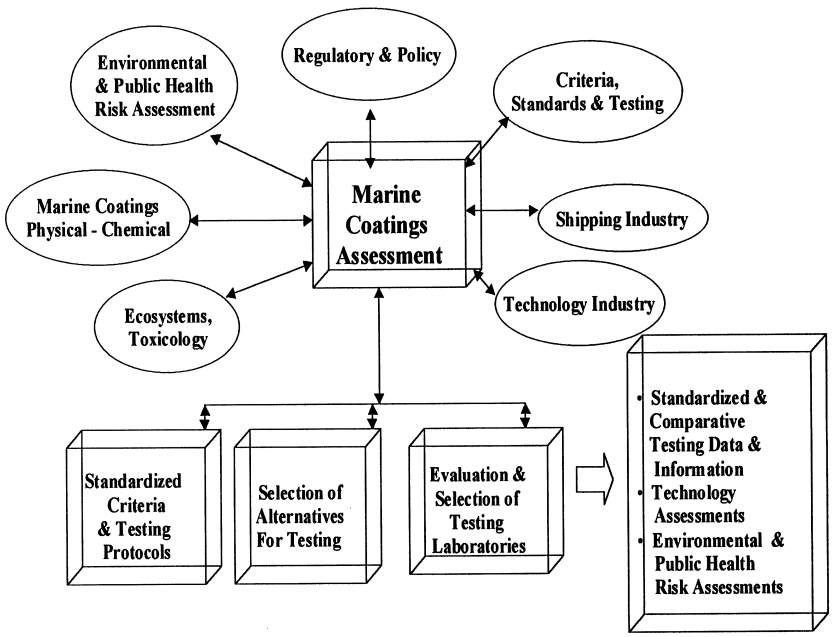

Fig. 2. Marine Coating Board structure and function.

process for alternatives. See Fig. 2 for an illustration of the components and activities that would be integrated.

15. Recommendations

An open, competitive, integrative, impartial process managed by a third party, neutral and independent organization (perhaps in cooperation with the class societies) is needed to support and complement the regulatory process. A Marine Coatings Board is needed so that the forces of the marketplace and regulatory process can work together to provide high quality, internationally standardized scientific data and performance information for environmental and public health risk assessments, benefits analysis and user-decisions for available alternative antifouling technologies.

15.1. Purpose

- Integrate the policy and regulatory requirements of the different nations into standardized MCB protocols.
- Develop a series of standardized assessment protocols through international expert working groups, which include performance, technical assessment, environmental and public health risk assessment, social and economic assessment requirements.
- Establish and fund comparative test and evaluation projects at international test and evaluation centers to provide data and information from the above series of standardized performance and technical assessment protocols.
- Provide short- and long-term toxic (acute and chronic exposure) data for assessment of environmental and public health risk assessments for available alternative antifouling technologies or products.
- Provide a fast track for the development and evaluation of acceptable alternative antifouling technologies.

15.2. Structure and organization

- The MCB would include stakeholders and interested parties and be managed by a neutral third party.
- The MCB would be formed to develop international standardized testing and evaluation protocols that would be reviewed and approved by a formal peer review process.
- The MCB would hold international peer review conferences and working group meetings (of international experts) to review and select available technologies for testing and evaluation.
- Alternatives would be identified and evaluated in international intercalibrated demonstration experiments utilizing scientific and regulatory criteria and standardized protocols developed by the MCB.
- The MCB would directly oversee the testing and evaluation of the most promising candidates. These would be bid out by request for proposals (RFP) to ship R&D groups, and industry and academic R&D laboratories across the world to conduct standardized assessments.
- The data and information from the MCB would be published on the internet and be available to anyone, anytime, anywhere.

A collective stakeholder consensus would guide the MCB in determining the most promising alternatives worldwide. The MCB would set up an intercalibrated experiment and bid out developed standardized test protocols to different ship R&D or academic labs across the world to conduct standardized comparative assessments. This would allow the regulatory process and the forces of the marketplace to work together to ‘comparatively’ evaluate the most suitable antifouling alternatives to TBT and toxic biocides and to get them into the marketplace as soon as possible. The above concept has been proposed *not* to compete or substitute for the regulatory process that national regulatory organizations conduct in reviewing and permitting the use of toxic and hazardous materials. Instead, its purpose is to complement their processes by providing them with the highest level of quality comparative data and information to support policy and decision-making in the shortest period of time. The deadline of the proposed IMO/MEPC 5-year phase out of TBT (as we know them) requires immediate action or ship owners may have to fall back to copper (which itself is facing regulation in many coastal waters) or alternatives that are not comparable or suitable (limited antifouling protection), higher fuel operation costs and more frequent drydocking intervals.

The anticipated costs of the MCB, its process, and its operation are trivial when compared to the potential cost savings to the shipping-related industries. The payback period is short, and the return-on-investment is quite high. For example, the MCB could spend $10 million/year in testing and evaluation for the next 5 years to provide data and information that are real solutions to the needs of the shipping industry for products and services. This amount would be less than 1–2% of the additional costs that Damodaran et al. (1999) have estimated as additional annual operating costs if TBT is banned without a comparable alternative. This $10 million/year for 5 years investment in the MCB by the shipping industry has a payback in preventing these estimated costs to the industry in the first 18 or 34 days of the beginning of the 6th year after the ban (depending on best or worse case data from Damodaran et al., 1999). So, the MCB is very cost-effective for both the shipping industry and the chemical and antifouling marine coatings technologies industry. The shipping industry is currently a very fragmented and divided business, owned by many different types of industries from banks and investments companies to shipping families. The MCB may also need start-up support of the coatings industry to redefine antifouling coatings.

In summary, 'national' regulations for TBT have worked in most regulated countries except in some ports and harbors where water circulation is poor or retention times are long (in Japan and in the oil offloading ports in Scotland), but they have shifted the problem to the unregulated countries. A total ban on the use of TBT has been recommended by many nations. Alternatives to TBT are available, but not proven and accepted on a global basis. Unfortunately, in the remaining less than 1000 days before the proposed IMO ban, an international independent process is not available to evaluate and select alternatives to TBT. The costs to shipowners for this failure have been estimated to range from $500 million to $1 billion annually. A third party, neutral, independent, international Marine Coatings Board has been proposed to complement the national regulatory process by providing the international standardized scientific data and information of the highest quality. The cost of the Marine Coating Board to evaluate available alternatives has been estimated to be less than $1/day per vessel in global commerce.

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References

Abd-Allah AMA. Occurrence of organotin compounds in water and biota from Alexandria harbors. Chemosphere 1995;30:707–715.

Abel, R. European policy and regulatory action for organotin-based antifouling paints. Chapter 2. In: Champ and Seligman (eds) In: Organotin: environmental fate and effects. Chapman & Hall. London. 1996:27–54.

Alzieu Cl, Héral M, Thibaud Y, Dardignac MJ, Feuillet M. Influence des peintures antisalissures à base d'organostanniques sur la calcification de la coquille de l'huître *Cassostrea gigas*. Rev Trav Int Pêches Marit 1981;45:101–116.

Alzieu Cl, Portman JE. Effect of tributyltin on the culture of

Crassostrea gigas and other species. Proceedings of the 50th Annual Shellfish Conference, 1984:87–104.

Alzieu Cl, Sanjuan J, Deltreil J, Borel M. Tin contamination in Arcachon bay: effects on oyster shell anomalies. Mar Pollut Bull 1986;17:494–498.

Alzieu Cl, Sanjuan J, Michell P, Borel M, Dreno LP. Monitoring and assessment of butyltins in Atlantic coastal waters. Mar Pollut Bull 1989;20(1):22–26.

Alzieu Cl. Environmental problems caused by TBT in France: assessment, regulations, prospects. Mar Environ Res 1991;32:7–17.

Alzieu C, Michel P, Tolosa I, Bacci E, Mee LD, Readman JW. Organotin compounds in the Mediterranean: a continuing cause for concern. Mar Environ Res 1991;32(1-4):2261–2270.

Alzieu Cl. Biological effects of trigutyltin on marine organisms. In: de Mora (ed). Tributyltin: case study of an environmental contaminant. Cambridge: Cambridge University Press, 1996:167–211.

Ariese F, Burgers I, Hattum B, van Horst B, van der Swart K, Ubbels G. Chemical monitoring Loswal northwest dumping location. Start situation 1996. Report R-97/05. Institute for Environmental Studies. Free University. Amsterdam. The Netherlands, 1997.

Ariese R, Hattum B, van Hopman G, Boon JP, Ten Hallers-Tjabbes CC. Butyltin and phenyltin compounds in liver and blubber samples of sperm whales (*Physeter macrocephalus*) stranded in the Netherlands and Denmark. Report W98/04. Institute for Environmental Studies. Free University. Amsterdam. The Netherlands, 1998:14.

Atkins WS. Assessment of the risks to health and to the environment of tin organic compounds in antifouling paint and of the effects of further restrictions on marketing and use. International Ltd. Draft Final Report to European Commission Directorate General 111. Volume A, February, 1998.

Bailey SK, Davis IM. Continuing impact of TBT, previously used in mariculture, on dogwhelk (*Nucella lapillus* L.) populations in a Scottish sea loch. Mar Environ Res 1991;32:187–199.

Bailey WA. Assessing impacts of organotin paint use. Proceedings Oceans '86 Organotin Symposium. Marine Technology Society. Washington, DC, 1986;4:1101–1107.

Batley GE, Fuhua C, Brockbank CI, Fleeg KJ. Accumulation of tributyltin by the Sydney rock oyster, *Saccostrea commercialis*. Aust J Mar Freshwater Res 1989;40:49–54.

Batley GE, Brockbank CI, Scammell MS. The impact of banning of tributyltin-based antifouling paints on the Sydney rock oyster, *Saccostrea commercialis*. Sci Total Environ 1992;122:301–314.

Batley G. The distribution and fate of tributyltin in the marine environment. In: de Mora, editor. Tributyltin: case study of an environmental contaminant. Cambridge: Cambridge University Press, 1996:139–166.

Bauer B, Fioroni P, Ide I et al. TBT effects on the female genital system of *Littorina littorea*: a possible indicator of tributyltin pollution. Hydrobiologia 1995;309:15–27.

Bauer B, Fioroni P, Schulte-Oehlmann U, Kalbfus W. The use of *Littorina littorea* for tributyltin (TBT) effect monitoring — results from the Berman TBT survey 1994/1995 and laboratory experiments. Environ Pollut 1997;96(3):299.

Bohlander GS, Montemarano LA. Biofouling, fleet maintenance and operational needs. Naval Res Rev DNR 1997;XLIX:9–18.

Bohlman MT. Personal communication. Sea Land Corporation. Charlotte NC 28209-4637. Letter to IMO/MEPC May 17 1999 5 p. Adopted by the Marshall Islands and Submitted to MEPC as MEPC 43/3/6, 1999.

Bosselmann, K. Environmental law and tributyltin in the environment. In: de Mora editors. Tributyltin: case study of an environmental contaminant. Cambridge:University Press, 1996:237–263.

Brancato MS, MacLellan D. Impacts of invasive species introduced through the shipping industry. Proceedings Oceans '99 International Symposium on the Treatment of Regulated Discharges from Shipyards and Drydocks. Marine Technology Society. Vol 4. Washington DC, In Press, 1999

Brancato MS. Personal communication. Parametrix Inc. 5808. Lake Washington Boulevard NE Suite 200. Tel. (425) 822-8880. Fax (425) 8898808. Kirkland, Washington 98033, 1999.

Cadée GC, Boon JP, Fischer CV, Mensink BP, Ten Hallers-Tjabbes CC. Why the whelk (*Buccinum undatum*) has become extinct in the Dutch Wadden Sea. Neth J Sea Res 1995;34:337–339.

Cardwell R, Brancato MS, Toll J, DeForest D, Tear L. Aquatic ecological risks posed by tributyltin in U.S. surface waters: pre-1989–1997 data. In: Champ, Fox, Mearns, editors. Treatment of regulated discharges from shipyards and drydocks. Published in the Proceedings of Oceans '99 Conference. Washington, DC: Marine Technology Society. ISBN No. 0-933957-24-6, vol. 4, 1999a:169–176.

Cardwell R, Brancato MS, Toll J, DeForest D, Tear L. Aquatic ecological risks posed by tributyltin in US surface waters: pre-1989–1997 data. Environ Toxicol Chem 1999b;18(3): 567–577.

CEFIC (European Chemical Industry Council). TBT copolymer antifouling paints: the facts. Document MEPC 33/INF. 14 for the 33rd meeting of the Marine Environment Protection Committee of the International Maritime Organization. London, 1992.

CEFIC (European Chemical Industry Council). Use of organotin compounds in antifouling paints. Results of TBT monitoring studies. Document MEPC 35/17 for the 35th meeting of the Marine Environment Protection Committee of the International Maritime Organization. London, 1994.

CEFIC (European Chemical Industry Council). Harmful effects on the use of antifouling paints for ships: a review of existing antifouling paints and the development of alternative systems. Document MIEPC 36/14/4 for the 36th meeting of the Marine Environment Protection Committee of the International Maritime Organization. London, 1996.

Champ MA. Etymology and use of the term 'pollution'. Can J Fish Aquat Sci 1983;40(2):5–8.

Champ MA. Introduction and overview. Proceedings of International Organotin Symposium. Marine Technology Society. Washington DC, 1986;4:i–viii.

Champ MA, Lowenstein FL. TBT: the dilemma of hi-technology antifouling paints. *Oceanus*. Woods Hole Oceanogr Inst 1987;30(3):69–77.

Champ MA, LW Pugh. Tributyltin antifouling paints: Introduction and overview. Proceedings Oceans '87 International Organotin Symposium. Marine Technology Society. Washington DC 1987;4:1296–1308.

Champ MA, Bleil DF. Research needs concerning organotin compounds used in antifouling paints in coastal environments. NOAA Technical Report Published by the Office of the Chief Scientist. National Ocean Pollution Office. 5 Parts Plus Appendices, 1988.

Champ MA, Seligman PF, editors . Organotin: environmental fate and effects. London: Chapman & Hall, 1996a:674.

Champ MA, Seligman PF. An introduction to organotin compounds and their use in antifouling coatings. In: Champ and Seligman, editors. Chapter 1. Organotin: environmental fate and effects. London: Chapman & Hall, 1996b:1–26.

Champ MA, Seligman PF. Research information requirements associated with the environmental fate and effects of organotin compounds. Chapter 29. Champ and Seligman, editors. In: Organotin: environmental fate and effects. London: Chapman & Hall, 1996c:601–614.

Champ MA, Wade TL. Regulatory policies and strategies for organotin compounds. Chapter 3. In: Champ and Seligman, editors. Organotin: environmental fate and effects. London: Chapman & Hall, 1996:55–94.

Champ MA. The tributyltin antifouling paint controversy. Sea Technol 1998;39(7):113.

Champ Michael A. Incorporating good environmental science in the current organotin regulatory debate. Editorial lessons learned. SETAC November Newsletter, 1999a.

Champ MA. The need for the formation of an independent, international marine coatings board. Mar Pollut Bull 1999b;38(4):239–246.

Champ MA, Fox TJ, Mearns AJ, editors. Treatment of regulated discharges from shipyards and drydocks. Washington, D.C.: Proceedings of the Special Sessions held at Oceans '99 in Seattle Washington, Sept 13–16, 1999. The Marine Technology Society. ISBN No. 0-933957-24-6, vol. 4, 1999:233.

Champ MA. An overview of the science and regulation of TBT and the potential for future liability for contaminated harbor sediments. In: Champ, Fox, Mearns, editors. Treatment of regulated discharges from shipyards and drydocks. Published in the Proceedings of Oceans '99 Conference. Washington, D.C.: Marine Technology Society. ISBN No. 0-933957-24-6, vol. 4, 197–212.

Champ MA. The organotin regulatory debate. Editorial. Maritime Reporter and Engineering News. January, 2000a:1.

Champ MA. The coating conundrum: incorporating good environmental science in the current organotin regulatory debate. The Maritime Reporter and Engineering News. 2000b;62(1):46–48.

Chau YK, Zhang S, Maguire RJ. Determination of butyltin species in sewage and sludge by gas chromatography-atomic absorption spectrophotometry. Analyst 1992a;117:1161–1164.

Chau YK, Zhang S, Maguire RJ. Occurrence of butyltin species in sewage and sludge in Canada. Sci Total Environ 1992b;121:271–281.

Chau YK, Maguire RJ, Brown M, Yang F, Batchelor SP. Occurrence of organotin compounds in the Canadian aquatic environment five years after the regulation of antifouling uses of tributyltin. Water Qual Res J Can 1997a;32:453–521.

Chau YK, Maguire RJ, Brown M, Yang F, Batchelor SP, Thompson JAJ. Occurrence of butyltin compounds in mussels in Canada. Appl Organomet Chem 1997b;11:903–912.

Cleary JJ, Stebbing ARD. Organotin and total tin in coastal waters of southwest England. Mar Pollut Bull 1985; 16(9):350–355.

Cleary JJ. Organotin in the marine surface microlayer and sub-surface waters of southwest England: relation to toxicity thresholds and the UK environmental quality standard. Mar Environ Res 1991;32(1-4):213–222.

Coghlan A. Lethal paint makes for the open sea. New Sci. 8 December, 1990:16.

Colin R, Chaumery CJ, Guermeur EI. Organo-tin concentrations in Brest naval port, in 1993 and 1994. Ecorade: the Bay of Brest: its state of environmental health. Paris-France Inst-Oceanogr 1997;73:17–24.

Commonwealth of Virginia. State Water Control Board Proceedings. 27–28 June. Richmond, Virginia, 1988.

Cortez L, Quevauviller Ph, Martin F, Donard OFX. Survey of butyltin contamination in Portugese coastal environments. Environ Pollut 1993;82:57–62.

Damodaran N, Toll J, Pendleton M et al. Cost-benefit analysis of TBT self-polishing copolymer paints and tin-free alternatives for use on deep-sea vessels. Published by the Organotin Environmental Program (ORTEP) Association. Bergkamen, Germany, 1998:43.

Damodaran N, Toll J, Pendleton M et al. Cost analysis of TBT self-polishing copolymer paints and tin-free alternatives for use on deep-sea vessels. In: Champ, Fox, Mearns, editors. Treatment of regulated discharges from shipyards and drydocks. Published in the Proceedings Oceans '99 Conference. Washington, D.C.: Marine Technology Society. ISBN No. 0-933957-24-6. Vol. 4. pp. 153–168.

Davies IM, McKie JC, Paul JD. Accumulation of tin and tributyltin from antifouling paint by cultivated scallops (*Pectin maximus*) and Pacific oysters (*Crassostrea gigas*). Aquaculture 1986;55(2):103–114.

Davies IM, Drinkwater J, McKie JC, Balls P. Effects of the use of tributyltin antifoulants in mariculture. Proceedings Oceans '87 International Organotin Symposium. Marine Technology Society. Washington DC, 1987;4:1477–1481.

Davies IM, McKie JC. Accumulation of total tin and tributyltin in muscle tissue of farmed Atlantic salmon. Mar Pollut Bull 1987;18(7):405–407.

Davies IM, Drinkwater I, McKie JC. Effects of tributyltin compounds from antifoulants on pacific oysters *Crassostrea* *gigas* in Scottish Sea Lochs, UK. Aquaculture 1988;74(34):319–330.

Davies IM, Bailey SK. The impact of tributyltin from large vessels on dogwhelk (*Nucella lapillus*) populations around Scottish oil ports. Mar Environ Res 1991;32:201–211.

Day KE, Maguire RJ, Milani D, Batchelor SP. Toxicity of tributyltin to four species of freshwater benthic invertebrates using spiked sediment bioassays. Water Qual Res J Can 1998;33:111–132.

de Mora SJ, King N, Miller M. Tributyltin and total tin in marine sediments: profiles and the apparent rate of TBT degradation. Environ Technol Lett 1989;10:901–908.

de Mora SJ, Stewart C, Phillips D. Sources and rate of degradation of tri(*n*-butyl)tin in marine sediments near Auckland, New Zealand. Mar Pollut Bull 1995;30(1):50–57.

de Mora SJ, editor. Tributyltin: case study of an environmental contaminant. Cambridge, UK: University Press, 1996a: 301.

de Mora SJ. The tributyltin debate: ocean transportation versus seafood harvesting. In: de Mora SJ, editor. Tributyllin: case study of an environmental contaminant. Cambridge UK: University Press, 1996b:1–20.

de Mora SJ, Pelletier E. Environmental tributyltin research: past, present, future. Environ Technol 1997;18:1169–1177.

de Mora SJ, Phillips D. Tributyltin (TBT) pollution in riverine sediments following a spill from a timber treatment facility in Henderson, New Zealand. Environ Technol 1997; 18:1187–1193.

Dethlefsen V, Cameron P, von Westernhagen H. Untersuchungen uber die haufigkeit von mibbildungen in fischembryonen der sudlichen Nordsss. Inf Fischwirtsch 1985;32:22–27.

Dirkx W, Lobinsky R, Ceulemans M, Adams F. Determination of methyl- and butyltin compounds in waters of the Antwerp Harbor. Sci Total Environ 1993;136:279–300.

Douglas EW, Evans SM, Frid CLJ, Hawkins ST, Mercer TS, Scott CL. Assessments of imposex in the dogwhelk (*Nucella lapillus*) and tributyltin along the north-east of England. Invertebrate Reprod Dev 1993;23:243–248.

Dowson PH, Bubb JM, Lester JN. Organotin distribution in sediments and waters of selected east coast estuaries in the UK. Mar Pollut Bull 1992;24:492–498.

Dowson PH, Bubb JM, Lester JN. Temporal distribution of organotins in the aquatic environment: five years after the 1987 UK retail ban on TBT based antifouling paints. Mar Pollut Bull 1993a;26(9):487–494.

Dowson PH, Bubb JM, Lester JN. Depositional profiles and relationships between organotin compounds in freshwater and estuarine sediment cores. Environ Monit Assess 1993b;28:145–160.

Dowson PH, Bubb JM, Lester JN. Persistence and degradation pathways of tributyltin in freshwater and estuarine sediments. Estuarine Coastal Shelf Sci 1996;42:551–562.

Dyrynda EA. Incidence of abnormal shell thickening in the Pacific oyster (*Crassostrea gigas*) in Poole Harbour UK subsequent to the 1987 TBT restrictions. Mar Pollut Bull 1992;24:156–163.

Eastin KE. Tributyltin paint — the Navy perspective. Sea Technol 1987;28(3):69.

ESI (EVS Solutions Inc.) Waterway sediment operable unit. Harbor island superfund site: sampling and analysis plan for TBT study. Prepared for the port of Seattle. Lockheed Martin and Todd Shipyards for submittal to EPA Region 10. 5 Appendices. Seattle WA. EVS Solutions Inc., Seattle, WA, 1998:30.

ESI (EVS Solutions Inc.) Waterway sediment operable unit. Harbor island superfund site: review of tissue residue effects data for tributyltin. Mercury, and polychlorinated biphenyls. Prepared for the port of Seattle. Lockheed Martin and Todd Shipyards for submittal to EPA Region 10. Seattle WA. EVS Solutions Inc., Seattle, WA, 1999a:37.

ESI (EVS Solutions Inc.) Waterway sediment operable unit, Harbor Island superfund site: tributyltin in marine sediments and the bioaccumulation of tributyltin: combined data report. Prepared for Port of Seattle, Lockheed Martin Corporation, and Todd Shipyards Corporation. 7 Appendices. EVS Solutions Inc., Seattle, WA, 1999b:42.

ESI (EVS Solutions Inc.) Waterway sediment operable unit, Harbor Island superfund site: technical memorandum: Topics related to the TBT field study at the Harbor Island superfund site. Waterway sediment operable unit. Prepared for the Port of Seattle. Lockheed Martin Corporation and Todd Shipyards for submittal to USEPA Region 10 Seattle WA. EVS Solutions Inc., Seattle, WA, 1999c:15.

Evans DA, Huggett RJ. Statistical modeling of intensive TBT monitoring data in two tidal creeks of the Chesapeake Bay. Mar Environ Res 1991;32(1–4):169–186.

Evans SM, Hutton A, Kendall MA, Samosir AM. Recovery of dogwhelks, *Nucella lapillus* (L.) suffering from imposex. Mar Pollut Bull 1991;22:331–333.

Evans SM, Hawkins ST, Porter J, Samosir AM. Recovery of dogwhelk populations on the Isle of Cumbrae, Scotland following legislation limiting the use of TBT as an antifoulant. Mar Pollut Bull 1994;28:15–17.

Evans SM, Leksono T, McKinnel PD. Tributyltin pollution: a diminishing problem following legislation limiting the use of TBT-based antifouling paints. Mar Pollut Bull 1995;30(1):14–21.

Evans SM, Evans PM, Leksono T. Widespread recovery of dogwhelks, *Nucella lapillus* L. from tributyltin contamination in the North Sea and Clyde Sea. Mar Pollut Bull 1996;32(3):263–269.

Evans SM. Assessments of tributyltin contamination from 1986 until 1997. The misues of imposex as a biological indicator of TBT pollution. In: Stewen, editor. Harmful effects of the use of antifouling paints for ships. Organotin Environmental Programme (ORTEP) Association. Germany: Witco GmbH, 1997:51–56.

Evans SM. Tributyltin pollution: the catastrophe that never happened. Mar Pollut Bull 1999a;38(8):629–636.

Evans SM. TBT or not TBT?: that is the question. Biofouling 1999b;14(2):117–129.

Evans SM. The environment: our joint responsibility. In: Champ, Fox, Mearns, editors. Treatment of regulated discharges from shipyards and drydocks. Published in the Proceedings of Oceans '99 Conference. Washington, DC: Marine Technology Society. ISBN No. 0-933957-24-6, vol. 4, 1999c:217–222.

Evans SM, Smith R. The effects of regulating the use of TBT-based antifouling paints on TBT contamination. In: Champ, Fox, Mearns, editors. Treatment of regulated discharges from shipyards and drydocks. Published in the Proceedings of Oceans '99 Conference. Washington, DC: Marine Technology Society. ISBN No. 0-933957-24-6, vol. 4, 1999:213–216.

Evans SM, Nicholson GJ. Assessing the impact of antifouling compounds in the marine environment. Lessons to be learned from the use and misuse of biological indicators of TBT contamination. In: Champ, Fox, Mearns, editors. Treatment of regulated discharges from shipyards and drydocks. Published in the Proceedings of Oceans '99 Conference. Washington, DC: Marine Technology Society. ISBN No. 0-933957-24-6, vol. 4, 1999:193–196.

Fent K, Stegeman JJ. Effects of tributyltin in vivo on hepatic cytochrome P450 forms in marine fish. Aquat Toxicol 1993;24:219–240.

Foale S. An evaluation of the potential of gastropod imposex as an indicator of tributyltin pollution in Port Phillip Bay, Victoria. Mar Pollut Bull 1993;26:546–552.

Folsvik N, Berge JA, Brevik EM, Walday M. Quantification of organotin compounds and determination of imposex in populations of dogwhelks (*Nucella lapillus*) from Norway. Chemosphere 1998;38:681–691.

Fox TJ, Beacham T, Schafran GC, Champ MA. Advanced technologies for removing TBT from ship washdown and drydock runoff wastewaters. In: Champ, Fox, Mearns, editors. Treatment of regulated discharges from shipyards and drydocks. Published in the Proceedings of Oceans '99 Conference. Washington, DC: Marine Technology Society. ISBN No. 0-933957-24-6, vol. 4, 1999:63–72.

Garcia-Romero B, Wade TL, Salata GC, Brooks JM. Butyltin concentrations in oysters from the Gulf of Mexico from 1989 to 1991. Environ Pollut 1993;81:103–111.

Gibbons TJ. Testimony of international paint company for the house committee on merchant marine and fisheries hearing. In: Hearing Record. September 30 1986. Serial No. 99-49 (65-830-0). US Government Printing Office. Washington DC 20402, 1986:35–60.

Gibbs PE, Pascoe PL, Burt GR. Sex change in the female dog-whelk, *Nucella lapillus*, induced by tributyltin from antifouling paints. J Mar Biol Assoc UK 1988;68:715–731.

Gibbs PE, Bryan GW, Pascoe PL. TBT induced imposex in the dogwhelk, *Nucella lapillus*: geographical uniformity of the response and effects. Mar Environ Res 1991;32(1-4):79–88.

Gibbs PE, Bryan GW. Reproductive failure in the gastropod *Nucella lapillus* associated with imposex caused by tributyltin pollution: a review. In: Champ and Seligman, editors. Organotin environmental fate and effects. London: Chapman and Hall, 1996a:259–280.

Gibbs PE, Bryan GW. TBT-induced imposex in neogastropod snails: masculinization to mass extinction. In: de Mora editor. Tributyltin: case study of an environmental contaminant. Cambridge: Cambridge University Press, 1996b: 212–236.

Gibbs PE, Bebiano MJ, Coelho MR. Evidence of the differential sensitivity of neogastropods to tributyltin (TBT) pollution, with notes on a species *Columbella rustica* lacking the imposex response. Environ Technol 1997;18:1219–1224.

Godfrey TW Jr. Personal Communication. President South Tidewater Association of Ship Repairs. Hampton Roads, Virginia, 1999.

Goldberg ED. TBT: an experimental dilemma. Environment 1986;22(17-20):42–44.

Goldberg ED. Marine metal pollutants: a small set. Mar Pollut Bull 1992;25:45–47.

Gollasch S. Personal Communication. The Institute for Marine Studies in Kiel Germany. E-Mail: sgollasch@aol.com, 1999.

Gomez-Ariza JL, Beltrdn R, Morales E, Giraldez I, Ruiz-Benitez M. Acid extraction treatment of sediment samples for organotin speciation; occurrence of butyltin and phenyltin compounds on the Cadiz coast, southwest Spain. Appl Organomet Chem 1995;9:51–64.

Green GA, Cardwell R, Brancato MS. Comment on elevated accumulation of tributyltin and its breakdown products in bottlenose dolphins (*Tursiops truncatus*) found stranded along the US Atlantic and Gulf coasts. Environ Sci Technol 1997;31(10):3032–3034.

Grovhoug JG, Fransham RL, Valkirs AO, Davidson BM. Tributyltin concentrations in water, sediment, and bivalve tissues from San Diego Bay and Hawaiian harbors. In: Champ and Seligman editors. Organotin: environmental fate and effects. London: Chapman & Hall, 1996:503–534.

Guolan H, Young W. Effects of tributyltin chloride on marine bivalve mussels. Water Res 1995;29(8):1877–1884.

Haas JC, Johnson LT. Encouraging superior alternative antifouling strategies for recreational boats. Draft technical report. San Diego, CA: University of California Sea Grant Program, 2000:85.

Hall LW. Monitoring organotin concentrations in Maryland waters of Chesapeake Bay. Interagency workshop on aquatic monitoring and analysis for organotin. Rockville MD: NOAA/NMPPO, 1986:27–28.

Hall LW, Lenkevich MJ, Hall WS, Pinkney AE, Bushong SJ. Monitoring organotin concentrations in Maryland waters of Chesapeake Bay. Proceedings Oceans '86 International Organotin Symposium. Marine Technology Society. Washington DC 1986;4:1275–1279.

Hall LW, Lenkevich MJ, Hall WS, Pinkney AE, Bushong SJ. Evaluation of butyltin compounds in Maryland waters of Chesapeake Bay. Mar Pollut Bull 1987;18:7883.

Hall LW. Tributyltin environmental studies in Chesapeake Bay. Mar Pollut Bull 1988;19(9):431–438.

Hardy JT, Cleary J. Surface microlayer contamination and toxicity in the German Bight. Mar Ecol Prog Ser 1992;91:203–210.

Harris JRW, Hamlin CC, Stebbing ARD. A simulation study of the effectiveness of legislation and improved dockyard practice in reducing TBT concentrations in the Tamar Estuary. Mar Environ Res 1991;32:279–292.

Hashimoto S, Watanabe M, Noda Y et al. Concentration and distribution of butyltin compounds in a heavy tanker route in the Strait of Malacca and in Tokyo Bay. Mar Environ Res 1998;45:169–177.

His E, Robert R. Développement des véligères de *Crassostrea gigas* dans le bassin d'Arcachon. Etudies sur les mortalitiés larvaires. Rev Trav Inst Pêches Marit 1983–1985;47:6388.

His E. Embryogenesis and larval development in *Crassostrea gigas:* experimental data and field observations on the effect of tributyltin compounds. Chapter 12. In: Champ and Seligman, editors. Organotin: environmental fate and effects. London: Chapman & Hall, 1996:239–258.

Horiguchi T, Shiraishi H, Shimisu M, Yamazaki S, Morita M. Imposex in Japanese gastropods (neogastropoda and mesogastropoda): effects of tributyltin and triphenyl from antifouling paints. Mar Pollut Bull 1995;31:402–405.

Huggetts RJ. Monitoring tributyltin in southern Chesapeake Bay. In: The Proceedings of the Interagency Workshop on Aquatic Monitoring and Analysis for Organotin. Sponsored by NOAA/NMPPO. Rockville MD, 1986:29–30.

Huggetts RJ. Statement for Senate Hearing. The effects of the chemical tributyltin TBT on the marine environment. Hearing Record. Senate Subcommittee on Environmental Protection of the Committee on Environment and Public Works. April 29. S. HGR 100-89. US Gov. Print. Office 73-832. Washington, DC, 1987;23–28 (oral) 68–74 (written).

Huggett RJ, Unger MA, Westbrook DJ. Organotin concentrations in southern Chesapeake Bay. Proceedings Oceans '86 International Organotin Symposium. Marine Technology Society. Washington DC, 1986;4:1262–1265.

Huggett RJ, Unger MA, Seligman RE, Valkirs AO. The marine biocide tributyltin. Environ Sci Technol 1992;26(2):232–237.

Huggett RJ, Evans DA, MacIntyre WG, Unger MA, Seligman PF, Hall PF. Tributyltin concentration in waters of the Chesapeake Bay. In: Champ and Seligman, editors. Organotin: environmental fate and effects. London: Chapman & Hall, 1996:485–502.

Huet M, Paulet YM, Le Pennec M. Survival of *Nucella lapillus* in a tributyltin-polluted area in west Brittany: a further example of a male genital defect (Dumpton syndrome) favouring survival. Mar Biol 1996;125:543–549.

Hwang HM, Oh JR, Kahng SH, Lee KW. Tributyltin compounds in mussels, oysters, and sediments of Chinhae Bay Korea. Mar Environ Res 1999;47:61–70.

IMO International Martime Organization. MEPC Marine Environmental Protection Committee of the International Maritime Organization IMO. Background papers and meeting notes. MEPC 29th and 30th Sessions. IMO. London, SE1 7SR, 1990.

Iwata H, Tanabe S, Mizuno T, Tatsukawa R. High accumulation of toxic butyltins in marine mammals from Japanese coastal waters. Environ Sci Technol 1995;29:2959–2962.

Johnson D. Discharge of tributyltin into state of Virginia waters, 1999.

Kalbfus W, Zellner A, Frey S, Th Knorr. Analysis of butyltin species in water, sediment and environmental matrices. Report No UBA-FB. November, 1996:50.

Kan-Atireklap S, Tanabe S, Sanguansin J. Contamination by butyltin compounds in sediments from Thailand. Mar Pollut Bull 1997;34:894–899.

Kannan K, Tanabe S, Iswata H, Tatsukawa R. Butyltins in muscle and liver of fish collected from certain Asian and Oceanian countries. Environ Pollut 1995;90:279–290.

Kannan KS, Corsolini S, Focardi S, Tanabe S, Tatsukawa R. Accumulation pattern of butyltin compounds in dolphin, tuna and shark collected from Italian coastal waters. Arch Environ Contam Toxicol 1996a;31:19–23.

Kannan K, Senthilkumar K, Sinha RK. Sources and accumulation of butyltin compounds in Ganges River dolphin, *Platanista gangetica*. Appl Organomet Chem, 1996b.

Kannan K, Falandysz J. Butyltin residues in sediment, fish, fish eating birds, harbor porpoise and human tissues from the Polish coast of the Baltic Sea. Mar Pollut Bull 1997;34:203–207.

Kannan K, Falandysz J. Butyltin residues in sediment, fish, fish-eating birds, harbor porpoise and human tissues from the polish coast of the Baltic Sea. Mar Pollut Bull 1998;34(3):203–207.

Kannan K, Guruge KS, Thomas NJ, Tanabe S, Giesy JP. Butyltin residues in southern sea otters (*Enhydra lutris nereis*) found dead along California coastal waters. Environ Sci Technol 1998;32:1169–1175.

Kannan K, Falandysz J. Response to the comment on: butyltin residues in sediment, fish, fish-eating birds, harbor porpoise and human tissues from the Polish coast of the Baltic Sea. Mar Pollut Bull 1999;38:61–63.

Kram ML, Stang PM, Seligman PF. Adsorption and desorption of tributyltin in sediments of San Diego Bay and Pearl Harbor. Appl Organomet Chem 1989;3:523–536.

Keeley KL. Personal Communication. US EPA Region 10 1200 Sixth Avenue. Seattle Washington. 98101. E-Mail: *keeley.karen@epa.gov*, 1999.

Key D, Nunny RS, Davidson PE, Leonard MA. Abnormal shell growth in the Pacific oyster *Crassostrea gigas*. Some preliminary results from experiments undertaken in 1975. International Council for the Exploration of the Sea ICES. Copenhagen, Denmark, CMK: 1976:117.

King N, Miller M, de Mora SJ. Tributyl tin levels for sea water, sediment, and selected marine species in coastal Northland and Auckland, New Zealand. N Z J Mar Freshwater Res 1989;23:287–294.

Ko MM, Bradley CC, Neller AH, Broom MJ. Tributyltin contamination of marine sediments of Hong Kong. Mar Pollut Bull 1995;31:249–253.

Langston WJ, Burt GR. Bioavailability and effects of sediment-bound TBT in deposit-feeding clams. *Scrobicularia-plana*. Mar Environ Res 1991;32:61–77.

Laughlin R. Bioaccumulation of TBT by aquatic organisms. In: Champ and Seligman, editors. Organotin: enviromental fate and effects. London: Chapman and Hall, 1996:331–356.

Law RJ, Blake SJ, Jones BR, Rogan E. Organotin compounds in liver tissue of harbor porpoises (*Phocoena phocoena*) and grey seals (*Halichoerus glypus*) from the coastal waters of England and Wales. Mar Pollut Bull 1998;36:241–247.

Law RJ, Blake SJ, Spurrier CJH, in press. Butyltin compounds in liver tissue of pelagic marine mammals stranded on the coasts of England and Wales. Mar Pollut Bull.

Law RJ, Evers EHG, submitted. The environmental distribution and effects of tributyltin — an update to mid-1999. J Mar Res 29.

Lee RF, Valkirs AO, Seligman PF. Importance of microalgae in the biodegradation of tributyltin in estuarine waters. Environ Sci Technol 1989;23:1515–1518.

Lee RF. Metabolism of tributyltin by marine animals and possible linkages to effects. Mar Environ Res 1991;32(1-4):29–36.

Lewis AG, Cave WR. The biological importance of copper in oceans and estuaries. Oceanogr Mar Biol Annu Rev 1982;20:471–695.

Ludgate JW. Testimony of International Paint USA Inc for the US House of Representatives Committee on Merchant Marine and Fisheries Hearing. In: Hearing Record. July 8, 1987. Serial No. 100-28 (78-297). US Government Printing Office. Washington DC 20402, 1987:73–86.

Macauley JM, Summers JK, Heitmuller PT et al. Annual statistical summary: EMAP-estuaries Louisianian Providence 1993. US EPA Environmental Research Laboratories. Gulf Breeze FL. EPA/620/R-94/002, 1994:82.

Maguire RJ, Hale EJ. Butyltins in the Great Lakes Basin. NWRI Report. Dec 1981:28.

Maguire RJ, Chau YK, Bengert GA, Hale EJ, Wong PTS, Kramar O. Occurrence of organotin compounds in Ontario lakes and rivers. Environ Sci Technol 1982;16:698–702.

Maguire RJ. Butyltin compounds and inorganic tin in sediments in Ontario. Environ Sci Technol 1984;18:291–294.

Maguire RJ, Tkacz RJ, Sartor DL. Butyltin species and inorganic tin in water and sediment of the Detroit and St. Clair Rivers. J Great Lakes Res 1985;11:320–327.

Maguire RJ, Tkacz RJ, Chau YK, Bengert GA, Wong PTS. Occurrence of organotin compounds in water and sediment in Canada. Chemosphere 1986;15:253–274.

Maguire RJ. Review of environmental aspects of tributyltin. Appl Organomatic Chem 1987;1:475–498.

Maguire RJ, Tkacz RJ. Concentration of tributyltin in the surface microlayer of natural waters. Water Pollut Res J Can 1987;22:227–233.

Maguire RJ. Aquatic environmental aspects of non-pesticidal organotin compounds. Water Pollut Res J Can 1991;26:243–360.

Maguire, R.J. Tributyltin in Canadian waters. In: Champ and Seligman, editors. Organotin: environmental fate and effects. London: Chapman & Hall, 1996a:535–552.

Maguire RJ. The occurrence, fate and toxicity of tributyltin and its degradation products in fresh water environments. In: de Mora, editor. Tributyltin: case study of an environmental contaminant. London: Cambridge University Press, 1996b:94–138.

Maguire RJ. Tributyltin — history and prognosis. Soc Environ Toxicol Chem News 1998;18(5):21–22. September.

Martin RC, Dixon DG, Maguire RJ, Hodson PV, Tkacz RJ. Acute toxicity, uptake, depuration and tissue distribution of tri-*n*-butyltin in rainbow trout, *Salmo gairdneri*. Aquat Toxicol 1989;15:37–52.

Matthiessen P, Gibbs PE. Critical appraisal of the evidence for tributyltin-mediated endocrine disruption in molluscs. Environ Toxicol Chem 1998;17:37–43.

MacLellan D, Brancato MS, DeForest D, Volosin J. An evaluation of risks to US Pacific Coast otters exposed to tributyltin. In: Champ, Fox, Mearns, editors. Treatment of regulated discharges from shipyards and drydocks. Published in the Proceedings of Oceans '99 Conference. Washington, DC: Marine Technology Society. ISBN No. 0-933957-24-6, vol. 4, 1999:177–184.

McCarty LS. Toxicant body residues: implications for aquatic bioassays with some organic chemicals. In: Mayes and Barron, Editors. In: Aquatic toxicology and risk assessment. American Society for Testing and Materials. Philadelphia, 1991:183–192.

McCarty LS, Mackay D. Enhancing ecotoxicological modeling and assessment. Environ Sci Technol 1993;27(9):1719–1728.

Meador JP. The effect of laboratory holding on toxicity response of marine infaunal amphipods to cadmium and tributyltin. J Exp Mar Biol Ecol 1993;174:227–242.

Meador JP, Varanasi U, Krone CA. Differential sensitivity of marine infaunal amphipods to tributyltin. Mar Biol 1993;116:231–239.

Meador JP. Comparative toxicokinetics of tributyltin in five marine species and its utility in predicting bioaccumulation and acute toxicity. Aquat Toxicol 1997;37:307–326.

Meador JP, Rice CA, in press. Impaired growth of the polychaete *Armandia brevis* exposed to tributyltin in sediment. Mar Environ Res.

Meador JP, In press. Predicting the fate and effects of tributyltin in marine systems. Rev Environ Contam Toxicol, 2000.

Mensink BP, Boon JP, Ten Hallers-Tiabbes CC, van Hattum B, Koeman JH. Bioaccumulation of organotin compounds and imposex occurrence in a marine food chain (Eastern Scheldt, the Netherlands). Environ Technol 1997a;18:1235–1244.

Mensink BP, van Hattum B, Ten Hallers-Tjabbes CC et al. Tributyltin causes imposex in the common whelk. *Buccinum undatum*: mechanism and occurrence. NIOZ-Rapport 1997-6. ISSN 0923-3210, 1997:56.

Messing AW, Ramirez LM, Fox TJ. A review to determine state-of-the-practice treatment technologies for reducing concentrations of organotin compounds in wastewater. AMRL Technical Report No 3060. Old Dominion University, Norfolk, VA 1997:57.

Michel P, Averty B. Tributyltin contamination in the Rade de Brest. Roadstead Programme. Third International Scientific Meeting: Proceedings Brest. 14–16 March 1995, 1995;2:8796.

Michel P, Averty B. Contamination of French coastal waters by organotin compounds: 1997. Mar Pollut Bull 1999;38:268–275.

Mikami M. Personal communication. Kansai Paint America Inc. Two Executive Drive Suite 785. Fort Lee NJ 02024. E-Mail: mmikami@kpamerica. com, 1999.

Milne A. Roughness and drag from the marine chemist's viewpoint. Proceedings of the International Workshop on Marine Roughness and Drag. London: Published by the Royal Institution of Naval Architects, 1990a.

Milne A. Cost/benefit analysis of SPC organotin antifoulings. International Maritime Organization IMO. Marine Environmental Protection Committee MEPC. Uses of tributyltin compounds in anti-fouling paints for ships. Document MEPC 30/INF.16. IMO London, 1990b:7–9.

Milne A. Self-polishing coatings in marine antifouling paints. Presented at The Chemical Society, The Royal Institute of Chemistry, Annual Chemical Congress. University of Newcastle Upon Typne, School of Marine Technology, ca. 1993:25.

Milne A. The costs and benefits of tributyltin and alternative antifoulants. In: The present status of TBT–copolymer antifouling paint. Organotin Environmental Programme Association ORTEP. The Hague, Netherlands, 1996:17–27.

Milne A. (personal communication). Director 3AM Ltd. 39 Sanderson Road, Newcastle Upon Tyne, NE2 2DR. UK (former Courtlands Senior Antifouling Paint Chemist).

Minchin D, Duggan CB, King W. Possible effects of organotins on scallop recruitment. Mar Pollut Bull 1987;18(11):604–608.

Minchin D. Recovery of a population of the flame shell, *Lima hians*, in an Irish bay previously contaminated with TBT. Environ Pollut 1995;90:259–262.

Minchin D, Oehlmann J, Duggan CB, Stroben E, Keatinge M. Marine TBT antifouling contamination in Ireland, following legislation in 1987. Mar Pollut Bull 1995;30:633–639.

Minchin D, Stroben E, Oehlmann J, Bauer C, Duggan CB, Keatinge M. Biological indicators used to map organotin contamination in Cork Harbor, Ireland. Mar Pollut Bull 1996;32:188–195.

Minchin A, Minchin D. Dispersal of TBT from a fishing port determined using the dogwhelk *Nucella lapillus* as an indicator. Environ Technol 1997;18:1225–1234.

Minchin D, Bauer B, Oehlmann J, Schulte-Oehlmann U, Duggan CB. Biological indicators used to map organotin contamination from a fishing port, Killybeg, Ireland. Mar Pollut Bull 1997;34(4):235–243.

Minchin D. Personal communication. Fisheries Research Centre, Department of the Marine, Abbotstown, Dublin 15, Ireland. dminchin@frc.ie, 1999.

Minchin D, Sheehan J. The significance of ballast water in the introduction of exotic marine organisms to Cork Harbor Ireland. ICES Coop Res Rep 224, 1999:12–24.

Moore DW, Dillon TM, Suedel BC. Chronic toxicity of tributyltin to the marine polychaete worm, *Neanthes arenaceodentata*. Aquat Toxicol 1991;21:181–198.

Moore JJ, Little AE, Harding MJC, Rodger GK, Davies IM. Surveys of dogwhelks *Nucella lapillus* in the vicinity of Sullom Voe, Shetland, August 1995. Oil Pollution Research Unit Neyland Pembrokeshire. Report No. OPRU/3/96, 1996.

Morgan E, Murphy J, Lyons R. Imposex in *Nucella lapillus* from TBT contamination in south and south-west Wales: a continuing problem around ports. Mar Pollut Bull 1998;36:840–843.

Murray LA, Waldock R, Reed J, Jones BR, in press. Sediment quality in dredged material disposed to sea from England and Wales. CATS 4: Conference on the Characterisation and Treatment of Sediments. Antwerp Belgium, 15–17 September, 1999.

NAVSEA US Naval Sea Systems Command. Environmental assessment of fleetwide use of organotin antifouling paint. NAVSEA. Washington, DC, 1984:128.

NAVSEA US Naval Sea Systems Command. Organotin antifouling paint: US Navy's needs, benefits, and ecological research. A report to congress. + Appendix. NAVSEA, Washington DC, 1986:38.

Nicholson CJ, Evans SM, Palmer N, Smith R. The value of imposex in the dogwhelk *Nucella lapillus* and the common whelk *Buccinum undatum* as indicators of TBT contamination. Invertebrate Reprod Dev 1998;34:289–300.

O'Connor TP. Mussel watch results from 1986 to 1996. Mar Pollut Bull 1998;37(1-2):14–19.

O'Connor TP, Pearce J. (Guest editors) U.S. coastal monitoring: NOAA's National status and trends results. Mar Poll Bull 1998;37(1-2):1–113.

Oehlmann J, Liebe S, Watermann B, Stroben E, Fioroni P, Deutsch U. New perspectives of sensitivity of littorinids to TBT pollution. Cah Biol Mar 1994;35:254–255.

Oehlmann J, Fioroni P, Stroben E, Markert B. Tributyltin TBT effects on *Ocinebrina aciculata* (Gastropoda: Muricidae): imposex development, sterilization, sex change and population decline. Sci Total Environ 1996;188:205–223.

Oehlmann J, Bauer B, Minchin D, Schulte-Oehlmann U, Fiorini P, Markert B. Imposex in *Nucella lapillus* and intersex in *Littorina littorea*: interspecific comparison of two TBT-induced effects and their geographical uniformity. Hydrobiologia 1998;378:199–213.

Oh JR. Studies on TBT contamination in marine environment of Korea. TR. Korea Ocean Research & Development Institute, 1998:211.

ORTEP (Organotin Environmental Programme Association). The present status of TBT copolymer antifouling paints. Proceedings of an International One Day Symposium on Antifouling Paints for Ocean Going Vessels. 21 February 1996. The Hague, The Netherlands, 1996.

ORTEP (Organotin Environmental Programme Association). Harmful Effects of the Use of Antifouling Paints for Ship. In: Stewen editor. Organotin Environmental Programme Association. ORTEP Association. Germany: Witco GmbH, 1997:187.

ORTEP (Organotin Environmental Programme Association). Further updates on the toxicology of tributyltin, including assessments of risks to humans, wildlife and aquatic life. In: Stewen, editor. Organotin Environmental Programme Association ORTEP Association. Germany: Witco Gmbh, 1998:181.

Poloczanska ES, Ansell AD. Imposex in the whelks *Buccinum undatum* and *Nepunea antiqua* from the westcoast of Scotland. Mar Environ Res 1999;47:203–212.

Proceedings of the International Organotin Symposium of the Oceans '86 Conference.Washington DC Sept 23–25. Vol. 4, pp. 1101–1330. Marine Technology Society. Washington DC and the IEEE Service Center. 445 Hoes Lane, Piscataway, NJ 08854, USA, 1986.

Proceedings of the International Organotin Symposium of the Oceans '87 Conference. Halifax NS Sept. 28–Oct 1. Vol 4, pp. 1296–1454. Marine Technology Society. Washington DC and the IEEE Service Center. 445 Hoes Lane, Piscataway, NJ 08854, USA, 1987.

Proceedings of the National Organotin Symposium of the Oceans '88 Conference. Baltimore MD. Oct 31–Nov 2. Vol 4. Marine Technology Society. Washington DC and the IEEE Service Center. 445 Hoes Lane, Piscataway, NJ 08854, USA, 1988.

Proceedings of the National Organotin Symposium of the Oceans '89 Conference. Seattle WA. Sept 18–21. Vol 3. Marine Technology Society. Washington DC and the IEEE Service Center. 445 Hoes Lane, Piscataway, NJ 08854, USA, 1989.

Proceedings of the Third International Organotin Symposium. In: Mee and Fowler, editors. Monaco. April 17–20. J Mar Environ Res. 1991. Special Issue on Organotin. 1990;32:292.

Prouse NJ, Ellis DV. A baseline survey of dogwhelk *Nucella lapillus* imposex in eastern Canada 1995 and interpretation in terms of tributyltin (TBT) contamination. Environ Technol 1997;18:1255–1264.

Reise K, Gollasch S, Wolff WJ. Introduced marine species of the North Sea coasts. Helgolander Meeresunters 1999;52:219–234.

Rees HL, Waldock R, Matthiessen P, Pendle MA. Surveys of the epibenthos of the Crouch Estuary (UK) in relation to TBT contamination. J Mar Biol Assoc UK 1999;79:209–223.

Reid B. A chemical on trial. Daily Press/The Times-Herald. New Port News. VA. Reprinted Section-Collection of Articles. 1986.

Rice SD, Short JW, Stickle WB. Uptake and catabolism of tributyltin by blue crabs fed TBT contaminated prey. Mar Environ Res 1989;27:137–145.

Ricketts RDM MV. The effects of the chemical tributyltin TBT on the marine environment. Hearing Record. Senate Subcommittee on Environmental Protection of the Committee on Environment and Public Works. April 29. S HGR 100-89. US Government Printing Office. 73-832. Washington DC, 1987:28–30 (oral) 76–84 (written).

Rilov G, Benayahu Y, Evans SM, Gasith A. Unregulated use of TBT-based antifouling paints and TBT pollution in Israel. Mar Ecol Prog Ser. In press, 1999.

Ritsema R, Laane RWPM, Donard OFX. Netherlands in 1988 and 1989: concentrations and effects. Mar Environ Res 1991;32(1-4):243–260.

Ritsema R. Dissolved butyltins in marine waters of the Netherlands three years after the ban. Appl Organomet Chem 1994;8:5–10.

Ritsema R, de Samle T, Loens L, de Jong AS, Donard OFX. Determination of butyltins in harbour sediment and water by aqueous phase ethylation GC-ICF-MS and hydride generation GC-AAS. Environ Pollut, 1998;99:271–277.

Rouhi AM. The squeeze on tributyltins. C&E News. Am Chem Soc. April 27. Washington DC, 1998:41–42.

Russell D, Brancato MS, Bennett HJ. Comparison of trends in tributyltin concentrations among three monitoring programs in the United States. J Mar Sci Technol 1996;1:230–238.

Saint-Louis R, Gobeil C, Pelletier E. Tributyltin and its degradation products in the St. Lawrence Estuary (Canada). Environ Technol 1997;18:1209–1218.

Saint-Jean SD, Courtenay SC, Pelletier E, Saint-Louis R. Butyltin concentrations in sediments and blue mussels *Mytilus edulis* of the southern Gulf of St. Lawrence, Canada. Environ Technol 1999;20:181–189.

St-Louis R, de Mora SJ, Pelletier E et al., in press. Recent butyltin contamination in Beluga whales (Delphinapterus leucas) from the St. Lawrence Estuary and Northern Quebec Canada. Appl Organomet Chem.

Salazar MH. Environmental significance and interpretation of organotin bioassays. In: Proceedings Oceans 1986. International Organotin Symposium, 23–25 September 1986. Washington DC. 1986;4:1240–1244.

Salazar MH, Salazar SM. TBT effects on juvenile bivalve growth. SETAC Eighth Annual Meeting. 9–12 November 1987. Pensacola Florida. Abstract only No. 330, 1987.

Salazar SM, Davidson BM, Salazar MH, Stang PM, Meyers-Schulte K, Henderson RS. Field assessment of a new site-specific bioassay system. In Proceedings, Oceans 1987 Conference. 28 Sept–1 October 1987. Halifax Novia Scotia. Canada. Organotin Symposium. 1987;4:1461–1470.

Salazar MH, Champ MA. Tributyltin and water quality: a question of environmental significance. Published in the Proceedings of the National Organotin. Marine Technology Society. Washington DC and the IEEE Service Center, 445 Hoes Lane, Piscataway, NJ 08854, USA, 1988;4:1497–1506.

Salazar MH, Salazar SM. Assessing site-specific effects of TBT contamination with mussel growth rates. Mar Environ Res 1991;32(1-4):131–150.

Salazar MH, Salazar SM. Mussels as bioindicators: effects of TBT on survival, bioaccumulation, and growth under natural conditions. In: Champ and Seligman editors. Organotin: environmental fate and effects. London: Chapman & Hall, 1996:305–330.

Salazar MH, Salazar SM. Using caged bivalves as part of an exposure–dose–response triad to support and integrated risk assessment strategy. In: De Peyster, Day, editors. Ecological risk assessment: a meeting of policy and science. SETAC Press, 1988:167–192.

Salazar MH. Personal communication. Applied biomonitoring. 11648 72nd Place NE. Kirkland WA 98034. Tel. (425) 823–3905, Fax (425) 814–4998, msalazar@cnw.com, 1999

Salazar MH, Salazar SM. Characterizing exposure and effects of TBT in bivalves using tissue chemistry and sublethal endpoints. Proceedings Puget Sound Sediment Management Annual Review Meeting. Lacey, April, WA, 1999:12. http.l/www.nws.usace.army.mil/dmmo/TBT EXP.PDF.

Salazar MH, Salazar SM. Unpublished manuscript. Using the exposure–dose–response triad in laboratory and field bioassays: lessons learned from caged bivalves and TBT. Presented at the 20th SETAC Meeting Philadelphia PA. 14–18 November, 1999. Platform presentation. http://members.tripod.com/mussels/mussels.htm.

Salazar MH. Mortality growth and bioaccumulation in mussels exposed to TBT: differences between the laboratory and the field. Proceedings, Oceans 1989 International Organotin Symposium. Marine Technology Society, Washington DC, 1989;2:530–536.

Schatzberg P. Organotin antifouling paints and the US Navy: a historical perspective. Proceedings Oceans '87 International Organotin Symposium. Marine Technology Society, Washington DC, 1987;4:1324–1333.

Seligman PF, Valkirs AO, Stang PM, Lee RF. Evidence for rapid degradation of tributyltin in a marina. Mar Pollut Bull 1988;19(10):531–534.

Seligman PF, Grovhoug JG, Valkirs AO et al. Distribution and fate of tributyltin in the United States marine environment. Appl Organomet Chem 1989;3:31–47.

Seligman PF, Grovhoug JG, Fransham RL, Davidson B, Valkirs AO. US Navy statutory monitoring of tributyltin in selected US harbors. Annual Report: 1989. Naval Ocean Systems Center Technical Report No 1346. Naval Ocean Systems Center, San Diego, CA, 1990:32.

Seligman PF, Adema CM, Grovhoug J et al. Environmental loading of tributyltin from drydocks and ship hulls. In: Champ and Seligman, editors. Organotin: environmental fate and effects. London: Chapman & Hall, 1996a:405–428.

Seligman PF, Maguire RJ, Lee RF, Hinga KR, Valkirs AO, Stang PM. Persistence and fate of tributyltin in aquatic ecosystems. In: Champ and Seligman, editors. Organotin: environmental fate and effects. London: Chapman & Hall, 1996b:429–458.

Shim WJ, Oh JR, Kahng SH, Shim JH, Lee SH. Tributyltin and triphenyltin residues in Pacific oyster (*Crassostrea gigas*) and rock shell (*Thais clavigera*) from the Chinhae Bay System Korea. J Korean Soc Oceanogr 1998a;3:90–99.

Shim WJ, Oh JR, Kahng SH, Shim JH, Lee SH. Accumulation of tributyl- and triphenyltin compounds in Pacific oyster, *Crassostrea gigas*, from the Chinhae Bay System, Korea. Arch Environ Contam Toxicol 1998b;35:41–47.

Sindermann CJ. Winning the games scientists play ISBN 0-306-41075-3. New York, NY: Plenum Press, 1982:290.

Smeenk C. Strandings of sperm whales *Physeter macrocephalus* in the North Sea: history and patterns. In: Jacques TG, Lambertsen RH, editors. Sperm whale deaths in the North Sea: science and management. Bulletin de l'Institut Royal des Sciences Naturelles de Belgique Biologie, 1997;67: 15–28.

Smith PJ. Selective decline in imposex levels in the dogwhelk *Lepsiella scobina* following a ban on the use of TBT antifoulants in New Zealand. Mar Pollut Bull 1996;32(3): 62–365.

Spooner N, Gibbs PE, Bryan GW, Goad LJ. The effect of tributyltin upon steroid titres in the female dogwhelk, *Nucella lapillus*, and the development of imposex. Mar Environ Res 1991;32(1-4):37–50.

Stang P, Lee R, Seligman P. Evidence for rapid, non-biological degradation of tributyltin in fine-grained sediments. Environ Sci Technol 1992;26(7):1382–1387.

Stebbing ARD. Organotins and water quality — some lessons to be learned. Mar Pollut Bull 1985;16(10):383–390.

Stebbing ARD, Dethlefsen V. Introduction to the Bremerhaven workshop on biological effects of contaminants. Mar Ecol Prog Ser 1992;91:1–8.

Stebbing, ARD. Organotins — what help from hindsight. In: Champ and Seligman, editors. Organotin: environmental fate and effects. Foreword. London: Chapman & Hall, 1996:xiii–xxv.

Stewart C, de Mora SJ. A review of the degradation of tri(*n*-butyl)tin in the marine environment. Environ Technol 1990;11:565–570.

Stewart C, de Mora SJ. Elevated tri(*n*-butyl)tin concentrations in shellfish and sediments from Suva Harbor, Fiji Applied. Organomet Chem 1992;6:507–512.

Stewart C, de Mora SJ, Jones MRL, Miller MC. Imposex in New Zealand neogastropods. Mar Pollut Bull 1992;24: 204–209.

Stewart C. The efficacy of legislation in controlling tributyltin in the marine environment. In: De Mora, editor. Tributyltin: case study of an environmental contaminant. Cambridge: University Press, 1996:237–297.

Stewart C, Thompson JAJ. Vertical distribution of butyltin residues in sediments of British Columbia harbors. Environ Technol 1997;18:1195–1202.

Stronkhorst J, Bowmer CT, Otten H. TBT contamination and toxicity of harbor sediments in the Netherlands. Paper presented at the Conference on Costs and Benefits of TBT based and Alternative Antifoulants. 4–6 December. Malta, 1995.

Stronkhorst J. TBT contamination and toxicity of sediments. The present status of TBT copolymer antifouling paints. Proceedings of an International One Day Symposium on Antifouling Paints for Ocean Going Vessels. 21st February 1996. The Hague, The Netherlands, 1996:47–60.

Svavarsson J, Skarphédinsdóttir H. Imposex in the dogwhelk *Nucella lapillus* (L.) in Icelandic waters. Sarsia 1995; 80:35–40.

Swain GW. Redefining antifouling coatings. J Prot. Coat. Linings 1999;16(9):26–35.

Swennen C, Ruttanadakul N, Ardseungnem S, Singh HR, Mensink BP, Ten Hallers-Tjabbes CC. Imposex in sublittoral and littoral gastropods from the Gulf of Thailand and

Strait of Malacca in relation to shipping. Environ Technol 1997;18:1245–1254.

Tanabe S, Prudente M, Mizuno T, Hasegawa J, Iwata H, Miyazaki N. Butyltin contamination in marine mammals from North Pacific and Asian waters. Environ Sci Technol 1998;32:192–198.

Tanguy A, Castro NF, Marhic A, Moraga D. Effects of an organic pollutant tributyltin on genetic structure in the Pacific oyster *Crassostrea gigas*. Mar Pollut Bull 1999; 38(7):550–559.

Tas JW, Opperhuizen A. Analysis of triphenyltin in fish. Mar Environ Res 1991;32(1–4):271–278.

Ten Hallers-Tjabbes CC, Kemp JF, Boon JP. Imposex in whelks *Buccinum undatum* from the open North Sea: relation to shipping traffic intensities. Mar Pollut Bull 1994;28:311–313.

Ten Hallers-Tjabbes CC, Boon JP. Whelks (*Buccinum undatum* L.) and dogwhelks (*Nucella lapillus* L.) and TBT — a cause for confusion. Mar Pollut Bull 1995;30:675–676.

Ten Hallers-Tjabbes CC, Everaarts JM, Mensink BP, Boon JP. The decline of the North Sea whelk (*Buccinum undatum* L.) between 1970 and 1990: a natural or a human-induced event? Mar Ecol PSZN 1 1996;17:333–343.

Tester M, Ellis DV. TBT controls and the recovery of whelks from imposex. Mar Pollut Bull 1995;30:90–91.

Tester M, Ellis DV, Thompson JAJ. Neogastropod imposex for monitoring recovery from marine TBT contamination. Environ Toxicol Chem 1996;15:560–567.

Thain JE. The acute toxicity of bis (tributyl tin) oxides to the adults and larvae of some marine organisms. ICES CM 1983;E13:5.

Thain JE, Waldock MJ, Waite ME. Toxicity and degradation studies of tributyltin TBT and dibutyltin DBTA in the aquatic environment. Proceedings Oceans '87 International Organotin Symposium. Marine Technology Society, Washington DC, 1987;4:1398–1404.

Thompson JAJ, Pierce RC, Sheffer MG et al. Organotin compounds in the aquatic environment: scientific criteria for assessing their effects on environmental quality. NRCC Assoc Comm Sci Crit Environ Qual. NRCC Publ No 22494, 1985:284.

Thompson JAJ, Douglas S, Chau YK, Maguire RJ. Recent studies of residual tributyltin in coastal British Columbia sediments. Appl Organomet Chem 1998;12:643–650.

Tong SL, Pang FY, Phang SM, Lai HC. Tributyltin distribution in the coastal environment of peninsular Malaysia. Environ Pollut 1996;91:209–216.

Toll J, Brancato MS, DeForest D. Regulating biocidal antifoulants: creating a level playing field. Proceedings Oceans '99 International Symposium on the Treatment of Regulated Discharges from Shipyards and Drydocks. Marine Technology Society, Washington DC, 1999:4, In Press.

UK DOE Department of Environment. The Control of Pollution Act COPA of 1985 the control of Pollution Act of 1985 the anti-fouling Paints Regulations of 1985 Statutory Instruments No. 2011, No. 2300, 1986. Joint Circular,7 March 1986 7/86. DOE London. HMSO, 1986a:4.

UK DOE Department of Environment. The Control of Pollution ACT COPA of 1986 which amended the Control of Pollution Act of 1985 the Anti-fouling Paints Regulations of 1986. Joint Circular. 24 February 1987 3/87. DOE, London. HMSO, 1986b:4.

UK DOE Department of the Environment Central Directorate of Environmental Protection. Organotin in antifouling paints environmental considerations. Her Majesty's Stationery Office, London, England. Pollution Paper No. 25, 1986c:82.

UK DOE Department of Environment. The Control of Pollution (Antifouling Paints and Treatments Regulations) 1987. Statutory Instruments No. 783 1987. Joint Circular 12 June 1987 19/87. Revision to Annex listing paints and treatment which conform to Regulations issued 5/1/88 3/1/89, 11/1/89. DOE London, HMSO, 1987:3.

Unger MA, Greaves J, Huttett RJ. Grignard derivatization and mass spectrometry as techniques for the analysis of butyltins in environmental samples. Chapter 6. In: Champ and Seligman, editors. Organotin: environmental fate and effects. London: Chapman & Hall, 1996:123–134.

US Congress. The Senate Congressional Record dated February 2, 1987a-S1481. US Government Printing Office, Washington DC 20402.

US Congress. The Senate hearing record for the effects of the chemical tributyltin (TBT) on the marine environment. Committee on Environment and Public Works. US Senate. April 29 1987. S Hrg Serial 100-89 73-832. US Government Printing Office, Washington DC 20402, 1987b:189.

US Congress. The Organotin Antifouling Paint Control Act of 1988. (33 USC 2401). Public Law 1988:100–333.

US EPA Environmental Protection Agency. Chesapeake Bay Program. Survey of tributyltin and dibutyltin concentrations at selected harbors in Chesapeake Bay — final report. CBP/TRS 14/87 + Appendix. Annapolis, MD. 1987:40–58.

US EPA Environmental Protection Agency. Report to Congress on Environmental Monitoring of Organotin. OPTS Office of Pesticides and Toxic Substances. Office of Pesticide Programs. Washington DC, 1991:23.

US EPA Environmental Protection Agency. Recommendations for screening values for tributyltin in sediments at Superfund sites in Puget Sound Washington. Technical report. EPA 91 O-R-96-014. Region 10. Seattle, Washington, 1996a.

US EPA Environmental Protection Agency. Report to congress on alternatives to organotin antifoulants and alternative antifoulant research. Draft: December, 1996. EPA Office of Pesticide Programs. Special Review and Registration Division 7508W 401 M Street SW. Washington DC 20460, 1996b:35.

US EPA Environmental Protection Agency. Integrated Risk Information System IRIS. http:/ww.epa.gov/iris., 1997.

US EPA. Technical memorandum: topics related to the tributyltin study at the Harbor Island superfund site waster sediment operable unit. US EPA Region 10. Seattle WA, 1999:15.

US Navy. Interim FONSI. Federal Register 1986;50: 120–25748.

US Navy. Navy Program to Monitor Ecological Effects of Organotin. A Report to Congress. Prepared by PS Seligman of Marine Environmental Support Office and Jill Bloom of the EPA Office of Prevention. Pesticides and Toxic Substances and EPA Office of Water. Sept 1 1997. Commonly Referred to as the Navy Risk Assessment for TBT, 1997:76..

Valkirs AO, Davidson B, Kear LL, Fransham RL, Grovhoug JG, Seligman PF. Long-term monitoring of tributyltin in San Diego Bay California. Mar Environ Res 1991;32: 151–167.

Valkirs AO, Davidson B, Kear LL, Fransham RL, Seligman PF, Grovhoug JG. Use of tributyltin by commercial sources and the US Navy: fate-and-effects assessment and management of impacts on the marine environment. pp. 133–166. In: de Peyster A, Day KE editors. Ecological Risk Assessment: A Meeting of Policy and Science. Proceeding from SETAC Workshop Ecological Risk Assessment: A Meeting of Policy and Science: Oct 8–9 1993. San Diego CA. Pensacola FL. Society of Environmental Toxicology and Chemistry, 1998:224.

Wade TE, Garcia-Romero B, Brooks JM. Oysters as biomonitors of butyltins in the Gulf of Mexico. Mar Environ Res 1991;32:233–241.

Waite ME, Waldock MJ, Thain JE, Smith DJ, Milton SM. Reductions in TBT concentrations in UK estuaries following legislation in 1986 and 1987. Mar Environ Res 1991;32:89–111.

Waite ME, Thain JE, Waldock MJ, Cleary JJ, Stebbing ARD, Abel R. Changes in concentrations of organotins in water and sediment in England and Wales following legislation. Chapter 27. In: Champ and Seligman, editors. Organotin: environmental fate and effects. London: Chapman & Hall, 1996:553–580.

Waldock MJ, Miller D. The acute toxicity of bis tributyl tin oxides to the adults and larvae of some marine organisms. ICES CM E 1983;13:5.

Waldock MJ, Thain LE, Miller D. The accumulation and depuration of bis tributyl tin oxide in oysters: a comparison between the Pacific oyster *Crassosstrea gigas* and the European flat oyster *Ostrea edulis*. ICES CM 1983;E52:9.

Waldock MJ. Tributyltin in UK estuaries, 1982–86: Evaluation of the environmental problem. Proceedings Oceans '86 Organotin Symposium. Marine Technology Society. Washington DC, 1986;4:1324–1330.

Wolniakowski KU, Stephenson MD, Ichikowa GS. Tributylin Concentrations and Oyster Deformations in Coos bay, Orgen. Proceedings Oceans '87 International Organotin Symposium. Marine Technology Society. Washington, DC, 1987;4:1438–1442.