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REVIEW OF AQUATIC ENVIRONMENTAL
ASPECTS OF NON-PESTICIDAL ORGANOTIN
COMPOUNDS SCHEDULED FOR ASSESSMENT
UNDER THE CANADIAN ENVIRONMENTAL
PROTECTION ACT

R. James Maguire

NWRI Contribution No. 91-78

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**REVIEW OF AQUATIC ENVIRONMENTAL ASPECTS OF NON-PESTICIDAL
ORGANOTIN COMPOUNDS SCHEDULED FOR ASSESSMENT UNDER THE
CANADIAN ENVIRONMENTAL PROTECTION ACT**

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MANAGEMENT PERSPECTIVE

Non-pesticidal organotin compounds are on the Priority Substances List of the Canadian Environmental Protection Act. They must be assessed to determine whether they are toxic according to the definition specified in section 11 of the Act. This report is a review of the aquatic environmental occurrence, persistence and toxicity of these chemicals. This information will be combined with data on the mammalian toxicity and exposure patterns of the chemicals (currently being prepared by the Department of National Health and Welfare) to provide environmental and human health risk assessments of non-pesticidal organotin compounds.

SOMMAIRE À L'INTENTION DE LA DIRECTION

Des composés organostanniques autres que des pesticides figurent sur la Liste des substances d'intérêt prioritaire de la Loi canadienne sur la protection de l'environnement. Ils doivent être évalués afin de déterminer s'ils sont toxiques aux termes de l'article 11 de la loi. Le présent rapport est une étude de la présence de ces produits chimiques dans les milieux aquatiques, de leur persistance et de leur toxicité. Ces informations seront combinées à des données sur la toxicité pour les mammifères et aux profils d'exposition des produits chimiques (préparées généralement par Santé et Bien-être social Canada) afin de fournir des évaluations relatives aux dangers des produits organostanniques autres que des pesticides pour l'environnement et la santé des humains.

ABSTRACT

Non-pesticidal organotin compounds are on the Priority Substances List of the Canadian Environmental Protection Act. They must be assessed to determine whether they are toxic according to the definition specified in section 11 of the Act. This report identifies those non-pesticidal monomethyltin, dimethyltin, monobutyltin, dibutyltin, monooctyltin and dioctyltin compounds in Canadian commerce, and reviews their uses, methods of analysis, environmental occurrence, and primarily their aquatic toxicity. The single largest use of organotin compounds in Canada and elsewhere is the stabilization of poly(vinyl chloride) (PVC). It is likely that the most important non-pesticidal route of entry of mono- and di- methyltin, butyltin and octyltin to the environment will be through leaching of PVC by water. Monomethyltin and dimethyltin are widespread in the global environment. Monobutyltin and dibutyltin have been found frequently in harbours, marinas and shipping channels in Canada and elsewhere, arising largely as degradation products from the use of the antifouling agent tributyltin which is now regulated in Canada. There are few reports in the literature on the occurrence of butyltin species in the environment as a result of non-pesticidal uses or uses of tributyltin other than as an antifouling agent. Monooctyltin and dioctyltin have not been found to date in Canada or elsewhere in water, sediment, biota, sewage treatment plant influent, effluent and sludges, and landfill leachates. Judging from concentrations which have been reported to date, it appears that the mono- and di- methyltin, butyltin and octyltin species pose no threat to aquatic organisms in Canada *vis-a-vis* acute toxicity. Data on the persistence of these species in aquatic environments are in some cases fragmentary or non-existent, but in general it appears that these species would not be persistent in aquatic environments, with half-lives estimated to be less than a few months at 20 °C.

RÉSUMÉ

Des composés organostanniques autres que des pesticides figurent sur la *Liste des substances d'intérêt prioritaire* de la Loi canadienne sur la protection de l'environnement. Ils doivent être évalués afin de déterminer s'ils sont toxiques aux termes de l'article 11 de la loi. On mentionne dans le présent rapport les composés de monométhylétain, de diméthylétain, de monobutylétain, de dibutylétain, de monooctylétain, et de dioctylétain autres que des pesticides que l'on trouve dans les magasins au Canada, et on étudie leurs utilisations, leurs méthodes d'analyse, leur présence dans l'environnement, et surtout leur toxicité pour le milieu aquatique. Au Canada et ailleurs, les composés organostanniques servent principalement à la stabilisation des chlorures de polyvinyl (PVC). Outre les pesticides, il est probable que le lessivage des PVC par l'eau soit la voie de pénétration la plus importante du monométhylétain, du dibutylétain, du butylétain et de l'octylétain dans l'environnement. Le monométhylétain et le diméthylétain sont largement répandus dans l'environnement à l'échelle mondiale. On trouve souvent du monobutylétain et du dibutylétain dans les ports, les marinas et les voies navigables au Canada et ailleurs, provenant en grande partie des produits de dégradation du tributylétain utilisé comme agent antisalissure et qui est maintenant réglementé au Canada. Il existe peu de rapports sur la présence des espèces de butylétain dans l'environnement ne découlant pas de leur utilisation comme pesticides ou de l'utilisation du tributylétain comme agent antisalissure. La présence de monooctylétain et de dioctylétain n'a pas été relevée au Canada ou ailleurs dans l'eau, les sédiments, le biote, les affluents, les effluents et les boues des usines de traitement des eaux usées, et dans les lixiviats des décharges. D'après les concentrations signalées jusqu'à maintenant, il semble que les espèces de monométhylétain, de diméthylétain, de butylétain et d'octylétain ne constituent pas, au Canada, une menace pour les organismes aquatiques en comparaison d'une toxicité aiguë. Les données sur la persistance de ces espèces dans les milieux

aquatiques sont, dans certains cas, fragmentaires ou inexistantes, mais il semble, en général, que ces espèces ne seraient pas persistantes dans des milieux aquatiques, avec des demi-vies évaluées à moins de quelques mois, à 20 °C.

PROLOGUE

The Canadian Environmental Protection Act (CEPA), proclaimed June 30, 1988, authorizes the Ministers of National Health and Welfare and of the Environment to conduct research and collect information on a wide variety of substances that may contaminate the environment and cause adverse effects on human health or the environment (Government of Canada, 1988). The term "substance" is defined in section 3 of CEPA and for the purposes of the Act includes chemicals in commerce, chemical contaminants in products or environmental media, and complex mixtures of substances in effluent streams and emissions. The estimated 30,000 to 40,000 chemicals that are manufactured in, or imported into, Canada and hundreds of effluent streams and emissions are candidates for assessment of their health and environmental impacts under CEPA. It is not possible to assess simultaneously all the substances that may pose a threat to health or the environment. Therefore it was necessary to select a manageable number that should be given priority for assessment, as required by subsection 12(1) of CEPA which states:

"The Ministers shall compile and may amend from time to time a list known as the Priority Substances List, and the List shall specify substances in respect of which the Ministers are satisfied priority should be given in assessing whether they are toxic or capable of becoming toxic."

Substances that appear on the Priority Substances List must be assessed to determine whether they are toxic according to the definition specified in section 11 of the Act, which states, in part:

"... a substance is toxic if it is entering or may enter the environment in a quantity or concentration or under conditions

- (a) having or that may have an immediate or long-term harmful effect on the environment;
- (b) constituting or that may constitute a danger to the environment on which human life depends; or
- (c) constituting or that may constitute a danger in Canada to human life or health."

In preparing the CEPA Priority Substances List for the first time, the Ministers of the Environment and of National Health and Welfare gave consideration to recommendations of academia, industry, environmental public interest groups and provincial governments, as provided for in subsection 12(3) of CEPA. A substance was selected for the List if it met at least one of the following three criteria:

- (a) The substance causes or has the potential to cause adverse effects on human health or the environment.
- (b) The substance accumulates or could accumulate to significant concentrations in air, water, soil, sediment or tissue.
- (c) The substance is released or may be released into the environment in significant quantities or concentrations.

The CEPA Priority Substances List of February 11, 1989 (Canada Department of Environment, 1989) contains the category non-pesticidal organotin compounds. This report identifies those non-pesticidal organotin compounds in Canadian commerce and reviews their aquatic environmental occurrence, persistence and toxicity (there is relatively little information available on terrestrial aspects). This information will be combined with data on the mammalian toxicity and exposure patterns of the chemicals (currently being prepared by the Department of National Health and Welfare) to provide environmental and human health risk assessments of non-pesticidal organotin compounds.

INTRODUCTION

Tin as a metal and in organic compounds has a wide variety of uses. In the past thirty years organotin compounds have developed into important industrial commodities. Tin is unsurpassed by any other metal in the number of its organic applications, which include uses as poly(vinyl chloride) (PVC) stabilizers, industrial catalysts, industrial and agricultural biocides, and wood preserving and antifouling agents, to mention only the major applications. The annual world consumption of tin in all forms was about 200×10^6 kg in 1976, and of this total about 28×10^6 kg was in the form of organotin compounds (Zuckerman *et al.*, 1978). World consumption of organotins in 1989 was 35×10^6 kg (Blunden and Chapman, 1990). The chemistry and toxicity of tin and organotin compounds have been reviewed extensively (*cf.* World Health Organization, 1980, 1990; Blunden *et al.*, 1984; Thompson *et al.*, 1985; Maguire, 1987; Snoeij *et al.*, 1987; Blunden and Evans, 1990; and references therein).

The increasing use of organotin compounds, some of which are very toxic, attracted the attention of environmental agencies in a number of countries in the 1970s. Organotin compounds as a class were placed on Canada's Environmental Contaminants Act Category III list in 1979 (Canada Department of Environment and Department of National Health and Welfare, 1979), which meant that further information was required on their occurrence, persistence and

toxicity in order to make environmental and human health risk assessments. The organotin compounds that are most likely to be released to the environment in Canada currently are those of methyltin, dimethyltin, n-butyltin, di-n-butyltin, tri-n-butyltin, n-octyltin, di-n-octyltin and hexakis(β, β -dimethylphenylethyl)stannoxane. The mono- and di-methyltins, butyltins* and octyltins are used mainly as PVC stabilizers. Tributyltin is an antifouling agent whose use in Canada is now strictly regulated under the Canadian Pest Control Products Act. It can also be used as a general lumber preservative and as a slimicide in cooling towers. Its use as a net preservative (e.g., in aquaculture) is not allowed. Hexakis(β, β -dimethylphenylethyl)stannoxane is a miticide. In addition to these compounds, triphenyltin may be introduced to coastal and harbour areas in Canada through its use as an antifouling agent on vessels painted in other countries. It is not a registered antifoulant in Canada.

Tributyltin has been the organotin compound of most concern because its use as an antifouling agent results in direct contact with aquatic environments, and because of its extremely high toxicity to aquatic organisms. There has been a great deal of research on environmental aspects of tributyltin over the past ten years [cf. Proceedings of Organotin Symposia of the Oceans '86,

*in this review alkyl groups are n-alkyl unless specified otherwise

'87, '88 and '89 Conferences, Proceedings of the Third International Organotin Symposium (1990), and references therein]. Lethal concentrations are in the range 0.04 - 16 $\mu\text{g Sn/L}$ for short-term exposures of aquatic organisms (e.g., Maguire, 1987; World Health Organization, 1990). In the 1980s an extensive investigation was made of the occurrence and persistence of tributyltin and its less toxic degradation products dibutyltin and monobutyltin in water and sediment in Canada (Maguire, 1987, and references therein). Tributyltin was mainly found in areas of heavy boating or shipping traffic, which was consistent with its use as an antifouling agent. In about 8% of the 269 locations across Canada at which samples were collected, tributyltin was found in water at concentrations which could cause chronic toxicity in a sensitive native species such as rainbow trout. It was also found in about 30% of sediment samples collected across Canada. Biological degradation in water and sediment appears to be the most important factor limiting the persistence of tributyltin in aquatic ecosystems. Estimates of the half-life of biological degradation of tributyltin in fresh water and sediment in Canada are in the range of a few weeks to 4-5 months, respectively.

In 1989 the Canadian government regulated tributyltin under the Pest Control Products Act (Canada Department of Agriculture, 1989). The maximum daily release rate of tributyltin is 4 μg of tributyltin per square cm of hull surface, and it is prohibited on vessels less than 25 m in length, with the exception of those with

aluminum hulls. These regulations should minimize the environmental impact of antifouling uses of tributyltin in Canada.

Organotin compounds in commerce in Canada in the period 1984 - 1986 are listed in Table 1, and their structures are shown in Figs. 1-7. Their Chemical Abstracts Service Registry Numbers, molecular formulas, molecular weights and synonyms are given in the Appendix. The identities of these compounds were obtained from Environment Canada's Domestic Substances List (Canada Department of Environment, 1990). This review and the CEPA assessment will only be concerned with the non-pesticidal organotin compounds in the following classes: monomethyltins, dimethyltins, monobutyltins, dibutyltins, monooctyltins and dioctyltins. These are chemicals 1 - 59 and 62 - 69 of Table 1. Pesticidal organotin compounds (70 - 74) and tin salts of organic anions (75 - 77) are excluded from this review, the former because they are pesticides, and the latter because they are not true organotin compounds, merely salts of the relatively innocuous inorganic tin. In addition, the two tributyltin compounds (60 and 61) with non-pesticidal uses shown in Table 1 are excluded from this review since environmental aspects of tributyltin have been extensively reviewed and since it has been regulated in Canada under the Pest Control Products Act as discussed above.

This document reviews the uses of the above-noted non-pesticidal organotin compounds, analytical methods for their determination, their occurrence in the environment, and their toxicity. The ultimate aim is to determine the chemical hazard to organisms in any environmental "compartment" such as water, sediment or air. The hazard posed by a chemical is a function of its toxicity, its concentration and its persistence.

USES OF NON-PESTICIDAL ORGANOTIN COMPOUNDS

Monoorganotin compounds are mainly used in the stabilization of PVC films during manufacture. Smaller quantities are used for glass coating (e.g., butyltin trichloride). The plastics industry consumes the largest proportion of diorganotin compounds for use as PVC stabilizers, the most important of which are dibutyltins and dioctyltins. Sulfur-containing compounds impart excellent heat stability to PVC, while non-sulfur containing compounds are used to provide resistance to light and weathering. Dioctyltin stabilizers have been used as additives for PVC food packaging products. Other important commercial uses for diorganotins are as catalysts in the production of polyurethane foams, and as agents in the cold curing of silicone rubber. Smaller quantities are used for glass coating (e.g., dimethyltin dichloride), as antihelminthics for poultry, and as stabilizers for lubricating oils, hydrogen peroxide, polyolefins and plastics. Triorganotin compounds are used mainly as biocides. Tetraorganotin compounds are used as stabilizers for oils (corrosion inhibitors), catalysts for olefin polymerization, and as intermediates in organotin synthesis (International Register of Potentially Toxic Chemicals, 1978).

All organotin compounds used in Canada presently are imported, although dibutyltin dilaurate was formerly manufactured by M&T Chemicals in Hamilton, Ontario. In 1979 the total market for organotin compounds in Canada was estimated to be 5.5×10^5 kg per

year, 95% of which was used for stabilizing PVC and as a catalyst for urethane foam manufacture, and the remaining 5% were used as pesticides (Martec, 1979a). On a global basis, it was estimated in the 1970s that 66-75% of all organotins produced were used as stabilizers for PVC. Current organotin use figures constitute confidential business information which is classified as "Protected" under the Canadian Environmental Protection Act, and are not shown here. Before 1980 the Canadian market was dominated by dibutyltin bis(isooctyl mercaptoacetate), which accounted for more than 75% by weight of organotin consumption in Canada (Jones and Millson, 1982). The situation was similar in the U.S.A., where 33×10^6 kg of dibutyltin bis(isooctyl mercaptoacetate) were produced in the period 1965 - 1974, comprising 64% of total organotin production in that period (Martec, 1979a). The chemical is used for its heat stabilization properties in the manufacture of rigid and semi-rigid PVC products which are in turn used extensively in house construction.

The industrial manufacture and use of organotin compounds in PVC stabilization has been reviewed (Bokranz and Plum, 1975; Mori et al., 1979; Davies and Smith, 1980; Blunden et al., 1985; Burley, 1987; Muller, 1989). PVC is degraded both by heat (to which it is subjected during processing at 180-200 °C) and by long-term exposure to sunlight, producing severe discolouration, a rapid deterioration in physical properties, and progressive embrittlement until the polymer disintegrates. This phenomenon is caused by the

elimination of hydrogen chloride from the polymer, starting from the labile, allylic chlorine atoms, and resulting in the formation of a polyene. The degradation may be prevented by the addition of 1 - 1.5% of certain dialkyltin compounds to the polymer before processing. In general, the diorganotin bis(isooctyl mercaptoacetates) are used in applications that require good stability to heat (e.g., in food and drink packaging), whereas the dialkyltin bis(carboxylates) are useful for providing long term stability to light, (e.g., in exterior PVC products). Di-n-octyltin bis(isooctyl mercaptoacetate) and di-n-octyltin maleate have low mammalian toxicity and are used in many countries as stabilizers for PVC food packaging and drink containers. Dimethyltin bis(isooctyl mercaptoacetate) is also approved in some European countries for PVC food packaging, and in the U.S.A. as a heat stabilizer in potable water PVC piping. In most cases up to 60% of the corresponding monoalkyltin compound is added to the dialkyltin stabilizer since it has been found that this combination gives a synergistic improvement in stabilization. The di- and monoalkyltin compounds are considered to be effective as stabilizers because they (i) inhibit the onset of the dehydrochlorination reaction by exchanging their anionic groups, X, with the reactive, allylic chlorine atoms in the polymer, (ii) react with, and thereby scavenge, the hydrogen chloride that is produced and that would otherwise induce further elimination, (iii) produce the compound HX, which may also help to inhibit other undesirable side reactions, and (iv) prevent breakdown of the

polymer initiated by atmospheric oxidation, i.e., by acting as antioxidants (Davies and Smith, 1980). Brooks et al. (1982) have shown that exposure to "artificial sunlight" of PVC sheets which had been stabilized with dibutyltin bis(isooctyl mercaptoacetate) led to complete conversion after 17 d to dibutylmonochlorotin isooctyl mercaptoacetate, the same product observed during the thermal degradation of PVC. Prolonged irradiation produced stannic oxychloride (SnOCl_2) (55% conversion of starting material after 6 months).

In the absence of spills of organotin compounds or discharges from organotin manufacturing plants, it appears that the most important route of entry of non-pesticidal organotin compounds to the environment will be through leaching of PVC pipe by water.

ANALYSIS OF METHYLTIN, BUTYLTIN AND OCTYLTIN SPECIES

The most suitable methods of organotin analysis are those that are species-specific. This is important since the toxicity of organotin compounds varies considerably with the nature and number of the organic groups bonded to the tin atom.

There are very few methods in the scientific literature for the non-pesticidal organotin compounds listed in Table 1 *per se*. However, there are many suitable methods for mono- and dimethyltin, butyltin and octyltin species as simple halides, and it is assumed that these methods will be suitable for all the non-pesticidal organotin compounds listed in Table 1. This assumption is supported by evidence that (i) some organotin compounds such as tributyltin and dimethyltin compounds in water exist as hydrated ions, halide or organic complexes, depending upon the nature and concentration of other solutes (Tobias, 1978; Maguire, 1987), (ii) organotin compounds used in PVC stabilization yield the corresponding chlorides (Poller, 1976; Boettner et al., 1981) or species in which, e.g., one isooctyl mercaptoacetate group is replaced by chloride (H.A. Muller, 1987), (iii) in water migrated tin compounds or those at the surface of PVC are readily hydrolyzed to simple aquo or chloro complexes (H.A. Muller, 1987; Quevauviller et al., 1991), and (iv) the fact that volatile alkyl derivatives of compounds such as dioctyltin bis(isooctyl

mercaptoacetate) can be formed as readily as those of dioctyltin dihalides (e.g., Jitsu *et al.*, 1969; Neubert and Wirth, 1975).

Table 2 lists analytical methods for mono- and di-methyltins, butyltins and octyltins. Many of these methods were developed for the determination of tributyltin or were developed for studies of the aquatic environmental methylation of tin. The most sensitive methods of analysis, adequate for environmental monitoring or toxicity testing, are those that involve conversion of the alkyltin species to volatile hydride or alkyl derivatives for subsequent chromatographic separation and determination. Two methods are especially popular because of their high sensitivity: gas chromatography (GC) with a flame photometric detector (FPD), and atomic absorption spectrophotometry (AAS) employing a quartz tube furnace with sample introduction from a gas chromatograph or by sweeping from a cold trap. They are discussed in more detail below.

The application of GC-FPD to the determination of volatile organotin compounds had its origin in the work of Aue and coworkers (Aue and Flinn, 1977, 1980; Aue and Hill, 1972a,b, 1973; Aue and Hastings, 1973), who demonstrated the remarkable sensitivity and selectivity against hydrocarbons that could be obtained in hydrogen rich flames, both with the FPD and the less sensitive flame ionization detector (FID). Other modifications involved using filters, no filters and metal foil masks. With the use of a quartz

wool plug mounted above the flame of a FPD, Aue and Flinn (1980) were able to detect as little as 0.02 pg (as Sn) of tetrapropyltin. This early work was done with packed column GCs, and tailing of the organotin peaks was observed, as was detector fouling due to SnO_2 formation (Aue and Flinn, 1977; Maguire and Huneault, 1981; Maguire and Tkacz, 1983). To a large degree, the development of capillary columns has obviated these difficulties (M.D. Muller, 1984, 1987; Unger et al., 1986). M.D. Muller's (1987) method is currently the most sensitive capillary column GC-FPD method, with a minimum detectable concentration for butyltin species in water of 0.3 ng Sn/L. This detection limit could probably be lowered if the simultaneous hydridization/extraction technique of Matthias et al. (1986b) were used before analysis by capillary column GC-FPD. Using a packed column GC-FPD method, Matthias et al. (1986b) demonstrated a three-fold improvement in sensitivity with the simultaneous hydridization/extraction technique, compared with combinations of sequential techniques. Ohhira and Matsui (1991) have recently shown that FPD determination of organotin compounds is more sensitive at 610 nm than at 393 nm. A recent refinement of the FPD method using quartz surface-induced luminescence at 390 nm (Jiang et al., 1991), is 30-100 times more sensitive than gas phase luminescence at 610 nm.

The quartz tube furnace AAS technique was first used by Chu et al. (1972) in the determination of arsine. It was first applied to butyltin species by Hodge et al. (1979) and has since undergone a

number of refinements in the determination of hydride and alkyl derivatives of butyltin species. Valkirs *et al.* (1985, 1986) have published a very sensitive quartz tube furnace AAS technique, with a minimum detectable concentration for butyltin species in water of 3 ng Sn/L. This procedure has been automated, allowing four to eight analyses per hour (Clavell *et al.*, 1986).

Newer gas chromatographic detectors for organotin analysis that are even more sensitive than those described above are those employing selected ion monitoring - mass spectrometry (SIM-MS) (e.g., Gilmour *et al.*, 1986; Tolosa *et al.*, 1991), chemical ionization - mass spectrometry (CI-MS) (e.g., Unger *et al.*, 1986) and atomic emission (AED) (e.g., Scott *et al.*, 1991). These methods are capable of determining picogram quantities or less of tin injected.

A rigorous comparison of different methods of analysis of organotins in water has only been done for tributyltin (Blair *et al.*, 1986b). Twenty-seven laboratories worldwide participated, using such techniques as direct solvent extraction, direct hydridization, extraction with hydridization, extraction with alkylation, complexation with spectrophotometric determination and direct injection - high performance liquid chromatography. The conclusion was that all the methods employed can give accurate results for total tin in an aqueous sample. In another comparison, di- and tributyltin compounds present in marine and estuarine

waters at less than 1 $\mu\text{g Sn/L}$ were determined using (i) simultaneous hydride generation - dichloromethane extraction followed by packed column GC-FPD, and (ii) hydride generation followed by sweeping into a quartz tube furnace AAS (Valkirs et al., 1987a). Generally good agreement was obtained from split samples. It is likely that the analytical methods compared in these two studies would be suitable for the non-pesticidal organotin compounds in Table 1.

Concentrations of organotin compounds have been reported in the literature in various ways: (i) as the cation (e.g., $\text{Bu}_2\text{Sn}^{2+}$), (ii) unspecified, and reported as the cation in Table 2, (iii) as the chloride, and (preferably) (iv) in terms of Sn (e.g., ng Sn/L). Reporting concentrations in terms of Sn is preferable because it allows easy calculations of mass balances of homologous species and comparisons of toxicity. In the remaining sections of this review, concentrations are given in the units specified by the original authors. It should be noted that the errors involved in comparing different units reported in occurrence or toxicity studies are quite small compared to experimental errors in determining concentrations of organotin species in environmental media.

ENVIRONMENTAL OCCURRENCE OF METHYLTIN, BUTYLTIN AND OCTYLTIN SPECIES

Analytical methods for organotin compounds have been developed only for the alkyltin or aryltin moiety (e.g., $\text{Bu}_2\text{Sn}^{2+}$) and not for the "anionic" moiety (e.g., chloride, acetate, laurate, etc.). The reason for this is that the toxicity of organotin compounds in general is a function of the kind and number of alkyl or aryl substituents, and not the anionic moiety (cf. toxicity section below). A consequence of the development of the analytical methods is that there are no data on the environmental occurrence of specific organotin compounds *per se* [e.g., dibutyltin bis(isooctyl mercaptoacetate)]. There will thus always be some uncertainty with regard to the exact identity of the organotin compound which was introduced to the environment when a particular organotin species is found. Nevertheless it is possible in some cases to infer with a fair degree of certainty what use of a particular alkyltin or aryltin species led to its being found in the environment.

Many studies have been made of the environmental occurrence of methyltin and butyltin species in particular, the former because of interest in the environmental methylation of tin, and the latter because of concern about tributyltin. Table 3 summarizes the results for monomethyltin, dimethyltin, monobutyltin and dibutyltin. Monooctyltin and dioctyltin species have not been found in the environment, although they have been observed in

laboratory experiments on leaching of PVC. The environmental occurrence of each species is discussed below.

(a) methyltin species

The monomethyltin and dimethyltin species are widespread in the global environment. Byrd and Andreae (1982) found that 5% of the total tin in some rivers in the U.S.A. and Germany was methylated (more in the case of the heavily polluted Rhine River). Braman and Tompkins (1979) found 54% of the total riverine tin in Florida rivers to be in methylated forms. Methyltin species may result from anthropogenic activity, because increasing concentrations of methyltin species with increasing anthropogenic tin influx have been observed in Chesapeake Bay (Brinckman et al., 1983), and dimethyltin has been observed in Germany in effluent from an organotin production plant and in effluent from a sewage treatment plant (Schebek et al., 1991). Alternatively, methyltin species may be produced through methylation of tin naturally present in the environment. Support for this contention is provided by the occurrence of these species in areas remote from industrial contamination (e.g., Braman and Tompkins, 1979; Maguire et al., 1982), and the demonstration that tin can be methylated in aquatic environments (*cf.* environmental dynamics section below). Finally, methyltin species may result from the methylation of tin of industrial origin. It should be noted that the monomethyltin and dimethyltin species found in the environment are not

degradation products of trimethyltin compounds introduced to the environment, because trimethyltin compounds are not used industrially or as pesticides.

In Canada, surveys of the occurrence of methyltin species in water have been made in 30 locations in Ontario in 1980-81 (Maguire et al., 1982), and in water and sediment at 265 locations across the country in 1982-85 (Maguire et al., 1986). In the Ontario survey the monomethyltin species was found in 28 of 30 locations sampled, and the dimethyltin species was found in 27 locations. Each species was found in remote locations such as the Turkey Lakes area 50 km north of Sault Ste. Marie, which suggests a biological origin, although atmospheric transport can not be ruled out because methyltin species have been observed in rain (Braman and Tompkins, 1979). In this survey, generally higher concentrations of both monomethyltin and dimethyltin were observed in industrial areas, in particular in harbours, which suggests anthropogenic input of methyltin species or biological methylation of anthropogenic tin. The highest concentrations observed were 1.2 $\mu\text{g/L}$ for monomethyltin and 0.4 $\mu\text{g/L}$ for dimethyltin.

The results of the cross-Canada survey contrasted sharply with those of the Ontario survey in that the methyltin species were only found infrequently in water and sediment. The monomethyltin species was found in water in only 6 of 265 locations (highest concentration 0.1 $\mu\text{g/L}$), and in sediment in only 14 locations

(although the highest concentration, 17 $\mu\text{g/g}$ dry weight, is the highest concentration in sediment that has ever been reported). The dimethyltin species was not found in any water sample, and in only one sediment sample (at 0.2 $\mu\text{g/g}$). The difference in frequency of occurrence of methyltin species between the two surveys may be related to the fact that the cross-Canada survey included many more locations remote from industrial activity than did the Ontario survey. Thus anthropogenic sources of methyltin species would have been de-emphasized in the cross-Canada survey. It is interesting to note that the very high concentrations of the monomethyltin species observed in sediment in the cross-Canada survey were all found in Atlantic coastal harbours. A small number of fish caught in 1982-84 were also analyzed for methyltin species. Only 4 of 18 fish (smelt and lake trout from Lake Ontario) contained monomethyltin at concentrations of 0.2-0.9 $\mu\text{g/g}$ wet weight. One of the lake trout also contained 0.2 $\mu\text{g/g}$ dimethyltin.

Methyltin species used as PVC stabilizers can be leached by water. For example, Boettner et al. (1981) found that dimethyltin was extracted from new PVC used for cold water delivery. In their experiment extractant water was pumped continuously for 22 days through a 8.9-m rectangular closed loop of 2.54 cm internal diameter PVC pipe having an inner surface area of 0.71 m^2 . At two to four day intervals the water was completely removed for analysis and replaced. The concentration of dimethyltin leached from PVC into pH 5 extractant water at 37 °C was 35 $\mu\text{g/L}$ for day 1, and

decreased from approximately 3.0 to 0.25 $\mu\text{g/L}$ per 24 hours in a biphasic manner from days 2 through 22.

Because methyltin species are used as PVC stabilizers it is expected that they would be found in sewage treatment plant influents in the same way that butyltin species have been found (*vide infra*). For example, in Germany, Schebek et al. (1991) found methyltin species in sewage treatment plant effluents from exclusively domestic sources, which suggested the use of methyltin-stabilized PVC pipes. However monthly sampling of sewage treatment plant influents, effluents and sludges in Montreal, Toronto, Hamilton, Sarnia and Vancouver over a seven month period in 1990-91 revealed no contamination by methyltin species (Chau et al., 1991).

(b) butyltin species

Butyltin species are not produced biologically, and therefore their environmental occurrence is due to anthropogenic input. As seen above, monobutyltin and dibutyltin species have a number of industrial uses, the most important being PVC stabilization. However, in most cases when these species have been found in the environment it has been because of the antifouling use of tributyltin. Tributyltin degrades biologically and abiotically by sequential debutylation, yielding dibutyltin, monobutyltin and inorganic tin (Woggon and Jehle, 1975; Fish et al., 1976; Kimmel et al., 1977; Maguire et al., 1983). Most of the data in Table 3

on the environmental occurrence of monobutyltin and dibutyltin were obtained in the course of surveys for tributyltin, which is generally orders of magnitude more toxic than dibutyltin or monobutyltin (cf. toxicity section below). In shipyards a related source of dibutyltin and monobutyltin is contamination of tributyltin-containing antifouling paint formulations (e.g., Jewett and Brinckman, 1981; Blair et al., 1986a; Adema et al., 1988). Takahashi and Ohyagi (1990) found 0.2 - 0.5% dibutyltin in various free association tributyltin antifouling paints and self-polishing poly(methyl methacrylate - octyl acrylate - tributyltin methacrylate) paints.

In addition to antifouling uses of tributyltin, the application of tributyltin in disinfectants which are applied as waxes, polishes, sprays and in laundry washes may allow contamination of sewage effluents. Industrial discharges of tributyltin used as a slimicide in the paper industry and for textile and lumber treatment, and in cooling water treatment, may be further sources of contamination and through degradation may be sources of dibutyltin and monobutyltin. For example, Blunden and Hill (1988, 1990) showed that tributyltin applied to timber is converted to tributyltin-oxygen species, Bu_3SnOX , which can disproportionate to Bu_4Sn and dibutyltin compounds. Bacci and Gaggi (1989) were among the first researchers to describe unambiguous tributyltin and dibutyltin input to water from an application other than antifouling paint on vessels - i.e., the use

of tributyltin as a slimicide in the cooling water of a large thermoelectric power plant in Leghorn (Livorno), Italy. This plant discharged ca. 7 kg of tributyltin per day.

Fent (1989) described the occurrence of butyltin species in raw wastewater in Zurich, Switzerland (population 350,000). Mean concentrations for three days were 199 ng/L for monobutyltin, 164 ng/L for dibutyltin and 173 ng/L for tributyltin. The monobutyltin and dibutyltin species could have been present through their use as PVC stabilizers or because they were degradation products of tributyltin. Mono-, di- and triphenyltin, tricyclohexyltin and dioctyltin were not detected. About 90% of each butyltin species was associated with particles, and thus removed to a large extent by sedimentation in the first step of a sewage treatment plant (cf. also Fent and Muller, 1991). In sewage sludge, the butyltin species were accumulated to relatively high concentrations. In one sample of digested sludge the following concentrations were determined: monobutyltin, 0.6 $\mu\text{g/g}$ dry weight; dibutyltin, 1.2 $\mu\text{g/g}$; tributyltin, 1.1 $\mu\text{g/g}$. It was concluded that (i) untreated wastewater could give rise to pollution of aquatic systems by butyltin species, (ii) since sludge is used as fertilizer in agriculture, butyltin species could be transferred to soils (but the biological availability of butyltin species in sludges has not been established), and (iii) the dumping of sludge at sea in large amounts could constitute a route of entry of butyltin species to marine ecosystems.

Another route of entry of butyltin species to the environment is leaching from landfills. Fent (1991) found tributyltin, dibutyltin and monobutyltin in a Swiss landfill leachate at concentrations of 18, 243 and 112 ng/L, respectively. More than 50% of each species was associated with particulates $> 1.2 \mu\text{m}$. Phenyltin compounds, tricyclohexyltin and dioctyltin were not found.

Dibutyltin dilaurate is used to control hexamitiasis, an endemic and life-threatening disease in turkeys. It is also used in combination with piperazine and phenothiazine as an anthelmintic in chickens. A recent survey of the incidence of dibutyltin residues in young turkeys in the United States revealed no contamination in turkey muscle tissue; however, dibutyltin residues were found in 8% of all turkey liver samples at concentrations in the range 0.2 - 6 mg/kg (Epstein et al., 1991).

Leaching of PVC by water is likely to be the most important non-pesticidal route of entry of butyltin species to aquatic environments. For example, Boettner et al. (1981) found that dibutyltin was extracted from chlorinated PVC (CPVC) pipe used for hot water delivery. In their experiment extractant water was pumped continuously through a 8.9-m rectangular closed loop of 2.54 cm internal diameter PVC pipe having an inner surface area of 0.71 m², over a 22-day period, during which time the water was completely removed for analysis and replaced at 2- to 4-day

intervals. The concentration of dibutyltin leached from CPVC into pH 5 extractant water at 72 °C was 2.6 µg/L for day 1, and decreased from 1.0 to 0.03 µg/L per 24 hours, again in a biphasic manner, from days 2 through 21. Wu et al. (1989) monitored the leaching of organotins from PVC pipe, and demonstrated that (i) extractable tin in PVC is located at the surface of the pipe, and (ii) the rate of diffusion of tin in the pipe to the pipe surface is negligible compared to the extraction rate of surface tin. Quevauviller et al. (1991) studied the release of monobutyltin and dibutyltin to water after the leaching of five different PVC materials under mild conditions over a period of one month in batch reactor systems. They found a rapid release of dibutyltin, monobutyltin and inorganic tin from new PVC pipe, with 80% of the total tin which was leached having been released within the first two days in static experiments.

In some studies reported to date the occurrence of monobutyltin and dibutyltin has been reasonably attributed to their use in PVC stabilization. Quevauviller et al. (1989) attributed the occurrence of monobutyltin and dibutyltin downstream from mining areas and the city of Alcacer do Sol, Portugal, to the use of organotin-stabilized PVC pipes. In Germany, Schebek et al. (1991) found monobutyltin and dibutyltin in sewage treatment plant effluents from exclusively domestic sources, which suggested the use of butyltin-stabilized PVC pipes.

In Canada, several surveys have been made of the occurrence of butyltin species in water and sediment. As stated above, these studies have been mainly concerned with tributyltin because of its extremely high toxicity. Maguire et al. (1982) found monobutyltin in only 2 of 30 locations sampled in Ontario in 1980-81. One location was a marina (8.48 $\mu\text{g/L}$) and the other location was a harbour (0.02 $\mu\text{g/L}$), which suggested that the monobutyltin found was a degradation product of tributyltin. Dibutyltin was found in 20 of the 30 locations at concentrations from 0.01 to 7.3 $\mu\text{g/L}$. In general, higher concentrations were found in harbours, marinas and shipping channels. This observation was also made in studies of the Detroit and St. Clair Rivers (Maguire et al., 1985a) and Toronto Harbour (Maguire and Tkacz, 1985). Sediment samples were also analyzed from 23 of 30 locations at which the water samples were collected in 1980-81 (Maguire, 1984). Monobutyltin was found in 5 of these 23 locations, all harbours or marinas, at concentrations of 0.01-0.58 $\mu\text{g/g}$. Dibutyltin was found in 6 harbours or marinas at concentrations of 0.01-0.25 $\mu\text{g/g}$. The biological availability of sediment-associated dibutyltin and monobutyltin has not been extensively investigated. However, oligochaetes can take up tributyltin from sediment and can degrade it to inorganic tin (Maguire and Tkacz, 1985), and it is assumed that dibutyltin and monobutyltin in sediment would also be bioavailable and biodegradable.

A survey of the occurrence of butyltin species in water and sediment samples collected across Canada in 1982-85 (Maguire et al., 1986) showed that dibutyltin was found in 27 of 221 water samples at concentrations up to $1.4 \mu\text{g/L}$, and monobutyltin was found in 33 locations at concentrations up to $1.9 \mu\text{g/L}$ (tributyltin was found at 43 locations). It is likely that these species arose from the use of tributyltin because tributyltin was found in 19 of the 27 locations at which dibutyltin was found. Dibutyltin was found in 61 of 235 sediment samples at concentrations up to $8.5 \mu\text{g/g}$, and monobutyltin was found in 47 locations at concentrations up to $4.7 \mu\text{g/g}$ (tributyltin was found at 78 locations). Again, the highest concentrations were in harbours, marinas and shipping channels, and probably resulted from the use of tributyltin as an antifouling agent. A small number of fish caught in 1982-84 was also analyzed for butyltin species. Only 1 of 18 fish samples (a two-fish herring composite from Vancouver Harbour) was found definitely to contain monobutyltin and dibutyltin species, at concentrations of $0.06 \mu\text{g/g}$ and $0.05 \mu\text{g/g}$, respectively. However, some yellow perch and white suckers from harbours on Lake Ontario contained traces of dibutyltin and monobutyltin, in addition to tributyltin. The most likely source of these butyltin species is again the antifouling use of tributyltin.

Other surveys for butyltins in selected Canadian locations have recently been reported. In a study of Severn Sound, Ontario, in 1989, Wong and Chau (1990) found dibutyltin frequently in

sediments, at concentrations up to $0.3 \mu\text{g/g}$. Its presence was probably due to the use of tributyltin antifoulants. Monobutyltin was not detected in these sediments. Chau et al. (1989) also demonstrated the accumulation of dibutyltin in caged mussels in Whitby and Oshawa Harbours in Ontario. Bright and Ellis (1990) found dibutyltin at concentrations up to about $0.35 \mu\text{g/g}$ in gastropods contaminated with tributyltin in coastal waters of British Columbia. However, their method was not specific for dibutyltin, and they did not analyze for monobutyltin. Cullen et al. (1990) found monobutyltin and dibutyltin in coastal waters of British Columbia, largely associated with tributyltin. Concentrations of monobutyltin and dibutyltin were up to 0.02 and $0.06 \mu\text{g/L}$ in seawater and up to 0.03 and $0.18 \mu\text{g/g}$ in oysters, respectively.

Some of the Canadian surveys for butyltin species have included the surface microlayer. Surface microlayers of natural waters (thickness $< 300 \mu\text{m}$) have long been of interest in environmental chemistry because they are often enriched in metals, lipophilic contaminants, nutrients, dissolved and particulate organic matter, and microorganisms (MacIntyre, 1974; Liss, 1975). In particular, contamination of the surface microlayer by high concentrations of toxic substances relative to subsurface water may pose hazards to organisms which spend part or all of their lives at the air-water interface (e.g., Von Westernhagen et al., 1987). Surface microlayer enrichment is also an important phenomenon in

the cycling of contaminants from water to the atmosphere as jet and film drops ejected from air bubbles bursting at the surface of water (MacIntyre, 1974; Liss, 1975; Blanchard and Syzdek, 1975; Piotrowicz et al., 1979).

Monobutyltin was not found in any surface microlayer sample in the 1980-81 Ontario survey, but dibutyltin was found 12 of 30 locations at concentrations up to 2600 $\mu\text{g/L}$. In a later survey of the surface microlayer of 74 locations in Ontario, Quebec and New York State in 1982-85, dibutyltin was found in 19 locations, at concentrations up to 365 $\mu\text{g/L}$ (Maguire and Tkacz, 1987). Monobutyltin was found in 41 locations, at concentrations up to 67 $\mu\text{g/L}$. In these two surveys microlayer enrichments of butyltin species were occasionally so high that the amount of a particular butyltin species in the surface microlayer was either a significant fraction of, or greater than, that in the entire depth of subsurface water. However, these large enrichments were rare, and probably transient because turbulent conditions would mix the surface microlayer with subsurface water. There are few studies of the temporal variability of concentrations of toxic substances in surface microlayers. Pellenbarg and Church (1979) have shown that microlayer enrichments can be significant over long periods. They observed that the microlayer in a Delaware salt marsh carried an average of 10% of the copper, 19% of the zinc and 23% of the iron in terms of the total metal flux, including the dissolved and particulate compartments. In contrast, Maguire and Tkacz (1988)

have shown that over a one year period the surface microlayer did not contribute significantly to the loading of chlorinated hydrocarbons from the Niagara River to Lake Ontario.

In the only Canadian study to date on the occurrence of monobutyltin and dibutyltin arising from either their own use or uses of tributyltin other than antifouling, Chau et al. (1991) analyzed sewage treatment plant influent, effluent and sludge collected monthly for 7 months in 1990-91 from sewage treatment plants in Montreal, Toronto, Hamilton, Sarnia and Vancouver. Concentrations of the butyltin species in the water and sludge samples were similar to those observed in the same type of samples from a sewage treatment plant in Zurich, Switzerland (Fent and Muller, 1991). Monobutyltin was found in all 36 influent and effluent samples. Concentration ranges were 1.9-20.6 $\mu\text{g/L}$ for influent and 0.7-14.5 $\mu\text{g/L}$ for effluent. In 25 of the 36 cases, there was a significant reduction in concentration in passing through the sewage treatment plant, but in no case was there total elimination of monobutyltin. Dibutyltin was found in only 3 influent samples and 1 effluent sample, at concentrations of 1-2 $\mu\text{g/L}$ (tributyltin was only found in 1 influent sample and no effluent sample). Monobutyltin was found in 6 of 36 sewage sludges, with a concentration range of 0.02-0.44 $\mu\text{g/g}$. Dibutyltin was found in 7 of 36 sludges, with a concentration range of 11-305 $\mu\text{g/g}$. These species were only found in sludge when tributyltin was also present, indicating that they likely arose from the use of

tributyltin. However, the ubiquitous presence of monobutyltin in sewage treatment plant influents in the general absence of tributyltin and dibutyltin indicates a non-pesticidal use.

The spreading of sewage sludge on land may constitute another route of entry of organotin species to the environment. However, the biological availability of sludge-associated butyltin species is unknown.

No butyltin species were found in leachate samples collected from five landfills in southern Ontario in 1990 (Chau et al., 1991).

(c) octyltin species

No octyltin species have been found to date in Canada or elsewhere in water, sediment, biota, sewage treatment plant influent, effluent and sludges, and landfill leachates. However, they have been shown to migrate from octyltin-stabilized PVC into edible oils, fat simulants and beer. Most early studies on foodstuffs employed only non-specific methods of analysis for total tin (e.g., Carr, 1969). Woggon et al. (1969) studied the migration of dioctyltin bis(isooctyl mercaptoacetate) from a PVC bottle into sunflower oil, and found 2 mg Sn/L after 6 months. Seidler et al. (1969) also demonstrated migration of dioctyltin bis(isooctyl

mercaptoacetate) from rigid PVC into edible oil, with about 0.45 mg/L accumulating in the oil after exposure for 10 days at 45 °C. Figge (1972) studied the migration of dioctyl[1-¹⁴C]tin bis(isooctyl mercaptoacetate) from PVC films into edible oils and fat simulants with contact times of 60 d at 20°C and 5 hr at 65 °C. At most 0.04% was extracted at 20 °C over 60 d in the various oils. It was not clear from this work if the radioactivity recovered referred to the intact stabilizer or (and perhaps more likely) dioctyltin dichloride. Koch and Figge (1971) studied the migration of dioctyltin bis(isooctyl mercaptoacetate) from PVC bottles into beer. They used a non-specific method for total tin. Beer which had been stored at 20 °C for 8 weeks in bottles stabilized with 1.13% of the octyltin stabilizer contained only 0.0017 mg/L of tin, which equals 0.01 mg/L stabilizer. This was far below the limit allowed by the U.S. Food and Drug Administration at the time (1 mg/L of stabilizer, or 0.16 mg/L tin). Varnes and Gaylor (1978) studied the leaching of dioctyltin bis(maleate) from food-grade PVC resin using a method of analysis for total organic tin. Water extracted < 0.01 mg Sn/L over 6 months at 50 °C, and acetic acid, ethanol and heptane extracted < 0.2 mg Sn/L. H.A. Muller (1987) has shown that the organotin species migrating into (test) foodstuffs from PVC which has been stabilized with dialkyltin-S,S'-bis(alkyl mercaptoacetate) and monoalkyltin-S,S',S''-tris(alkyl mercaptoacetate) are in part the intact stabilizer compounds, but mainly their reaction products. The latter are formed by replacement of one mercaptoacetate group by chloride:

$R_2SnCl(SCH_2COOR')$ and $RSnCl(SCH_2COOR')_2$, respectively (this was for fat-simulating foodstuffs). In aqueous foodstuffs, migrated tin compounds or those at the surface of the PVC appear to be readily hydrolyzed to monooctyltin and dioctyltin species.

(d) derivatives of methyltins and butyltins

In addition to methyltin and butyltin species found in the environment, some hydride derivatives of methyltin species and some methylated butyltins have been found. The likely source of these chemicals is the formation of hydrides of methyltin species and biological methylation of butyltin species, respectively. These chemicals have only been found infrequently at low concentrations relative to the parent methyltins and butyltins, but their occurrence demonstrates that biological transformation of organotin species is possible.

Dimethyltin dihydride was found in estuarine water in Baltimore Harbor at 200 ng/L (Blair et al., 1981; Jackson et al., 1982). Butyltin trihydride was found in Baltimore Harbor at 0.05-0.1 μ g/L (Jackson et al., 1982). Butylmethyltin dihydride was tentatively identified in oysters from English estuaries (Rapsomanikis and Harrison, 1988). This compound may arise from methylation of monobutyltin followed by hydride formation. In Canada, dibutyldimethyltin and tributylmethyltin were found in the sediments of a few harbours sampled in 1980-81 (Maguire, 1984).

This finding raised the possibility that methylation of butyltin species in aquatic environments may be a significant pathway of transformation. The cross-Canada survey has shown, however, that butylmethyln compounds were only found infrequently in water and sediment. For example, dibutyldimethyltin was only found in 4 of 221 water samples, compared with 27 water samples for dibutyltin. It was not found in any sediment sample. Butyltrimethyltin was only found in one of 235 sediment samples, and in no water sample. These results suggested that environmental methylation of butyltin species was in general not a significant pathway of transformation. In addition to methylated butyltin species, tetrabutyltin was found in the cross-Canada survey. However, it was only found in 2 water samples and 1 sediment sample. It might have resulted from the disproportionation of other butyltin species, or, and perhaps more likely, it might have been introduced to water as a contaminant of some pesticidal tributyltin formulation.

ENVIRONMENTAL DYNAMICS AND PERSISTENCE OF METHYLTIN, BUTYLTIN AND OCTYLTIN SPECIES

With regard to the behaviour of a chemical in the environment, it should be noted that there are many factors which influence its persistence, including its physical and chemical properties, and ecosystem-specific properties such as (for aquatic ecosystems) the nature and concentration of dissolved and suspended material, nature and concentration of microbial populations, temperature, degree of insolation, etc. Important physical, chemical and biological removal mechanisms for aquatic ecosystems, for example, are (i) volatilization and adsorption to suspended solids and sediment, (ii) chemical and photochemical degradation or transformation, and (iii) uptake and transformation by microorganisms, respectively. A more detailed description of the way in which physical-chemical properties and ecosystem-specific properties determine the fate of chemicals has recently been given by Howard (1989). Most of the data in the literature on the persistence of organotin compounds refer to the aquatic environment.

In considering the aquatic persistence of organotin compounds there are a few points to bear in mind:

- (1) The persistence of organotins should not be defined in terms of the time required for anion exchange. This gives a false

impression of degradation kinetics when what is significant is the loss of alkyl or aryl groups. In general, the aquatic toxicity of organotin compounds is independent of the nature of the counter ion, but the toxicity declines greatly in the series triorganotin > diorganotin > monoorganotin (cf. toxicity section below).

- (2) The persistence of any chemical in model ecosystems should not be defined solely in terms of dissipation from water if it is not known whether the chemical is simply adsorbed to the sediment or container walls [Some organotin species adsorb to Teflon and glass under certain conditions (Meinema et al., 1978; Maguire et al., 1983)]. Chemicals adsorbed to sediment could be mobilized through simple desorption, sediment resuspension, or ingestion by benthic biota. This point underscores the importance of determining the mass balance in degradation or persistence experiments.
- (3) Although the use of high-pressure mercury lamps in the laboratory yields much useful information on chemical mechanisms, the use of environmentally more appropriate light sources such as sunlight or filtered light of suitable intensity will yield more reliable information on persistence with regard to sunlight photolysis. It should also be noted that sunlight photolysis will be reduced in deep and/or turbid

waters. Dissolved organic matter in natural waters may either enhance or retard the photolysis of other solutes.

The environmental dynamics and persistence of the methyltin, butyltin and octyltin species are summarized below:

(a) methyltin species

Interest in the environmental behaviour of methyltin species is concerned primarily with the phenomenon of environmental methylation of tin, which converts it (i) to more volatile forms subject to atmospheric distribution, and (ii) to more toxic forms.

Huey et al. (1974) first reported that methyltin could be produced by the addition of inorganic tin (II) or tin (IV) to sediments. Since then, many workers have shown that inorganic tin and methyltin compounds can be methylated by microorganisms and plants in nutrient medium or natural water-sediment mixtures (e.g., Chau et al., 1980; Guard et al., 1981; Hallas et al., 1982; Craig and Rapsomanikis, 1984; Maguire and Tkacz, 1985; Gilmour et al., 1985, 1987; Ashby and Craig, 1987; Rapsomanikis et al., 1987; Weber and Alberts, 1990). In general, the yields of methyltin species in these studies were significantly higher under anaerobic conditions than under aerobic conditions, but were much

less than 1%, although a total yield for all four methylated species of 3.2% (0.03% methyltin, 0.08% dimethyltin, 2.86% trimethyltin and 0.19% tetramethyltin) has been reported (Rapsomanikis et al., 1987). Moreover, the production of $\mu\text{g/L}$ concentrations in these studies may have toxicological significance *vis-a-vis* chronic exposures to sensitive organisms. Methyltins were not observed in sterile controls or in poisoned controls in these studies. The mechanism of the environmental methylation of tin and methyltin species is a subject of a great deal of interest. It appears that environmental methylation can be effected chemically by biogenic chemicals (carbanion and carbonium ion routes) and in strictly biological processes. In addition, the production of some methyltin species through disproportionation of other methyltin species has been proposed (Craig and Rapsomanikis, 1984; Rapsomanikis et al., 1987).

Brinckman et al. (1985) have provided a good discussion of chemical mechanisms. It appears that exocellular biogenic metabolites solubilize and methylate heavy metals. Methyl iodide (MeI), ubiquitous in marine environments, solubilizes bulk metals and refractory binary and ternary metal sulfides, producing methylated sulfur coproducts. Stannous sulfide and chloride react with MeI to produce methyltin (IV) species and SnO_2 is solubilized but not methylated by MeI to an as yet unidentified soluble tin species. Dimethyl- β -propiothetin, a common algal metabolite, can react with cell-permeable I^- to produce MeI and with OH^- to form

Me₂S. Several recent studies have demonstrated the methylation of tin by MeI and other chemicals (Ring and Weber, 1988; Shugui et al., 1989a,b; Hamasaki et al., 1991; Ashby and Craig, 1991), and some studies have shown yields of all four methyltins in the range 2-20% (Craig and Rapsomanikis, 1985; Rapsomanikis and Weber, 1985; Lee and Weber, 1988).

Metals in anaerobic sulfide rich sediments are often considered unavailable for biological transformation due to the extreme insolubility of metal sulfides. Gilmour et al. (1985) have, however, shown that sediment tin methylation rates were highest under anaerobic conditions. Sediment slurries spiked with 50 mg/L SnCl₄ produced µg/L concentrations of methyltin and dimethyltin over two months. The same observation was made with whole sediments, as opposed to slurries, even when the concentration of sulfide exceeded that of inorganic tin. The production of methyltin was significantly correlated with numbers of both sulfate-reducing and sulfate-oxidizing bacteria. Heat inactivated slurries did not methylate tin. Yields were < 0.02%. They suggested that methyltin production could be a hazard in anoxic aqueous systems such as drinking water aquifers and in above-ground basins with anaerobic sediment pore waters. However, there have been no studies of the occurrence of methyltin species in such systems.

The environmental production of hydride derivatives of methyltin species (Blair *et al.*, 1981; Jackson *et al.*, 1982; Rapsomanikis and Harrison, 1988) appears to be a rare and unimportant transformation pathway.

Little work has been done on the persistence of methyltin species in aquatic environments. At concentrations commonly found in the environment, species such as dimethyltin likely exist as hydrated cations (Tobias, 1978). The alkyltin (IV) species behave as hard acids, and will tend to interact more strongly with nitrogen and oxygen donors, particularly negatively charged ones, than with sulfur donors.

The distribution of methyltin species depends upon their degree of substitution. In estuarine water methyltin trichloride will be almost completely adsorbed to particulate material, and dimethyltin dichloride will be partly adsorbed and partly dissolved (and trimethyltin chloride will be predominantly in the dissolved phase) (Donard and Weber, 1985). Quevauviller *et al.* (1989) found much lower concentrations of monomethyltin than dimethyltin and trimethyltin in estuarine sediments, which they attributed to the lower lipophilicity of monomethyltin. Wright and Weber (1991) showed that macroalgae could accumulate significant quantities of inorganic tin and methyltin species. It is possible that biological methylation of inorganic tin and/or demethylation of monomethyltin occurred within the protoplasm of the algal cell.

Blunden (1983) studied the photolytic degradation ($\lambda > 235 \text{ nm}$) of methyltin species in water at relatively high concentrations (8.2 g/L). The photolysis of trimethyltin appeared to proceed via dimethyltin directly to inorganic tin without formation of monomethyltin, in contrast to photolysis in carbon tetrachloride, in which trimethyltin was degraded by sequential demethylation to inorganic tin. The half-life of photolysis of trimethyltin in water was 60 h, and half-lives of photolysis of dimethyltin and monomethyltin in separate experiments in which they were the starting materials were estimated to be 300 h and $> 300 \text{ h}$, respectively. If the same mechanism were operative at concentrations of methyltin species in the ng/L - $\mu\text{g/L}$ range observed in the environment, corresponding half-lives of sunlight photolysis would be longer.

By analogy with butyltin species, for which there is more information, it is likely that microbial degradation of methyltin species will be important, and that in the absence of methylating organisms, methyltin species in aquatic environments would not be persistent, with half-lives less than a few months. It may be that in some aquatic environments methyltin species are continuously being methylated and demethylated. Despite the fact that methylation of inorganic tin and methyltin species in aquatic environments is a slow process with apparently low yields in some experiments reported to date, it should not be discounted as a

significant phenomenon over long periods of time which may be instrumental in the global distribution and cycling of tin.

(b) butyltin species

Butyltin species are not produced biologically, but in addition to being taken up by biota (cf. occurrence section) they can be biologically transformed, and they are subject to a variety of physical and chemical processes such as adsorption to suspended solids and sediment, and chemical and photochemical degradation. In addition, ecosystem-specific properties such as those described above will affect their persistence, and their toxicity. Most of the work on the environmental dynamics and persistence of monobutyltin and dibutyltin has been done in the course of studies on tributyltin. Tributyltin degrades biologically and abiotically by sequential debutylation, yielding dibutyltin, monobutyltin and inorganic tin (Akagi and Sakagami, 1971; Casida et al., 1971; Fish et al., 1976; Kimmel et al., 1977; Woggon and Jehle, 1975; Barug, 1981; Maguire et al., 1983, 1984). Biological degradation probably also involves the formation of less stable hydroxylated intermediates (Lee et al., 1987; Ishizaka et al., 1989b).

The logarithms of the n-octanol - water partition coefficients ($\log K_{ow}$) for monobutyltin trichloride, dibutyltin dichloride and

tributyltin chloride are 0.09, 0.05 and 2.2, respectively (Tsuda et al., 1988), which indicates that monobutyltin and dibutyltin would not be bound to the organic portion of sediment, or taken up by biota, to the same extent as the more lipophilic tributyltin species. This conclusion was confirmed by the relative order of the maximum logarithm of the bioconcentration factor for carp (*C. carpio* L.) tissues: 3.5 for tributyltin, 1.0 for dibutyltin and 2.1 for monobutyltin (Tsuda et al., 1988). A similar pattern was seen for rainbow trout (*S. gairdneri* = *O. mykiss*) (Martin et al., 1989).

Although it appears true that monobutyltin and dibutyltin are bound to a lesser extent to sediments and suspended particulate matter than is tributyltin, there are conflicting reports in the literature concerning the extent of binding. These discrepancies can probably be attributed to (i) differences in composition of the various solid phases (including the possibility that some species are associated with paint chips), (ii) differences in the nature and concentrations of other solutes, including differences in salinity and dissolved organic matter concentrations, and (iii) a lack of equilibrium in harbour areas close to active sources of tributyltin. Hinga et al. (1987) determined that the mean sediment-water partition coefficient for dibutyltin was about 100 times smaller than that for tributyltin (values were 4.5×10^2 and 4×10^4 , respectively). Stang and Seligman (1987) noted that there was a wide variation in adsorption coefficients of butyltin species

to sediments which may reflect lack of equilibrium and differences in sediment composition. They determined partition coefficients for different sediments in the following ranges: tributyltin, 6.2×10^3 – 5.5×10^4 ; dibutyltin, 2.1×10^3 – 2.6×10^4 ; monobutyltin, 1.7×10^3 – 2.9×10^4 , and they noted that the presence of paint chips may have affected these values. A consequence of the lower affinity for sediment of dibutyltin and monobutyltin compared to tributyltin may be that when tributyltin is degraded in sediment, dibutyltin and monobutyltin are released to the water column (Seligman et al., 1989).

Tributyltin does not volatilize from water (Maguire et al., 1983), and it is unlikely that the more hydrophilic dibutyltin and monobutyltin would volatilize. In 90% water - 10% acetonitrile solutions, monobutyltin and dibutyltin were stable for at least 9 days, but were degraded by light of wavelength 300 nm (Maguire et al., 1983). The half-life of degradation of monobutyltin was 0.4 day, and after 9 days the concentration of inorganic tin accounted for about 70% of the initial monobutyltin concentration. The half-life of dibutyltin was > 9 days (at which time there was about 30% decomposition), and monobutyltin and inorganic tin were the only products observed. The half-life for tributyltin was about 1 day. The butyltin species are less stable at acidic pH values. At pH 2 in seawater under laboratory light, half-lives of disappearance of monobutyltin, dibutyltin and tributyltin were < 1, 1 and 4 days, respectively (Burns et al., 1987).

The persistence of dibutyltin and monobutyltin species in aquatic ecosystems is likely also to depend strongly on ecosystem-specific characteristics such as temperature and the kinds and concentrations of butyltin-tolerant and -degrading organisms, as is the case with tributyltin (e.g., Maguire, 1987; Lee et al., 1987; Seligman et al., 1990; Meyers-Schulte and Dooley, 1990). Reported half-lives for dibutyltin are 5 to > 15 days for fresh water and seawater in Japan (Hattori et al., 1988), and 1-17 days for seawater in the U.S.A. (Francois et al., 1989; Adelman et al., 1990). Thain et al. (1987) estimated a half-life of 90 d for dibutyltin in seawater at 5 °C. Reported half-lives for monobutyltin are 2-14 days for seawater in the U.S.A. (Francois et al., 1989); however, Adelman et al. (1990) found no degradation of monobutyltin in seawater over a 50 day period. Despite the range of values for the half-lives of monobutyltin and dibutyltin, most work indicates that these chemicals are not usually persistent in aquatic ecosystems.

The environmental production of methyl and hydride derivatives of butyltin species (Jackson et al., 1982; Maguire, 1984; Maguire and Tkacz, 1985; Maguire et al., 1986; Rapsomanikis and Harrison, 1988) appears to be a rare and unimportant transformation pathway.

(c) octyltin species

The literature on the environmental persistence of monooctyltin is scanty. Akagi and Sakagami (1971) showed that dioctyltin could be degraded by ultraviolet light (from a sterilizing lamp of unknown wavelengths) to monooctyltin and dioctyltin. Absolute kinetics of degradation of dioctyltin were not reported, but it appeared to degrade at about one third the rate of dibutyltin. Mazayev et al. (1976) reported half-lives of 2-6 days in water and water-sediment systems, and described the reaction as "hydrolysis" (probably meaning cleavage of octyl groups). However, a non-specific method of analysis was used, the products were not stated, and little experimental detail was given. It is difficult to estimate the persistence of monooctyltin and dioctyltin in aquatic ecosystems using only these two studies as a basis for prediction.

TOXICITY OF METHYLTIN, BUTYLTIN AND OCTYLTIN SPECIES

The toxicity of tin compounds has been studied extensively (e.g., Luijten, 1972; Piver, 1973; U.S. Environmental Protection Agency, 1975; Hall and Pinkney, 1985; Snoeij et al., 1987; Cooney and Wuertz, 1989; Boyer, 1989; and references therein). Organotin compounds are more toxic than inorganic tin compounds. Progressive introduction of organic groups to the tin atom in any $R_nSn^{(4-n)+}$ series produces maximal biological activity against all species when $n = 3$, i.e., for the triorganotin compounds. However, within the class of triorganotin compounds there are considerable variations in toxicity with the nature of the organic substituents (Davies and Smith, 1980). For insects, trimethyltin compounds are the most toxic; for mammals, the triethyltin compounds; for Gram-negative bacteria, the tri-n-propyltin compounds; for Gram-positive bacteria, yeasts, fungi and fish, the tri-n-butyltin compounds. Further increase in the n-alkyl chain length produces a sharp drop in toxicity. Triphenyltin compounds are particularly toxic to fungi and phytoplankton (Wong et al., 1982), while tricyclohexyltin compounds show high acaricidal activity (Davies and Smith, 1980).

The variation of X within any particular series of R_3SnX compounds usually has little effect on biological activity (Polster and Halacka, 1971; Davies and Smith, 1980). In addition, some mammalian toxicity data for dialkyltin and monoalkyltin compounds

have indicated only a modest variation in toxicity with nature of X substituent (World Health Organization, 1980). However, it is reasonable to expect that a mechanism featuring binding of dialkyltin compounds to dithiol groups, which has been proposed for dialkyltin compounds, would show some dependence of toxicity on the nature of the X groups. This dependence is borne out by a study of the acute oral toxicities to rats of Me_2SnCl_2 and $\text{Me}_2\text{Sn}(\text{SCH}_2\text{CO}_2\text{Oct}^i)_2$, where the former compound which would be expected to react readily with dithiol groups is moderately toxic, whereas the latter in which two Sn-S bonds are already present in the molecule is essentially non-toxic (Davies and Smith, 1982). It may be that aquatic organisms show less sensitivity to the nature of the X substituent than do mammals in toxicity tests because dissolution of many organotin compounds in water yields simply the organotin cation, whereas with oral administration of organotin compounds to mammals there may not be enough time for the X substituents to be cleaved.

The mechanism of toxic action of triorganotin compounds appears to be through disruption of oxidative phosphorylation, by (a) secondary responses caused by discharge of a hydroxyl-chloride gradient across mitochondrial membranes, (b) interaction with the basic energy conservation system involved in the synthesis of ATP, and (c) an interaction with mitochondrial membranes to cause swelling and disruption (Selwyn, 1976; Aldridge, 1976). [The biological effects of tetraorganotins in mammals appear to be

caused principally by their biologically- and abiotically-produced triorganotin degradation product (Davies and Smith, 1980).] Dialkyltin compounds also show the same trend of decreasing toxicity with increasing alkyl chain length observed with trialkyltin compounds, and they are considerably less toxic than trialkyltin species. The mechanism of toxic action may be through interfering with alpha-keto acid oxidation, possibly by binding to dithiol groups (Cain et al., 1977; Penninks and Seinen, 1980; Davies and Smith, 1980; Penninks et al., 1983). Dimethyltin dichloride inhibits RNA polymerase from *E. coli* (Yamada, 1977). [Dialkyltin compounds more strongly inhibit the polymerase reaction than trialkyltin compounds.] Dibutyltin dichloride and dioctyltin dichloride are immunosuppressive compounds that impair thymus-dependent immunity in animals (Vos, 1977; Li et al., 1982; Smialowicz et al., 1988). Little work has been done on the mechanism of toxic action of monoorganotin compounds since these are far less toxic than the triorganotin or even the diorganotin compounds.

Some notable effects of organotin compounds on aquatic and terrestrial organisms are described below, followed by a discussion primarily of the acute toxicities of the mono- and di- methyltin, butyltin and octyltin species which are summarized in Table 4, and their relevance to concentrations which have been found in the environment.

Aquatic organisms

The toxicity of organotin compounds to aquatic organisms has received a good deal of attention in the past ten years (cf. reviews of Hall and Pinkney, 1985, and Cooney and Wuertz, 1989). Wong et al. (1982) showed a direct relationship between the toxicity of organotin compounds to algae and their octanol-water partition coefficients. Cooney et al. (1989) tested the effects of various organotins on 29 yeasts representing 10 genera. Yeasts varied in their sensitivity from strain to strain, but tributyltin was the most toxic compound tested. Mono- and dimethyltin were the least toxic. Triphenyltin, dibutyltin, monobutyltin, trimethyltin, triethyltin and diethyltin showed intermediate toxicity. Eng et al. (1988) showed that the total surface area parameter is an excellent predictor of toxicity provided that the toxicity is primarily related to hydrophobicity, as it is for diorganotin compounds, rather than to electronic or steric effects. Organisms included in this study were neuroblastoma cells, fibroblasts, algae (*A. falcatus*), water fleas (*D. magna*), and mud crabs (*R. harrisii*).

Some of the toxicity studies summarized in Table 4 also demonstrate the marked dependence of toxicity on the number of alkyl substituents bonded to tin (cf. Wong et al., 1982; Walsh et al., 1985; Vighi and Calamari, 1985; Laughlin et al., 1985; Dooley and Kenis, 1987; Salazar and Salazar, 1989; Josephson et al., 1989; Nagase et al., 1991). For example, concentrations of

butyltin species which produced a 50% inhibition in the alga *A. falcatus* were 25, 6.8 and 0.02 mg/L for monobutyltin, dibutyltin and tributyltin, respectively (Wong et al., 1982).

Holwerda and Herwig (1986) exposed freshwater clams (*A. anatina*) to 38 µg/L dibutyltin chloride for 7 months (weekly static renewal). They noted decreases in weight and carbohydrate stores. The kidney was the main site of accumulation of total tin. Intoxicated clams appeared to rely on anaerobic energy metabolism as indicated by elevated levels of lactate, succinate, acetate and propionate.

Wester and Canton (1987) investigated histopathological effects in guppies (*Poecilia reticulata*) after exposure to bis(tributyltin) oxide (TBTO) and dibutyltin dichloride (DBTC). Both compounds induced thymus atrophy, hyperplasia of hemopoietic interstitial tissue of the kidney, and increase in number of free peritoneal cells in the body cavity, which were probably neutrophilic granulocyte analogs. In the liver vacuolation was increased, predominantly due to accumulation of fat and glycogen; also muscle glycogen was increased. Effects of dibutyltin occurred at concentrations which were almost three orders of magnitude higher than that of tributyltin, and there were no indications that the tributyltin-induced thymus atrophy was caused by the dibutyltin metabolite. In addition, at the higher concentrations of tributyltin epidermal hyperplasia and keratitis occurred, as well

as changes in the rods and cones layer of the retina. The "no-observed-effect concentration" (NOEC) of TBTO was 0.01 $\mu\text{g/L}$, based on thymus atrophy, liver vacuolation or hyperplasia of the hemopoietic tissue, and this was 30-fold lower than when based upon classical toxicological criteria (growth). In the dibutyltin study thymus atrophy was still present at the lowest concentration tested (320 $\mu\text{g/L}$) whereas the NOEC for mortality and behaviour was 1800 $\mu\text{g/L}$. Atrophy of the thymus, keratitis and skin irritation seem to be similar to that found in mammals. It is concluded that long-term low dose tributyltin exposure of guppies causes thymus atrophy, increase of granulocytes, accumulation of glycogen and fat in liver, and changes in cornea, retina and skin. Except for the changes in the eye and skin, similar effects were induced by dibutyltin, although at manifold higher concentrations.

Wester et al. (1990) did not observed thymic atrophy in the medaka fish and rainbow trout after exposure to butyltin compounds, although it has been observed in the rat and the guppy. Thus the thymotoxicity of butyltins appears to be a species-specific phenomenon. In the medaka TBTO and DBTC have effects on the liver and gas gland, probably as a result of impaired glycogen breakdown. In addition, these compounds were toxic to kidney and retina. NOEC levels using histopathology were 0.32 $\mu\text{g/L}$ for TBTO and < 320 $\mu\text{g/L}$ for DBTC.

De Vries et al. (1991) investigated the comparative toxicity of various organotin compounds in early life stages of rainbow trout (*O. mykiss*). Beginning with yolk sac fry, trout were continuously exposed for 110 days. DBTC and diphenyltin dichloride were about 1000 times less toxic than tributyltin chloride (TBTC1) and triphenyltin chloride. For DBTC, a NOEC of 40 µg/L was found. This concentration is about 2.5 times higher than the maximum concentration observed for subsurface natural waters, although it has been exceeded occasionally by concentrations in surface microlayers in Canadian harbours (cf. Table 3). For TBTC1 the NOEC was 40 ng/L. Histopathological examination revealed depletion of glycogen in liver cells of both di- and triorganotin-exposed fish, except in the case of trimethyltin. Atrophy of the thymus, the most prominent sign of toxicity of di- and tributyltin compounds in mammals, was not observed in early life stages of rainbow trout. Tail melanization (blackening) was observed at dibutyltin concentrations five-fold higher than the NOEC. At the end of the exposure period, resistance to infection was examined by an intraperitoneal challenge with *Aeromonas hydrophila*, a bacterium pathogenic to fish. Resistance to the bacterial challenge was decreased with dibutyltin at 5 x NOEC. This might be indicative of a suppressed immune function or generally diminished fish health.

Terrestrial organisms

In the 1950s it was found that poisoning of rats and mice by dibutyltin salts was accompanied by damage to the biliary tract and the liver (Barnes and Stoner, 1958). Bile-duct damage in rats could be produced by a single oral dose of dibutyltin dichloride of 50 mg/kg body weight. This dose on three successive days killed 30-50% of the rats (Barnes and Magee, 1958).

A great deal of work on the effects of organotin compounds in animals has been done over the past twenty years by a research group at the University of Utrecht in The Netherlands (cf. Snoeij et al., 1987 and references therein). Seinen et al. (1977a) observed for DBTC and dioctyltin dichloride (DOTC) a dose-related reduction in the weight of the thymus, spleen and popliteal lymph, associated with lymphocyte depletion in the cortex of the thymus and the thymus-dependent areas in female weanling rats fed for two weeks at dietary levels of 0, 50 and 150 mg/kg. In contrast dimethyltin dichloride and octyltin trichloride did not cause atrophy of the lymphoid organs. Immune suppression was most pronounced in rats exposed to DBTC and DOTC during the developmental phase of the lymphoid system (Seinen et al., 1977b). Altered immune functions were not found in mice or guinea pigs exposed to DBTC or DOTC. From this study it was concluded that both DBTC and DOTC induced immune suppression in rats by a selective inhibition of T-lymphocyte activity.

Mazur (1979) studied the effects of dioctyltin bis(isooctyl mercaptoacetate) on pregnant female rats (Wistar variety), Syrian hamsters and white rabbits. The compound was administered during organogenesis by esophageal probe (rats and hamsters) in doses of 1/1000, 1/500, 1/250, 1/100 and 1/50 LD₅₀ (which for the rats was 100 mg/kg), or stomach probe (rabbits) in doses of 1/1000, 1/500 and 1/100 LD₅₀. There was some evidence for embryotoxicity (skeletal abnormalities in rat and hamster, but rabbit much less sensitive). For rats, a dose of 1/50 of the LD₅₀ caused maternal toxicity. There was no direct relation between the level of organotin dose and extent of developmental abnormalities, and no conclusive evidence for teratogenicity.

As stated above, the metabolism of butyltin species, at least, appears to involve hydroxylated intermediates. Ishizaka et al. (1989b) found the acid-stable intermediates butyl(3-hydroxybutyl)tin dichloride, butyl(4-hydroxybutyl)tin dichloride and butyltin trichloride in male rats after intraperitoneal administration of dibutyltin dichloride. The major acid-stable intermediate, butyl(3-hydroxybutyl)tin dichloride, showed a tendency to accumulate in the kidney. Butyl(4-hydroxybutyl)tin dichloride was detected only in urine extracts. Dibutyltin dichloride, butyl(3-hydroxybutyl)tin dichloride and butyltin trichloride were found in the brain in spite of differences in their relative polarities.

To date the only reported case of poisoning by a chemical under review here has been the accidental poisoning of cattle, birds and mink in Israel by dibutyltin dilaurate which was added to turkey feed to control hexamitiasis (Shlosberg and Egyed, 1978).

Meyer et al. (1987) assessed human health data collected at a plant which produced organic and inorganic tin chemicals. No chronic health problems related to occupational exposure were detected by biological testing history or physical examination. A statistical evaluation of blood parameters was conducted in an attempt to detect any significant changes. The blood values of all workers were within normal limits. Workers in both the organic and inorganic tin production areas of the plant were found to have statistically significant reductions ($p < 0.05$) of their red blood cell count, hemoglobin and hematocrit levels when compared with either a non-exposed or a control group. In both analyses, the changes in the blood values were too small to be of clinical concern.

There are some therapeutic uses of organotin compounds. Dibutyltin oxide has been used for the treatment of intestinal flukes in trout (Mitchum and Moore, 1969), and lately dibutyltin dichloride, dioctyltin dichloride and some other complexes of diorganotin compounds have shown some promise as antiviral agents and in cancer chemotherapy (Cardarelli et al., 1984; Penninks et

al., 1986; Ward et al., 1988; Penninks et al., 1989; Saxena and Huber, 1989).

Acute toxicity

Table 4 summarizes the results of toxicity tests of the mono- and di- methyltin, butyltin and octyltin species to aquatic and terrestrial organisms, and their significance is discussed below.

A complication in studies of the toxicity of monoorganotin and diorganotin compounds may be the presence of triorganotin compounds which can be orders of magnitude more toxic. There appears to have been only one article in the literature which has considered this point. Wester et al. (1990) used dibutyltin dichloride of 98% purity in a study of the effects of dibutyltin and tributyltin on fish (medaka). NOEC levels using histopathology were 0.32 $\mu\text{g/L}$ for tributyltin and $< 320 \mu\text{g/L}$ for dibutyltin, a difference of approximately one thousand. However, considerable amounts of tributyltin were found in dibutyltin-exposed fish, and subsequent analysis indicated that the dibutyltin dichloride used contained tributyltin chloride at a concentration (0.33%) that could have accounted for the observed toxicity of dibutyltin. This point should be kept in mind in considering the toxicity of monoorganotin and diorganotin compounds in the studies summarized below, although it should be noted that the toxicity of organotin compounds is generally manifested in different ways, depending upon the degree

of substitution. Nevertheless, it may be that reported toxicities of monoorganotin and diorganotin compounds are only upper limits.

(a) methyltin species

Table 4 shows that concentrations of monomethyltin that inhibit 50% of growth (i.e., EC_{50} or IC_{50}) of bacteria, yeasts, *D. magna* and some algae are greater than 1 mg/L. However, some diatoms are inhibited at concentrations as low as 0.08 mg/L. The figure of 0.08 mg/L is 67 times higher than the highest concentration of monomethyltin observed in water. Acute oral LD_{50} values for monomethyltin in rats are in the range 0.9-1.4 g/kg body weight.

Concentrations of dimethyltin that inhibit 50% of bacteria, algae, yeasts and *D. magna*, and that kill 50% of mud crab larvae and some fish are all greater than 0.07 mg/L, and are usually greater than 1 mg/L. The figure of 0.07 mg/L is about 150 times higher than the highest concentration of dimethyltin observed in water. Acute oral LD_{50} values in the rat are 0.08 and 1.2 g/kg body weight for dibutyltin dichloride and dibutyltin bis(isooctyl mercaptoacetate), respectively.

(b) butyltin species

Table 4 shows that concentrations of monobutyltin that inhibit 50% of growth of bacteria, yeasts, an alga and *D. magna*, and that kill 50% of red killifish, are all greater than 5 mg/L, which is 2600 times higher than the highest concentration observed in natural waters, and 240 times higher than the highest concentration observed in sewage treatment plant influents. Acute oral LD₅₀ values for monobutyltin in mice and rats are in the range 1-3 g/kg body weight.

The NOEC for early life stages of rainbow trout (*O. mykiss*) is 0.04 mg/L (de Vries et al., 1991). Concentrations of dibutyltin which inhibit 50% of bacteria, algae, yeasts, zooplankton, and *D. magna*, and which kill 50% of algae, mosquito larvae, polychaetes, mud crab larvae, adult crabs, red killifish, and golden orfe are also all \geq 0.04 mg/L, which is about 2.5 times greater than the highest concentration which has ever been observed in the most contaminated harbours and marinas, and many more times higher than concentrations usually observed. A concentration of 0.04 mg/L is 20 times higher than the highest concentration observed to date in sewage treatment plant effluent in Canada. It should be noted that dibutyltin has been found in surface microlayers in Canadian harbours and marinas at concentrations up to 2.6 mg/L, a concentration which, if sustained, is acutely toxic to many species

of aquatic organisms. Acute oral LD₅₀ values for dibutyltin in mice and rats are in the range 0.06-1.3 g/kg body weight.

(c) octyltin species

There appear to be no data on the toxicity of monooctyltin to aquatic organisms. Acute oral LD₅₀ values for monooctyltin in rats and mice are in the range 1.5-4.7 g/kg body weight.

There are few data on the toxicity of dioctyltin to aquatic organisms (cf. Table 4). Steinhauser et al. (1985) showed that dioctyltin was very toxic to bacteria and *D. magna*, with EC₅₀ values of 0.002 and 0.005 mg/L, respectively. By contrast, the 24-h LC₅₀ value for dioctyltin against mosquito larvae was > 2.5 mg/L. There appear to be no data on the environmental occurrence of dioctyltin with which to compare the toxicity values. The results of Steinhauser et al. (1985) indicate that dioctyltin is more toxic to bacteria and *D. magna* than monomethyltin, dimethyltin, monobutyltin and dibutyltin. This result is unexpected in view of the general trends in toxicity in the literature, and it should be verified. Acute oral LD₅₀ values for dioctyltin in mice and rats are in the range 0.8-7 g/kg body weight.

CONCLUSION

This review of environmental aspects of non-pesticidal organotin compounds in use in Canada will serve as a background document for the CEPA assessment. Following are conclusions, identified data gaps and research needs, largely from an aquatic environmental perspective:

(i) organotin use

Based on data to 1986, PVC stabilization is the single largest use of organotin compounds in Canada and elsewhere. It is likely that the most important non-pesticidal route of entry of mono- and di- methyltin, butyltin and octyltin to the environment will be through leaching of PVC by water.

A survey of industry will be made in 1991 under Section 16 of CEPA to determine what non-pesticidal organotins are presently being used in Canada, for what purpose, in what geographical areas and in what quantities.

(ii) analytical methods for organotins

Methods currently available are suitable for the monitoring of environmental concentrations of organotins, and for toxicity studies. They tend to be time-consuming and expensive.

Environmental monitoring agencies might consider a less expensive screening step whereby sample extracts in organic solvents are analyzed by atomic absorption spectrophotometry for total organic tin before a full-scale speciation technique is employed.

(iii) environmental occurrence of methyltin, butyltin and octyltin species

Monomethyltin and dimethyltin are widespread in the global environment. They may arise from the environmental methylation of tin or as a result of anthropogenic input. They have been found in remote areas of Canada, but generally higher concentrations were observed in industrial areas, in particular in harbours, which suggests that there is some anthropogenic source. However, the source may environmental methylation of tin of anthropogenic origin rather than input of methyltin species because such species were not found in a study of sewage treatment plant influents, effluents and sludges in five Canadian cities, nor were they found in leachates from five landfills in southern Ontario. Concentrations of monomethyltin and dimethyltin that are acutely toxic to some aquatic organisms are 67 and 150 times higher, respectively, than the highest concentrations of these species reported for natural waters.

Monobutyltin and dibutyltin have been found frequently in harbours, marinas and shipping channels in Canada and elsewhere.

Their presence was due to antifouling uses of tributyltin, whose degradation proceeds by sequential debutylation, yielding dibutyltin, monobutyltin and finally the practically non-toxic inorganic tin. It is anticipated that tributyltin contamination of the Canadian environment will decline substantially with recently imposed controls. There are few reports in the literature in which the environmental occurrence of the monobutyltin and dibutyltin species can be unambiguously attributed to either non-pesticidal uses of the species themselves, or non-antifouling uses of tributyltin. However, the recent study of Chau *et al.* (1991) showing the widespread occurrence of monobutyltin in sewage treatment plant influents (and effluents), coupled with the lower frequency of occurrence of tributyltin, may indicate some non-pesticidal input to the environment. No butyltin species were found in leachates from five landfills in southern Ontario.

Concentrations of monobutyltin that are acutely toxic to some aquatic organisms are 2600 times higher than the highest concentration reported for natural waters, and 240 times higher than the highest concentration observed in sewage treatment plant influents. Concentrations of dibutyltin that are acutely toxic to aquatic organisms are 2.5 times higher than the highest concentration observed in the most contaminated harbours and marinas (and many more times higher than concentrations usually observed in natural waters), and 20 times higher than the highest concentration observed to date in sewage treatment plant effluent in Canada.

Monooctyltin and dioctyltin have not been found to date in Canada or elsewhere in water, sediment, biota, sewage treatment plant influent, effluent and sludges, and landfill leachates.

(iv) environmental dynamics and persistence of methyltin, butyltin and octyltin species

Methyltin species can be produced environmentally, and may be distributed atmospherically. It may be that in some aquatic environments methyltin species are continuously being methylated and demethylated. In the absence of methylating organisms, it is likely, by analogy with butyltin species, that microbial degradation of methyltin species will be important, and half-lives at 20 °C are expected to be less than a few months.

The most important factors limiting the persistence of monobutyltin and dibutyltin in aquatic environments are likely to be microbial degradation and sunlight photolysis. Half-lives at 20 °C are expected to be less than a few months. Half-lives of biological degradation would of course be slower at lower temperatures.

So few data are available on monooctyltin and dioctyltin in aquatic ecosystems that it is difficult to estimate their persistence with confidence. However, by analogy with the

methylytin and butylytin species, it is expected that the octylytin species would not be persistent in aquatic environments (i.e., half-lives would probably be less than a few months).

(v) toxicity of methylytin, butylytin and octylytin species to aquatic organisms

Most of the aquatic toxicity studies on non-pesticidal organotin compounds in the literature refer only to acute toxicity. Judging from concentrations which have been reported to date, it appears that the mono- and di- methylytin, butylytin and octylytin species pose no threat to aquatic organisms in Canada *vis-a-vis* acute toxicity.

There are no aquatic toxicity data for monooctylytin, and more data are needed on the aquatic toxicity of dioctylytin, in particular a verification of its reported high toxicity to bacteria and *D. magna*.

The biological availability and toxicity of sediment-associated (and sewage sludge-associated) organotin species in general is an area which requires more research, as does the relationship between the degree of accumulation in aquatic organisms and chronic toxicity. However, these problems are not as urgent for the organotin species under review as they are for the more lipophilic and toxic triorganotin species.

The contamination of dibutyltin formulations by the much more toxic tributyltin may lead to over-estimations of the toxicity of dibutyltin, as discussed earlier. The CEPA Section 16 notice referred to above will also require organotin importers to state the relative percentages of mono-, di, tri- and tetraorganotin constituents in any particular organotin product.

ACKNOWLEDGEMENT

I thank E.D. Brien of the Commercial Chemicals Branch of the Department of Environment for the information on organotin compounds in commerce in Canada in 1986.

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Table 1. Organotin compounds in commerce in Canada in 1984 - 1986.*

No.	Chemical Name	CAS No.
(A) Non-pesticidal		
<u>Monomethyltin compounds</u>		
1	methyltin trichloride	993-16-8
2	methyltin sulfide	33397-79-4
3	methyltin tris(laurylmercaptide)	52165-03-4
4	methyltin tris(isooctyl mercaptoacetate)	54849-38-6
5	methyltin tris(2-ethylhexyl mercaptoacetate)	57583-34-3
6	methyltin tris(2-mercaptoethyl oleate)	59118-79-5
7	methyltin tris(lauroyloxyethylmercaptide)	67859-62-5
8	methyltin tris(2-tetradecanoyloxyethylmercaptide)	68928-38-1
9	methyltin tris(2-linoleoyloxyethylmercaptide)	68928-40-5
10	methyltin tris(decyloxyloxyethylmercaptide)	68928-50-7
<u>Dimethyltin compounds</u>		
11	dimethyltin dichloride	753-73-1
12	dimethyltin sulfide	13269-74-4
13	dimethyltin bis(isooctyl mercaptoacetate)	26636-01-1
14	dimethyltin bis(isooctyl mercaptopropionate)	42378-34-7
15	1,1,3,3-tetramethyl-1,3-bis(oleoyloxy)distannoxane	43136-18-1
16	dimethyltin bis(laurylmercaptide)	51287-84-4
17	dimethyltin bis(2-ethylhexyl mercaptopropionate)	57057-50-8
18	dimethyltin bis(2-ethylhexyl mercaptoacetate)	57583-35-4
19	dimethyltin bis(2-oleoyloxyethylmercaptide)	67859-63-6
20	dimethyltin bis(2-linoleoyloxyethylmercaptide)	67859-64-7
21	dimethyltin bis(decyloxyloxyethylmercaptide)	67874-41-3
22	dimethyltin bis(lauroyloxyethylmercaptide)	68928-42-7
23	dimethyltin bis(neodecanoate)	68928-76-7
<u>Mono-n-butyltin compounds</u>		
24	n-butyltin trichloride	1118-46-3
25	n-butylstannic acid	2273-43-0
26	n-butylchlorotin dihydroxide	13355-96-9
27	n-butyltin sulfide	15666-29-2
28	n-butyltin tris(2-ethylhexanoate)	23850-94-4
29	n-butyltin tris(isooctyl mercaptoacetate)	25852-70-4
30	n-butylthiostannonic acid	26410-42-4
31	n-butyltin tris(2-ethylhexyl mercaptoacetate)	26864-37-9
32	n-butyltin tris(isooctyl mercaptopropionate)	36118-60-2
33	n-butyltin tris(2-oleoyloxyethylmercaptide)	67361-76-6
34	n-butyltin tris(2-decanoyloxyethylmercaptide)	67874-51-5
35	n-butyltin tris(2-tetradecanoyloxyethylmercaptide)	68928-34-7
36	n-butyltin tris(2-linoleoyloxyethylmercaptide)	68928-37-0
37	n-butyltin tris(2-lauroyloxyethylmercaptide)	68928-52-9
<u>Di-n-butyltin compounds</u>		
38	di-n-butyltin dilaurate	77-58-7
39	di-n-butyltin maleate	78-04-6
40	di-n-butyltin oxide	818-08-6
41	di-n-butyltin diacetate	1067-33-0

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Table 1 cont'd

No.	Chemical Name	CAS No.
42	di-n-butyltin bis(laurylmercaptide)	1185-81-5
43	di-n-butyltin bis(2-ethylhexanoate)	2781-10-4
44	di-n-butyltin sulfide	4253-22-9
45	di-n-butyltin distearate	5847-55-2
46	di-n-butyltin bis(methyl maleate)	15546-11-9
47	di-n-butyltin bis(2-ethylhexyl maleate)	15546-12-0
48	di-n-butyltin bis(n-butyl maleate)	15546-16-4
49	di-n-butyltin bis(acetylacetonate)	22673-19-4
50	di-n-butyltin bis(isooctyl maleate)	25168-21-2
51	di-n-butyltin bis(isooctyl mercaptoacetate)	25168-24-5
52	di-n-butyltin bis(isooctyl mercaptopropionate)	26761-46-6
53	di-n-butyltin bis(2-mercaptoethyl laurate)	28570-24-3
54	di-n-butyltin bis(lauryl maleate)	33466-31-8
55	di-n-butyltin bis(tetradecyl maleate)	60659-60-1
56	di-n-butyltin bis(2-oleoyloxyethylmercaptide)	67361-77-7
57	di-n-butyltin bis(2-linoleoyloxyethylmercaptide)	67859-61-4
58	di-n-butyltin bis(pentadecyl maleate)	68299-23-0
59	di-n-butyltin bis(2-decanoyloxyethylmercaptide)	68928-47-2
<u>Tri-n-butyltin compounds</u>		
60	tri-n-butyltin hydride	688-73-3
61	stannane, [(2-octyl-1,4-dioxo-1,4-butanediyl) bis(oxy)]bis[tributyl-	67701-37-5
<u>Mono-n-octyltin compounds</u>		
62	n-octyltin tris(isooctyl mercaptoacetate)	26401-86-5
63	n-octyltin tris(2-ethylhexyl mercaptoacetate)	27107-89-7
<u>Di-n-octyltin compounds</u>		
64	di-n-octyltin dilaurate	3648-18-8
65	di-n-octyltin bis(2-ethylhexyl mercaptoacetate)	15571-58-1
66	di-n-octyltin maleate	16091-18-2
67	di-n-octyltin bis(laurylmercaptide)	22205-30-7
68	di-n-octyltin bis(isooctylmercaptoacetate)	26401-97-8
69	di-n-octyltin bis(neodecanoate)	68299-15-0
(B) Pesticidal		
70	bis(tri-n-butyltin) oxide	56-35-9
71	tri-n-butyltin chloride	1461-22-9
72	tri-n-butyltin fluoride	1983-10-4
73	tri-n-butyltin methacrylate	2155-70-6
74	hexakis(β,β -dimethylphenylethyl)stannoxane	13356-08-6
(C) Tin salts of organic anions**		
75	stannous 2-ethylhexanoate	301-10-0
76	stannous oxalate	814-94-8
77	stannous n-octanoate	1912-83-0

*Canada Department of Environment (1990). CAS No. means Chemical Abstracts Service Registry number.

**These are not true organometallic compounds.

Table 2. Analytical methods for mono- and di- methyltin, n-butyltin and n-octyltin compounds.

Medium	Method	Minimum detectable concentration or amount	Comments	Reference
<u>MeSn³⁺</u>				
trichloride	GC-TCD of Bu ₃ MeSn			Neubert and Wirth (1975)
trichloride	HPLC-RID			Jessen <u>et al.</u> (1979)
trichloride	HPLC-RID	100000000 pg as Sn		Burns <u>et al.</u> (1980)
trichloride	HPLC-RID			Glockling (1980)
trichloride	HPLC separation, then hydride generation, then electrothermal AAS	1000 pg as chloride		Glockling (1980)
trichloride	HPLC with flame AAS	19000 pg as Sn		Burns <u>et al.</u> (1981)
trichloride	GC with quartz tube furnace AAS	1100 pg as Sn		Burns <u>et al.</u> (1981)
trichloride	HPLC with hydride generation, then quartz tube furnace AAS	8.2 pg as Sn		Burns <u>et al.</u> (1981)
trichloride	GC-FID of MeSnCl ₃	100000 pg		Tam <u>et al.</u> (1983)
trichloride	HPLC with fluorescence detection after complexation with 2',3',4',5',7-pentahydroxyflavone			Langseth (1984a)
trichloride	HPLC with UV epifluorescence detection	27 pg		Blair <u>et al.</u> (1987)
tris(isooctyl mercaptoacetate)	GC-TCD of Bu ₃ MeSn		no redistribution	Neubert and Wirth (1975)
water	hydride evolution, cryogenic trapping, detection with flame emission detector	1.6 pg as Sn		Braman and Tompkins (1979)

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Table 2 cont'd

Medium	Method	Minimum detectable concentration or amount	Comments	Reference
water	water saturated with NaCl, extracted with benzene/tropolone, analyzed by GC-AAS with quartz tube furnace	40 ng/L for 5 L sample		Chau <u>et al.</u> (1981)
water	extraction with dichloro-methane-tropolone or -diethyldithiocarbamate and injection into on-column hydride generation GC-AAS with quartz tube furnace			Clark <u>et al.</u> (1987), Clark and Craig (1988)
water	adsorption to C ₁₈ column, elution with tetrahydrofuran/acetic acid/tropolone, HPLC-AAS	30 ng Sn/L	long alumina tube AAS	Kadokami <u>et al.</u> (1988)
fresh water	quartz tube furnace AAS determination of MeSnH ₃	500 pg		Hodge <u>et al.</u> (1979)
fresh water	extract with pentane/diethyldithiocarbamate at pH 5, determined as MePe ₃ Sn by GC-AAS with quartz tube furnace	10 ng Sn/L		Dirkx <u>et al.</u> (1989)
estuarine water	quartz tube furnace AAS of MeSnH ₃	30 pg as Sn		Donard <u>et al.</u> (1986a,b)
seawater	hydride generation, then quartz tube AAS, GF-AAS or FPD	60, 60 and 15 pg, respectively		Andreae (1983), Andreae and Byrd (1984)
seawater	HPLC-hydride generation-DCP emission spectroscopy	11000 ng/L	suitable for clam and tuna tissue	Krull and Panaro (1985)
seawater	BuSnH ₃ determined by AAS with long alumina tube furnace	400 pg		Chamsaz <u>et al.</u> (1988)

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Table 2 cont'd

Medium	Method	Minimum detectable concentration or amount	Comments	Reference
sediment	hydride generation-GC-SIM-MS	0.001-0.005 ng Sn/g for 1-3 g sediment		Gilmour et al. (1986)
sediment	extracted with acidic methanol and determined as MeSnH ₃ by quartz tube furnace AAS	0.4 ng/g		Cooney et al. (1988)
algae	homogenized, then quartz tube furnace AAS determination of MeSnH ₃			Ishii (1982)
eelgrass leaf tissue	HCl extraction, hydride generation, then quartz tube furnace AAS	500 pg Sn		Francois and Weber (1988)
oysters	sonication in HCl then quartz tube furnace AAS of MeSnH ₃	1100 pg as Sn		Han and Weber (1988)

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Table 2' cont'd

Medium	Method	Minimum detectable concentration or amount	Comments	Reference
<u>Me₂Sn²⁺</u>				
dichloride	GC-TCD of Bu ₂ Me ₂ Sn			Neubert and Wirth (1975)
dichloride	HPLC-electrochemical detection			MacCrehan <u>et al.</u> (1977)
dichloride	HPLC-RID			Jessen <u>et al.</u> (1979)
dichloride	HPLC-RID	90000000 pg as Sn		Burns <u>et al.</u> (1980)
dichloride	GC-TCD	8250000 pg as Sn		Burns <u>et al.</u> (1980)
dichloride	HPLC-RID			Glockling (1980)
dichloride	HPLC separation, then hydride generation, then electrothermal AAS	1000 pg as chloride		Glockling (1980)
dichloride	field ionization and field desorption MS			Weber <u>et al.</u> (1980)
dichloride	HPLC separation, digestion then GF-AAS		discontinuous method since only part of effluent is sampled, giving nomographic presentation of results; Zr-treated furnaces reduced species-dependent sensitivity	Vickrey <u>et al.</u> (1980a)
dichloride	HPLC with flame AAS	17000 pg as Sn		Burns <u>et al.</u> (1981)
dichloride	GC with quartz tube furnace AAS	1000 pg as Sn		Burns <u>et al.</u> (1981)
dichloride	HPLC with hydride generation, then quartz tube furnace AAS	8.6 pg as Sn		Burns <u>et al.</u> (1981)
dichloride	GC with hydride generation, then quartz tube furnace AAS	1.5 pg as Sn		Burns <u>et al.</u> (1981)

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Table 2 cont'd

Medium	Method	Minimum detectable concentration or amount	Comments	Reference
dichloride	GC-FID of Me_2SnCl_2	100000 pg		Tam <u>et al.</u> (1983)
dichloride	HPLC with fluorescence detection after complexation with 2',3,4',5,7-pentahydroxyflavone			Langseth (1984a,b)
dichloride	HPLC with UV epifluorescence detection	>27 pg		Blair <u>et al.</u> (1987)
dichloride	HPLC with cyanopropyl-silica column deactivated with ICI			Praet <u>et al.</u> (1990)
bis(isooctyl mercaptoacetate)	GC-TCD of $\text{Bu}_2\text{Me}_2\text{Sn}$		no redistribution	Neubert and Wirth (1975)
dichloride in air	adsorption on glass fibre filter, desorption with hexane, hydride derivatization and GC-FID	20000 ng Sn/L solution		Vainiotalo and Hayri (1990)
water	hydride evolution, cryogenic trapping, detection with flame emission detector	0.7 pg as Sn		Braman and Tompkins (1979)
water	spectrophotometric determination of 3-hydroxyflavone complex			Aldridge and Street (1981)
water	water saturated with NaCl, extracted with benzene/tropolone, analyzed by GC-AAS with quartz tube furnace	40 ng/L for 5 L sample		Chau <u>et al.</u> (1981)
water	GC-FID of Me_2SnH_2 obtained by a combination of helium purging and diethyl ether extraction			Woolfins and Cullen (1984)

Table 2 cont'd

Medium	Method	Minimum detectable concentration or amount	Comments	Reference
water	adsorption to C_{18} column, elution with tetrahydrofuran/acetic acid/tropolone, HPLC-AAS	30 ng Sn/L	long alumina tube AAS	Kadokami <u>et al.</u> , (1988)
water	extraction with dichloromethane-tropolone or -diethyldithiocarbamate and injection into on-column hydride generation GC-AAS with quartz tube furnace			Clark <u>et al.</u> , (1987) Clark and Craig (1988)
water extracts of new PVC and CPVC pipe	Me_2H_2Sn generation, then quartz tube furnace AAS	500 pg	linear range 0.05-1 $\mu g/L$ for 100 ml sample	Boettner <u>et al.</u> , (1981)
fresh water	quartz tube furnace AAS determination of Me_2SnH_2	500 pg		Hodge <u>et al.</u> , (1979)
fresh water	extract with pentane/diethyldithiocarbamate at pH 5, determined as Me_2Pe_2Sn by GC-AAS with quartz tube furnace	7 ng Sn/L		Dirkx <u>et al.</u> , (1989)
estuarine water	quartz tube furnace AAS of Me_2SnH_2	30 pg as Sn		Donard <u>et al.</u> , (1986a,b)
seawater	hydride generation, then quartz tube AAS, GF-AAS or FPD	60, 60 and 15 pg, respectively		Andreae (1983), Andreae and Byrd (1984)
seawater	HPLC-hydride generation-DCP emission spectroscopy	12000 ng/L	suitable for clam and tuna tissue	Krull and Panaro (1985)
seawater	Me_2SnH_2 determined by AAS with long alumina tube furnace	500 pg		Chamsaz <u>et al.</u> , (1988)
sediment	hydride generation-GC-SIM-MS	0.001-0.005 ng Sn/g for 1-3 g sediment		Gilmour <u>et al.</u> , (1986)

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Table 2 cont'd

Medium	Method	Minimum detectable concentration or amount	Comments	Reference
sediment	extracted with acidic methanol and determined as Me_2SnH_2 by quartz tube furnace AAS	0.4 ng/g		Cooney et al. (1988)
algae	homogenized, then quartz tube furnace AAS determination of Me_2SnH_2			Ishii (1982)
eelgrass leaf tissue	HCl extraction, hydride generation, then quartz tube furnace AAS	500 pg Sn		Francois and Weber (1988)
oysters	sonication in HCl then quartz tube furnace AAS of Me_2SnH_2	1100 pg as Sn		Han and Weber (1988)
rat tissue	extraction with ethyl acetate, partitioning into hexane, HPLC with fluorescence detection of 2',3,4',5,7-pentahydroxy-flavone complex	100-1000 pg	>91% recovery with 750 ng spike	Yu and Arakawa (1983)
rat tissue	homogenate acidified, extracted with ethyl acetate, fluorescence determination with 2',3,4',5,7-pentahydroxyflavone	0.1 ng/mL ethyl acetate		Arakawa et al. (1983)
human urine	purge and trap GC-FPD of Me_2SnH_2			Olson et al. (1983)

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Table 2 cont'd

Medium	Method	Minimum detectable concentration or amount	Comments	Reference
<u>BuSn³⁺</u>				
tribromide	GC-TCD of BuMe ₃ Sn			Steimmeyer et al. (1965)
trichloride	ASV	40000 ng/L		Woggon et al. (1972)
trichloride	field ionization and field desorption MS			Weber et al. (1980)
trichloride	oxidation to Sn (IV), spectrophotometric determination of salicylideneamino-2-thiophenol complex			Imura and Suzuki (1983)
trichloride	substoichiometric isotope dilution analysis by complexation with salicylideneamino-2-thiophenol			Imura and Suzuki (1983)
trichloride	HPLC with fluorescence determination after complex formation with 2',3,4',5,7-pentahydroxyflavone			Langseth (1984a)
trichloride	BuSnH ₃ determination by GF-AAS, quartz tube furnace AAS or flame emission spectrometry	20-50 pg as Sn		Andreae and Byrd (1984)
trichloride	GC-WIPES of BuPe ₃ Sn	5 pg as Sn		Van Loon and Brzezinska-Paudyn (1986)
trichloride	GC-FID or GC-SIM-MS of BuH ₃ Sn or BuPe ₃ Sn		mass spectra reported	Dooley (1986)

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Table 2 cont'd

Medium	Method	Minimum detectable concentration or amount	Comments	Reference
trichloride	microbore or capillary HPLC with UV epifluorescence microscope determination of flavonol complex	157 pg		Blair <u>et al.</u> (1987)
trichloride	HPLC with UV epifluorescence detection	157 pg		Blair <u>et al.</u> (1987)
trichloride	HPLC-GF-AAS		enhanced sensitivity with added transition metal ions	Parks <u>et al.</u> (1988)
trichloride	direct injection into GC-FPD or DCP			Knull <u>et al.</u> (1989)
trihalides	GC-TCD of BuPr ₃ Sn			Jitsu <u>et al.</u> (1969)
trihalides	GC-FID or GC-FPD with carrier gas doping by HCl, HBr and HI			Aue <u>et al.</u> (1989)
water	extraction with tropolone in benzene or chloroform, then GC-MS of BuMe ₃ Sn		70-90% recovery at 1 ng/L; BuSn ₃ in water probably adsorbs to glass	Meinema <u>et al.</u> (1978)
water	simultaneous hydride formation/extraction, analysis of BuSnH ₃ by GC-FPD or -MS	22 ng Sn/L for 100 mL sample		Matthias <u>et al.</u> (1986b)
water	extraction with dichloro-methane-tropolone or -diethyldithiocarbamate and injection into on-column hydride generation GC-AAS with quartz tube furnace			Clark <u>et al.</u> (1987), Ashby <u>et al.</u> (1988) Clark and Craig (1988)

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Table 2 cont'd

Medium	Method	Minimum detectable concentration or amount	Comments	Reference
water	extracted with tropolone/benzene, determination of BuHx_3Sn by GC-SIM-MS or GC-FPD	<2 ng/L		Greaves and Unger (1988)
water	adsorption to C_{18} column, elution with tetrahydrofuran/acetic acid/tropolone, then HPLC-AAS	30 ng Sn/L	long alumina tube AAS	Kadokami et al. (1988)
water extracts of new PVC and CPVC pipe	BuH_3Sn generation, then quartz tube furnace AAS	500 pg	Linear range 0.05-1 $\mu\text{g/L}$ for 100 mL sample	Boettner et al. (1981)
water extracts of new PVC and CPVC pipe	extraction with tropolone in benzene, then GC-FID of BuPe_3Sn	1500-3000 ng/L (1 L sample)		Boettner et al. (1981)
fresh water	quartz tube furnace AAS determination of BuSnH_3	900 pg		Hodge et al. (1979)
fresh water	extraction with tropolone/benzene, then GC-FPD determination of BuPe_3Sn	100 pg	95-100% recovery at 10 ng/L	Maguire and Huneault (1981)
fresh water	see method above, but with quartz tube furnace GC-AAS	50-100 pg	AAS more stable than FPD	Maguire and Tkacz (1983)
fresh water	extracted as chloride with tropolone- C_{18} , determined as BuEt_3Sn by capillary GC-FPD	3 pg		Muller, M.D. (1987)
fresh water	extract with pentane/diethyldithiocarbamate at pH 5, determined as BuPe_3Sn by GC-AAS with quartz tube furnace	9 ng Sn/L		Dirkx et al. (1989)

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Table 2 cont'd

Medium	Method	Minimum detectable concentration or amount	Comments	Reference
estuarine water	purge and trap GC-FPD of BuSnH_3	20 ng Sn/L for 10 mL sample		Jackson <u>et al.</u> (1982)
estuarine water	quartz tube furnace AAS of BuSnH_3	50 pg as Sn		Donard <u>et al.</u> (1986a,b)
estuarine water	quartz tube furnace AAS of BuSnH_3	11 pg as Sn		Randall <u>et al.</u> (1986a)
estuarine water	extraction with hexane/tropolone, determination of BuHex_3Sn by capillary column GC-FPD or GC-MS-Cl		30% recovery at 3 ng Sn/L	Unger <u>et al.</u> (1986)
seawater	BuSnH_3 determination by quartz tube furnace AAS	5 ng/L	high hydrocarbon and sulfide concentrations interfere	Valkirs <u>et al.</u> (1985)
seawater	time-integrated column accumulator (C-18 silica)			Schatzberg <u>et al.</u> (1986)
seawater	BuSnH_3 determined by AAS with long alumina tube furnace	500 pg		Chamsaz <u>et al.</u> (1988)
seawater	purge and trap hydride generation - quartz tube furnace AAS		optimized procedure	Stallard <u>et al.</u> (1989)
seawater	purge and trap hydride generation - quartz tube furnace AAS		monobutyltin is stable in filtered water for 4 months at 4 °C; some losses at 25 °C	Quevaullier and Donard (1991)
sediment	dry sediment extracted with tropolone in benzene, then quartz tube furnace GC-AAS of BuPe_3Sn	3 ng Sn/g	quantitative recovery in range 3-300 ng Sn/g	Maguire (1984)

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Table 2 cont'd

Medium	Method	Minimum detectable concentration or amount	Comments	Reference
sediment	BuSnH ₃ generation from water-sediment slurries, then quartz tube furnace AAS			Valkirs <u>et al.</u> (1985)
sediment	extraction with HCl and dichloromethane-tropolone, transferred to ethanol, and injected into on-column hydride generation GC-AAS with quartz tube furnace			Clark <u>et al.</u> (1987), Clark and Craig (1988), Ashby <u>et al.</u> (1988)
sediment	extracted with acidic methanol and determined as BuSnH ₃ by quartz tube furnace AAS	0.4 ng/g		Cooney <u>et al.</u> (1988)
sediment	sonicated with methanol-HCl, extracted with toluene/isobutyl acetate/tropolone or hexane/isobutyl acetate, determined as chloride by GC-FPD	30 ng Sn/g for 1 g sample		Siu <u>et al.</u> (1989)
sediment	extraction with acetic acid, purge and trap hydride generation - quartz tube furnace AAS		some losses from sediment after freezing, freeze-drying or air drying	Quevauxiller and Donard (1991)
sediment and sewage sludge	extracted with tropolone in diethyl ether, determined as BuEt ₃ Sn by capillary GC-FPD	3 pg		Muller, M.D. (1987)
sediment and fish	acidify with HCl, extract with ethyl acetate, determined as BuSnCl ₃ by GC-ECD	10-20 ng/g		Tsuda <u>et al.</u> (1987)

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Table 2 cont'd

Medium	Method	Minimum detectable concentration or amount	Comments	Reference
sediment, biological tissues	extraction with cold acetic acid for 4 hr, then hydride generation and GC-AAS with quartz tube furnace		generally better than HCl or methanol	Desauziers <u>et al.</u> (1989)
sediment, mussel tissue	homogenized, acidified, extracted with dichloromethane, determination of BuHex ₃ Sn by GC-FPD		optimized procedure	Stallard <u>et al.</u> (1989)
bacterial, fungal cultures	extraction with tropolone in benzene, then GC-MS of BuMe ₃ Sn			Barug (1981)
algae	homogenized, then quartz tube furnace AAS determination of BuSnH ₃			Ishii (1982)
eelgrass leaf tissue	HCl extraction, hydride generation, then quartz tube furnace AAS	400 pg Sn		Francois and Weber (1988)
eelgrass	sonication in HCl then quartz tube furnace AAS of BuSnH ₃	400 pg as Sn	65% recovery at 600 ng/g	Francois <u>et al.</u> (1989)
oysters	sonication in HCl then quartz tube furnace AAS of BuSnH ₃	1100 pg as Sn		Han and Weber (1988)
oysters	extraction of homogenate with HCl and benzene/tropolone, determination of BuPe ₃ Sn by GC-FPD	5-9 pg		Ohhira and Matsui (1990)
rat tissues	homogenate acidified, extracted with ethyl acetate, HPLC determination after complexation with 2',3',4',5,7-pentahydroxyflavone	18 ng/mL extract	>92% recovery	Arakawa <u>et al.</u> (1983)

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Table 2 cont'd

Medium	Method	Minimum detectable concentration or amount	Comments	Reference
human urine	purge and trap GC-FPD of BusH ₅			Olson et al. (1983)
human urine	extraction with hexane, HPLC with fluorescence determination after complex formation with 2',3,4',5,7-pentahydroxyflavone	18 ng/mL extract	93% recovery with 2 µg spike	Arakawa et al. (1983)

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Table 2 cont'd

Medium	Method	Minimum detectable concentration or amount	Comments	Reference
<u>Bu₂Sn²⁺</u>				
dibromide	GC-TCD of Bu ₂ Me ₂ Sn			Steinmeyer et al. (1965)
dichloride	field ionization and field desorption MS			Weber et al. (1980)
dichloride	HPLC separation, digestion then GF-AAS		discontinuous method since only part of effluent is sampled, giving nonographic presentation of results; Zr-treated furnaces reduced species-dependent sensitivity	Vickrey et al. (1980a)
dichloride	GF-AAS	13-22 pg	Zr-treated furnaces reduced species-dependent sensitivity	Vickrey et al. (1980b)
dichloride	ion-exchange HPLC-GF-AAS	200000 ng/L		Jewett and Brinckman (1981)
dichloride	spectrophotometric - diphenylcarbazone complex	3000000 pg	can be determined in presence of mono-, tri- and tetrabutyltin	Skeel and Bricker (1961)
dichloride	substoichiometric isotope dilution analysis by complexation with salicylideneamino-2-thiophenol			Imura and Suzuki (1983)
dichloride	HPLC with fluorescence determination after complexation with 2',3,4',5,7-pentahydroxyflavone			Langseth (1984a,b)

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Table 2 cont'd

Medium	Method	Minimum detectable concentration or amount	Comments	Reference
dichloride	Bu ₂ SnH ₂ determination by GF-AAS, quartz tube furnace AAS or flame emission spectrometry	20-50 pg as Sn		Andreas and Byrd (1984)
dichloride	GF-AAS			Burns et al. (1984)
dichloride	GC-MIPES of Bu ₂ Pe ₂ Sn	5 pg as Sn		Van Loon and Brzezinska-Paudyn (1986)
dichloride	GC-FID or GC-SIM-MS of Bu ₂ H ₂ Sn or Bu ₂ Pe ₂ Sn		mass spectra reported	Dooley (1986)
dichloride	HPLC with UV epifluorescence detection	>157 pg		Blair et al. (1987)
dichloride	HPLC-GF-AAS (continuous)			Nygren et al. (1988)
dichloride	HPLC-GF-AAS		enhanced sensitivity with added transition metal ions	Parks et al. (1988)
dichloride	direct injection into GC-FPD or DCP			Krull et al. (1989)
dichloride	ion exchange HPLC with post-column derivatization with 2',3,4',5,7-pentahydroxyflavone and fluorescence determination	25000 pg Sn		Kleibohmer and Cammann (1989)
dichloride	HPLC with cyanopropyl-silica column deactivated with ICl			Praet et al. (1990)
dihalides	GC-ICD of Bu ₂ Pr ₂ Sn			Jitsu et al. (1969)
oxide	GF-AAS			Burns et al. (1984)
bis(isooctyl)-mercaptoacetate	cation exchange HPLC with GF-AAS			Gutknecht et al. (1982)

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Table 2 cont'd

Medium	Method	Minimum detectable concentration or amount	Comments	Reference
various compounds	spectrophotometric determination of catechol violet complex			Adamson (1962)
various compounds	IR		qualitative	Udris (1971)
various compounds	oxidation to Sn (IV), spectrophotometric determination of salicylideneamino-2-thiophenol complex			Imura and Suzuki (1983)
various compounds	GC-FID or GC-FPD with carrier gas doping by HCl, HBr and HI			Aue et al. (1989)
dichloride in air	adsorption on glass fibre, desorption with hexane, hydride derivatization and GC-FID	20 ng Sn/mL solution		Vainiotalo and Hayri (1990)
dichloride in distilled water	extraction with 2,6-diacetylpyridine bis(benzoylhydrazone) in hexane, then HPLC-UV			Careri et al. (1990)
dichloride and ditaurate in sediment and fish	acidification with HCl, extraction with hexane, determination of Bu_2SnH_2 by GC-ECD	0.5-1 ng/g	> 90% recovery at 1-10 μ g/20 g	Tsuda et al. (1986)
oxide in seawater	quartz tube furnace AAS of Bu_2SnH_2	2 ng Sn/L		Balls (1987a)
water	extracted with tropolone in benzene or chloroform, then GC-MS of Bu_2Me_2Sn		70-90% recovery at 1 ng/L	Heinema et al. (1978)
water	spectrophotometric determination of 3-hydroxyflavone complex			Aldridge and Street (1981)

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Table 2 cont'd

Medium	Method	Minimum detectable concentration or amount	Comments	Reference
water	extraction with benzene after addition of HCl and NaCl, then GC-ECD of Bu_2SnH_2	800 ng/L as chloride	74% recovery	Hattori <u>et al.</u> (1984)
water	GC-FID of Bu_2SnH_2 obtained by a combination of helium purging and diethyl ether extraction	2000 pg		Woolfins and Cullen (1984)
water	simultaneous hydride formation/extraction, analysis of Bu_2SnH_2 by GC-FPD or -MS	22 ng Sn/L for 100 mL sample		Matthias <u>et al.</u> (1986b)
water	extracted as chloride with tropolone- C_{10} , determined as $\text{Bu}_2\text{Et}_2\text{Sn}$ by capillary GC-FPD	3 pg		Muller, M.D. (1987)
water	extraction with dichloromethane/tropolone or diethyl dithiocarbamate and injection into on-column hydride generation GC-AAS with quartz tube furnace			Clark <u>et al.</u> (1987), Ashby <u>et al.</u> (1988)
water	adsorption to C_{18} column, elution with tetrahydrofuran/acetic acid/tropolone, HPLC-AAS	30 ng Sn/L	long alumina tube AAS	Kadokami <u>et al.</u> (1988)
water	extraction with pentane/diethyldithiocarbamate at pH 5, determination of $\text{Bu}_2\text{Pe}_2\text{Sn}$ by GC-AAS with quartz tube furnace	10 ng Sn/L		Dirkx <u>et al.</u> (1989)

Table 2 cont'd

Medium	Method	Minimum detectable concentration or amount	Comments	Reference
water extracts of new PVC and CPVC pipe	Bu ₂ H ₂ Sn generation, then quartz tube furnace AAS	500 pg	Linear range 50-1000 ng/L for 100 mL sample	Boettner et al. (1981)
water extracts of new PVC and CPVC pipe	extraction with tropolone in benzene, conversion to Bu ₂ Pe ₂ Sn, then GC-FID	1500-3000 ng/L (1 L sample)		Boettner et al. (1981)
fresh water	quartz tube furnace - AAS determination of Bu ₂ SnH ₂	1000 pg		Hodge et al. (1979)
fresh water	extraction with tropolone in benzene, conversion to Bu ₂ Pe ₂ Sn, then GC-FPD	100 pg	> 95% recovery from water at 10 ng/L	Maguire and Huneault (1981)
	see method above, but with quartz tube furnace GC-AAS	50-100 pg	AAS more stable than FPD	Maguire and Tkacz (1983)
estuarine water	purge and trap GC-FPD of Bu ₂ SnH ₂	40 ng Sn/L for 10 mL sample		Jackson et al. (1982)
estuarine water	quartz tube furnace AAS of Bu ₂ SnH ₂	50 pg as Sn		Donard et al. (1986a,b)
estuarine water	quartz tube furnace AAS of Bu ₂ SnH ₂	14 pg as Sn		Randall et al. (1986a)
estuarine water	extraction with hexane/tropolone, determination of Bu ₂ Hex ₂ Sn by capillary column GC-FPD or GC-MS-CI		60% recovery at 3 ng Sn/L	Unger et al. (1986)
estuarine water	extracted with tropolone/benzene, determination of Bu ₂ Hex ₂ Sn by GC-SIM-MS or GC-FPD	<2 ng/L		Greaves and Unger (1988)

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Table 2 cont'd

Medium	Method	Minimum detectable concentration or amount	Comments	Reference
estuarine and seawater	(1) simultaneous hydride generation-dichloromethane extraction, then GC-FPD, or (2) hydride generation, purge and trap collection, then quartz tube furnace AAS		good agreement between the two methods	Valkirs <u>et al.</u> (1987)
seawater	extraction with carbon tetrachloride, oxidation to Sn^{4+} , spectrophotometric determination of phenylfluorone complex	7000 ng Sn/L		Mor <u>et al.</u> (1973)
seawater	Bu_2SnH_2 determination by quartz tube furnace AAS	5 ng/L	high hydrocarbon and sulfide concentrations interfere	Valkirs <u>et al.</u> (1985)
seawater	time-integrated column accumulator (C-18 silica)			Schatzberg <u>et al.</u> (1986)
seawater	Bu_2SnH_2 determined by AAS with long alumina tube furnace	500 pg		Chamsaz <u>et al.</u> (1988)
seawater	simultaneous hydride generation/extraction then GF-AAS	60 pg		Chamsaz and Winefordner (1987)
seawater	purge and trap hydride generation - quartz tube furnace AAS		optimized procedure	Stallard <u>et al.</u> (1989)
seawater	simultaneous extraction/hydride formation, cleanup and GC-FPD	100 pg (25 ng/L for 400 mL)		Nagase (1990)
seawater	purge and trap hydride generation - quartz tube furnace AAS		dibutyltin is stable in filtered water for 4 months at 4 °C; some losses at 25 °C	Quevauviller and Donard (1991)

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Table 2 cont'd

Medium	Method	Minimum detectable concentration or amount	Comments	Reference
sediment	dry sediment extracted with tropolone in benzene, then quartz tube furnace GC-AAS of $BuPe_3Sn$	2.5 ng Sn/g	quantitative recovery in range 3-300 ng Sn/g	Maguire (1984)
sediment	extraction with methanol/hydrochloric acid, transfer to benzene, then GC-ECD of Bu_2SnH_2	40 ng/g as chloride	76% recovery	Hattori et al. (1984)
sediment	Bu_2SnH_2 generation from water-sediment slurries, then quartz tube furnace AAS			Valkirs et al. (1985)
sediment	extraction with hydrochloric acid and dichloromethane/tropolone, transfer to ethanol, and injection into on-column hydride generation GC-AAS with quartz tube furnace			Clark et al. (1987), Clark and Craig (1988), Ashby et al. (1988)
sediment	extracted with acidic methanol, determined as Bu_2SnH_2 by quartz tube furnace AAS	0.4 ng/g		Cooney et al. (1988)
sediment	sonicated with methanol-HCl, extracted with toluene/isobutyl acetate/tropolone or hexane/isobutyl acetate, determined as chloride by GC-FPD	30 ng Sn/g for 1 g sample		Siu et al. (1989)
sediment	extraction with acetic acid, purge and trap hydride generation - quartz tube furnace AAS		some losses from sediment after freezing, freeze-drying or air drying	Quevaliller and Dorand (1991)

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Table 2 cont'd

Medium	Method	Minimum detectable concentration or amount	Comments	Reference
sediment and microbial biofilms	refluxed with acidified methanol, then extraction into cyclohexane, and GC-FPD of Bu_2Sn_2			Matthias <i>et al.</i> (1989)
sediment and fish	acidify with hydrochloric acid, extract with ethyl acetate, determine Bu_2SnH_2 by GC-ECD	0.5-1 ng/g		Tsuda <i>et al.</i> (1987)
sediment and sewage sludge	extracted with tropolone in diethyl ether, determined as $\text{Et}_2\text{Bu}_2\text{Sn}$ by capillary GC-FPD	3 pg		Muller, M.D. (1987)
sediment, biological tissues	extraction with cold acetic acid for 4 hr, then hydride generation and GC-AAS with quartz tube furnace		generally better than HCl or methanol	Desauziers <i>et al.</i> (1989)
sediment, mussel tissue	homogenization, acidification, extraction with dichloromethane, determination of $\text{Bu}_2\text{Hex}_2\text{Sn}$ by GC-FPD		optimized procedure	Stallard <i>et al.</i> (1989)
soil	extraction with wet diethyl ether/formic acid, TLC with scintillation counter		70% recovery	Barug and Vonk (1980)
bacterial and fungal cultures	extraction with tropolone in benzene, then GC-MS of $\text{Bu}_2\text{Me}_2\text{Sn}$			Barug (1981)
eelgrass leaf tissue	HCl extraction, hydride generation, then quartz tube furnace AAS	500 pg Sn		Francois and Weber (1988)
eelgrass	sonication in HCl then quartz tube furnace AAS of Bu_2SnH_2	500 pg as Sn	63% recovery at 700 ng/g	Francois <i>et al.</i> (1989)

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Table 2 cont'd

Medium	Method	Minimum detectable concentration or amount	Comments	Reference
fish	tissue dispersed in HCl, extracted with hexane/tropolone, $\text{Bu}_2\text{Sn}^{2+}$ separated from Bu_3Sn^+ with alkali wash, determination by GF-AAS			Ward et al. (1981)
dichloride and dilaurate in fish tissue	acidification with HCl, extraction with with 3/2 ethyl acetate/hexane, determination of Bu_2SnH_2 by GC-ECD	1-2 ng/g	75-98% recovery at 1-10 $\mu\text{g}/10\text{ g}$	Tsuda et al. (1986)
fish	extraction with HCl/methanol transferred to hexane, then GC-FPD of $\text{Bu}_2\text{Me}_2\text{Sn}$	20 ng/g	>90% recovery at 100-2000 ng/g	Sasaki et al. (1988a)
fish	extracted with methanol/dichloromethane, determination of Bu_2SnH_2 by reaction GC-FPD	100 pg as Sn		Sullivan et al. (1988)
yellowtail	homogenization/extraction NaCl, HCl and diethyl ether/hexane, determination of $\text{Bu}_2\text{Et}_2\text{Sn}$ by GC-FPD	5 ng/g	>90% recovery	Sasaki et al. (1988b)
fish	homogenization/extraction with NaCl, HCl and diethyl ether/hexane, cleanup on Sep-Pak Florisil, determination of $\text{Bu}_2\text{Et}_2\text{Sn}$ by GC-FPD	200 pg		Ishizaka et al. (1989a)
oysters	sonication in HCl, then quartz tube furnace AAS determination of Bu_2SnH_2	1100 pg as Sn		Han and Weber (1988)
oysters	extraction from homogenate with HCl and benzene/tropolone, determination of $\text{Bu}_2\text{Pe}_2\text{Sn}$ by GC-FPD	5-9 pg		Ohhira and Matsui (1990)

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Table 2 cont'd

Medium	Method	Minimum detectable concentration or amount	Comments	Reference
shellfish	dichloromethane extraction, then GC-FID of $\text{Bu}_2\text{Hex}_2\text{Sn}$			Page (1989)
rat tissue	extraction with ethyl acetate, partitioning into hexane, then HPLC with fluorescence detection of 2',3,4',5,7-pentahydroxyflavone complex	100-1000 pg	98% recovery at 100 ng spiked	Yu and Arakawa (1983)
rat tissue	homogenate acidified, extracted with ethyl acetate, fluorescence determination after complexation with 2',3,4',5,7-pentahydroxyflavone	120 ng/mL ethyl acetate	>98% recovery	Arakawa et al. (1983)
human urine	extraction with hexane, fluorescence determination after complex formation with 2',3,4',5,7-pentahydroxyflavone	120 ng/mL extract	99% recovery at 1 nmol	Arakawa et al. (1983)
dilaurate in poultry feeds	extracted with chloroform, transferred to methanol, then flame AAS	10000 ng/g		George et al. (1973)
fishing nets treated with Bu_3Sn^+	extraction with ethanol, then DPPD	14000 ng/L extract		Hasebe et al. (1982)
textiles	extraction with methanol/5% hydrochloric acid, diluted with water, extracted with 4-(2-pyridylazo)resorcinol in dichloromethane, oxidized to Sn (IV), then GF-AAS			Kojima (1979)

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Table 2 cont'd

Medium	Method	Minimum detectable concentration or amount	Comments	Reference
<u>OctSn³⁺</u>				
trichloride	ASV	40000 ng/L		Woggon et al. (1972)
trichloride	oxidation to Sn (IV), spectrophotometric determination of salicylideneamino-2-thiophenol complex			Imura and Suzuki (1983)
trialkylides	GC-TCD of OctPe ₃ Sn			Jitsu et al. (1969)
tris(2-ethylhexyl mercaptoacetate)	GC-TCD of OctEt ₃ Sn		no redistribution	Figge et al. (1977)
stannonic acid and tris(isooctyl-mercaptoacetate)	GC-TCD of OctMe ₃ Sn		no redistribution	Neubert and Wirth (1975)
water	extraction with diethyl ether at pH 2, then GC-SIM-MS of OctBu ₃ Sn	5000 ng/L		Ruthenberg and Madetzki (1988)

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Table 2 cont'd

Medium	Method	Minimum detectable concentration or amount	Comments	Reference
<u>Oct₂Sn²⁺</u>				
dichloride	oxidation to Sn (IV), spectrophotometric determination of salicylidenamino-2-thiophenol complex			Imura and Suzuki (1983)
dihalides	GC-TCD of Oct ₂ Pe ₂ Sn			Jitsu et al. (1969)
bis-maleate	GF-AAS	10000 ng/L		Varnes and Gaylor (1978)
bis(2-ethylhexyl mercaptoacetate)	GC-TCD of Oct ₂ Et ₂ Sn		no redistribution	Figge et al. (1977)
oxide and bis(isooctylmercaptoacetate)	GC-TCD of Oct ₂ Me ₂ Sn		no redistribution	Neubert and Wirth (1975)
various compounds	IR		qualitative	Udriš (1971)
various compounds	spectrophotometric determination of catechol violet complex			Adamson (1962)
dichloride in air	adsorption on glass fibre, desorption with hexane, hydride derivatization and GC-FID	20 ng Sn/mL solution		Vainiotalo and Hayri (1990)
maleate and isooctyl-mercaptoacetate in PVC films	extraction with various solvents, oxidation to Sn (IV), GF-AAS	100 pg		Trachman et al. (1977)
water	extraction with diethyl ether at pH 2, then GC-SIM-MS of Oct ₂ Bu ₂ Sn	50000 ng/L		Ruthenberg and Madetzki (1988)

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Table 2 cont'd

Medium	Method	Minimum detectable concentration or amount	Comments	Reference
rat tissue	extraction with ethyl acetate, partitioning into hexane, HPLC with fluorescence detection of 2',3,4',5,7-pentahydroxyflavone complex	100-1000 pg	>96% recovery with 2 µg spike	Yu and Arakawa (1983)

*All concentrations in sediment and sludge are by dry weight, and in biota are by wet weight, unless otherwise noted. Concentrations are reported in various ways by the original authors: (i) as the cation (e.g., Bu_4Sn^{2+}), (ii) unspecified, hence reported as the cation, (iii) as the chloride, and (preferably) (iv) in terms of Sn (e.g., ng Sn/L). Reporting concentrations in terms of Sn is preferable because it allows easy calculations of mass balances of homologous species and comparisons of toxicity. Note however that the errors involved in comparing different units reported here are quite small compared to experimental errors in determining concentrations of organotin species in environmental media.

Abbreviations used:

AAS	atomic absorption spectrophotometry
ASV	anodic stripping voltammetry
Bu	n-butyl
Cl	chemical ionization
CPVC	chlorinated poly(vinyl chloride)
DCP	direct current plasma
DPPD	differential pulse polarographic detector
ECD	electron capture detector
Et	ethyl
FID	flame ionization detector
FPD	flame photometric detector
GC	gas chromatography
GF	graphite furnace
Hex	n-hexyl
HPLC	high performance liquid chromatography
Me	methyl
MIPES	microwave induced plasma emission spectrometry
MS	mass spectrometry
Oct	n-octyl
Pe	n-pentyl
Pr	n-propyl
PVC	poly(vinyl chloride)
RID	refractive index detector
SIM	selected ion monitoring
TCD	thermal conductivity detector
TLC	thin layer chromatography
UV	ultraviolet

Table 3. Environmental occurrence of mono- and di- methyltin and n-butyltin species*

Compound	Medium	Concentration	Location	Reference
<u>MeSn³⁺</u>	tap water	0.5-8.1 ng Sn/L	Florida	Braman and Tompkins (1979)
	rain water	0.7-22 ng Sn/L	Florida	Braman and Tompkins (1979)
	rain water	n.d.-10 ng Sn/L	La Jolla, CA	Tugrul <u>et al.</u> (1983)
	fresh water	6-18 ng/L as chloride	Lake Michigan	Hodge <u>et al.</u> (1979)
	fresh water	n.d.-12 ng/L	Florida	Braman and Tompkins (1979)
	fresh water	400-1200 ng/L	Ontario harbours and marinas	Chau <u>et al.</u> (1982)
	fresh water	100-1200 ng/L	Ontario lakes, rivers and harbours	Maguire <u>et al.</u> (1982)
	fresh water	n.d.-79 ng/L	rivers in USA and Germany	Byrd and Andreae (1982)
	fresh water	n.d.-100 ng Sn/L	Canada	Maguire <u>et al.</u> (1986)
	fresh water	8.1 ng/L	harbour in Antwerp, Belgium	Dirkx <u>et al.</u> (1989)
	fresh and seawater	n.d.-21 ng/L	Turkey	Yemenicioglu <u>et al.</u> (1987)
	estuarine water	n.d.-8 ng Sn/L	Florida	Braman and Tompkins (1979)
	estuarine water	29-63 ng/L	Great Bay Estuary, NH, USA	Donard <u>et al.</u> (1986a)
	estuarine water surface microlayer	182-964 ng/L	Great Bay Estuary, NH, USA	Donard <u>et al.</u> (1986a)
	estuarine sediment pore water	690 ng/L	Great Bay Estuary, NH, USA	Donard <u>et al.</u> (1986a)
	salt marsh sediment pore water	320 ng/L	Great Bay Estuary, NH, USA	Donard <u>et al.</u> (1986a)
	seawater	n.d.-8 ng/L as chloride	California	Hodge <u>et al.</u> (1979)
	seawater	n.d.-15 ng Sn/L	Florida	Braman and Tompkins (1979)
	seawater	n.d.-42 ng Sn/L	USA and Turkey	Tugrul <u>et al.</u> (1983)
	seawater	69 ng/L	Great Bay Estuary, NH, USA	Donard <u>et al.</u> (1986a)

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Table 3 cont'd

Compound	Medium	Concentration	Location	Reference
	sediment	n.d.-10.6 ng Sn/g	USA, Turkey, Japan	Tugrul <u>et al.</u> (1983)
	sediment	n.d.-17000 ng Sn/g	Canada	Maguire <u>et al.</u> (1986)
	sediment	220 ng/g	Great Bay Estuary, NH, USA	Donard <u>et al.</u> (1986b)
	sediment	20-350 ng/g	two Swiss lakes	Donard <u>et al.</u> (1986b)
	sediment	n.d.-80 ng Sn/g	Great Bay Estuary, NH, USA	Randall <u>et al.</u> (1986b)
	sediment	n.d.-0.8 ng/g	Turkey	Yemenicioglu <u>et al.</u> (1987)
	sediment	n.d.-21 ng/g	Portugese rivers, estuaries and sea water	Quevauviller <u>et al.</u> (1989)
	sediment	639-1049 ng Sn/g	Lake Maryut, Egypt	Aboul Dahab <u>et al.</u> (1990)
	sediment	n.d.-20 ng Sn/g	Rhine River, Germany	Schebek <u>et al.</u> (1991)
	sediment	n.d.-45 ng Sn/g	Germany	Schebek <u>et al.</u> (1991)
	algae	n.d.-1.2 ng/g	Mission and San Diego Bays, CA	Ishii (1982)
	seaweed	30-71 ng/g	Great Bay Estuary, NH, USA	Donard <u>et al.</u> (1987)
	eelgrass	n.d.-2300 ng Sn/g	Great Bay Estuary, NH, USA	Francois and Weber (1988)
	mussels	n.d.-7 ng/g	Portugese estuaries and harbours	Quevauviller <u>et al.</u> (1989)
	oysters	95-120 ng Sn/g	Great Bay, NH, USA	Han and Weber (1988)
	limpets	n.d.-13.9 ng/g	Turkey	Yemenicioglu <u>et al.</u> (1987)
	smelt (single fish) (<u>O. mordax</u>)	400 ng Sn/g	Lake Ontario (Cobourg)	Maguire <u>et al.</u> (1986)
	lake trout (<u>S. namaycush</u>)	n.d.-880 ng Sn/g	Lake Ontario, Canada	Maguire <u>et al.</u> (1986)
	various marine organisms	n.d.-27 ng Sn/g dry wt.	Mediterranean Sea (Turkey)	Tugrul <u>et al.</u> (1983)
	shells (various organisms)	n.d.-0.5 ng Sn/g	Florida	Braman and Tompkins (1979)
	human urine	n.d.-320 ng Sn/L	Florida	Braman and Tompkins (1979)

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Table 3 cont'd

Compound	Medium	Concentration	Location	Reference
<u>Me₂Sn²⁺</u>	tap water	0.4-2.2 ng Sn/L	Florida	Braman and Tompkins (1979)
	organotin production plant effluent	470 ng Sn/L	Germany	Schebek <u>et al.</u> (1991)
	STP effluent (domestic effluent only)	0.8-2 ng Sn/L	Germany	Schebek <u>et al.</u> (1991)
	STP effluent (40% domestic - 60% industrial effluent)	0.3-12 ng Sn/L	Germany	Schebek <u>et al.</u> (1991)
	rain water	n.d.-4.8 ng Sn/L	Florida	Braman and Tompkins (1979)
	fresh water	n.d.-63 ng/L as chloride	Lake Michigan	Hodge <u>et al.</u> (1979)
	fresh water	n.d.-7.5 ng/L	Florida	Braman and Tompkins (1979)
	fresh water	n.d.-400 ng/L	Ontario lakes, rivers and harbours	Maguire <u>et al.</u> (1982)
	fresh water	n.d.-264 ng/L	rivers in USA and Germany	Byrd and Andreea (1982)
	fresh water	6.0 ng/L	harbour in Antwerp, Belgium	Dirkx <u>et al.</u> (1989)
	fresh water	n.d.-0.6 ng Sn/L	Rhine River, Germany	Schebek <u>et al.</u> (1991)
	fresh and seawater	n.d.-26 ng/L	Turkey	Yemenicioglu <u>et al.</u> (1987)
	estuarine water	0.8-4.6 ng Sn/L	Florida	Braman and Tompkins (1979)
	estuarine water	n.d.-0.8 ng Sn/L	Ochlocknee River, Florida	Andreea (1983)
	estuarine water	n.d.-50 ng/L	Great Bay Estuary, NH, USA	Donard <u>et al.</u> (1986a)
	estuarine water surface microlayer	196 ng/L	Great Bay Estuary, NH, USA	Donard <u>et al.</u> (1986a)
	estuarine sediment pore water	20 ng/L	Great Bay Estuary, NH, USA	Donard <u>et al.</u> (1986a)
	salt marsh sediment pore water	58 ng/L	Great Bay Estuary, NH, USA	Donard <u>et al.</u> (1986a)
	seawater	n.d.-45 ng/L as chloride	California	Hodge <u>et al.</u> (1979)

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Table 3 cont'd

Compound	Medium	Concentration	Location	Reference
	seawater	n.d.-7 ng Sn/L	Florida	Braman and Tompkins (1979)
	seawater	n.d.-45 ng Sn/L	USA and Turkey	Tugrul <u>et al.</u> (1983)
	seawater	31 ng/L	Great Bay Estuary, NH, USA	Donard <u>et al.</u> (1986a)
	sediment	n.d.-13.4 ng Sn/g	USA, Turkey, Japan	Tugrul <u>et al.</u> (1983)
	sediment	n.d.-200 ng Sn/g	Canada	Maguire <u>et al.</u> (1986)
	sediment	n.d.-49 ng Sn/g	Great Bay Estuary, NH, USA	Randall <u>et al.</u> (1986b)
	sediment	20-40 ng/g	two Swiss lakes	Donard <u>et al.</u> (1986b)
	sediment	50 ng/g	Great Bay Estuary, NH, USA	Donard <u>et al.</u> (1986b)
	sediment	n.d.-4.5 ng/g	Turkey	Yemenicioglu <u>et al.</u> (1987)
	sediment	n.d.-160 ng/g	Portugese rivers, estuaries and sea water	Quevauviller <u>et al.</u> (1989)
	sediment	154-270 ng Sn/g	Lake Maryut, Egypt	About Dahab <u>et al.</u> (1990)
	sediment	n.d.-10 ng Sn/g	Rhine River, Germany	Schebek <u>et al.</u> (1991)
	sediment	n.d.-82 ng Sn/g	Germany	Schebek <u>et al.</u> (1991)
	algae	n.d.-2.2 ng/g as chloride	San Diego Bay	Seidel <u>et al.</u> (1980)
	algae	n.d.-2.2 ng/g	Mission and San Diego Bays, CA	Ishii (1982)
	seaweed	13-52 ng/g	Great Bay Estuary, NH, USA	Donard <u>et al.</u> (1987)
	mussels	n.d.-10 ng/g dry wt.	Portugese estuaries and harbours	Quevauviller <u>et al.</u> (1989)
	limpets	n.d.-6.7 ng/g dry wt.	Turkey	Yemenicioglu <u>et al.</u> (1987)
	lake trout (<u>S. namaycush</u>) (single fish)	180 ng Sn/g	Lake Ontario (Cobourg), Canada	Maguire <u>et al.</u> (1986)
	various marine organisms	n.d.-37 ng Sn/g dry wt.	Mediterranean Sea (Turkey)	Tugrul <u>et al.</u> (1983)
	shells (various organisms)	n.d.-0.2 ng Sn/g	Florida	Braman and Tompkins (1979)

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Table 3 cont'd

Compound	Medium	Concentration	Location	Reference
	human urine	n.d.-170 ng Sn/L	Florida	Braman and Tompkins (1979)

Table 3 cont'd

Compound	Medium	Concentration	Location	Reference
<u>BuSn³⁺</u>				
	dry dock wash water	n.d.-120000 ng/L	England	Waldock <u>et al.</u> (1988)
	landfill leachate	112 ng/L	Switzerland	Fent (1991)
	organotin production plant effluent	2130 ng Sn/L	Germany	Schebek <u>et al.</u> (1991)
	untreated wastewater	136-554 ng/L	Zurich, Switzerland	Fent and Muller (1991)
	STP influent	142-250 ng/L	Zurich, Switzerland	Fent (1989, 1991)
	STP influent	1900-20600 ng Sn/L	five Canadian cities	Chau <u>et al.</u> (1991)
	STP effluent	20 ng/L as chloride	Switzerland	Muller, M.D. (1987)
	STP effluent	700-14500 ng Sn/L	five Canadian cities	Chau <u>et al.</u> (1991)
	tertiary STP effluent	4-17 ng/L	Zurich, Switzerland	Fent and Muller (1991)
	STP effluent (domestic effluent only)	9 ng Sn/L	Germany	Schebek <u>et al.</u> (1991)
	STP effluent (40% domestic - 60% industrial effluent)	9-19 ng Sn/L	Germany	Schebek <u>et al.</u> (1991)
	sewage sludge	200-6000 ng/g as chloride	Switzerland	Muller, M.D. (1987)
	sewage sludge	1200 ng/g	Zurich, Switzerland	Fent (1989, 1991)
	sewage sludge	100-970 ng/g	Zurich, Switzerland	Fent and Muller (1991)
	sewage sludge	n.d.-440 ng Sn/g	five Canadian cities	Chau <u>et al.</u> (1991)
	fresh water	20-1200 ng/L (as chloride)	Lake Michigan (possible contamination by PVC-containing samplers)	Hodge <u>et al.</u> (1979)
	fresh water	100-400 ng/L	Ontario harbours and marinas	Maguire <u>et al.</u> (1982)
	fresh water	n.d.-100 ng Sn/L	Toronto Harbour	Maguire and Tkacz (1985)
	fresh water	n.d.-100 ng Sn/L	Detroit and St. Clair Rivers, Canada	Maguire <u>et al.</u> (1985a)

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Table 3 cont'd

Compound	Medium	Concentration	Location	Reference
	fresh water	n.d.-1900 ng Sn/L	Canada	Maguire <u>et al.</u> (1986)
	fresh water	12-50 ng/L as chloride	Switzerland	Muller, M.D. (1987)
	fresh water	n.d.-113 ng/L	Rivers Bure and Yare, England	Waite <u>et al.</u> (1989)
	fresh water	n.d.-13 ng/L	California marinas	Stang and Goldberg (1989)
	fresh water	14-1200 ng/L	harbour in Antwerp, Belgium	Dirkx <u>et al.</u> (1989)
	fresh water	0.7-2.3 ng Sn/L	Rhine River, Germany	Schebek <u>et al.</u> (1991)
	fresh water	n.d.-75 ng/L	marinas in Lake Lucerne, Switzerland	Fent and Hurn (1991)
	fresh water	0.9-6 ng Sn/L	Germany	Schebek <u>et al.</u> (1991)
	fresh and estuarine water	<1-13 ng Sn/L	tributary of Chesapeake Bay	Unger <u>et al.</u> (1986)
	fresh water surface microlayer	n.d.-90 ng Sn/L	St. Clair River, Canada	Maguire <u>et al.</u> (1985a)
	fresh water surface microlayer	n.d.-36 ng Sn/L	Ontario Lakes and rivers	Maguire and Tkacz (1987)
	estuarine water	50-300 ng Sn/L	Baltimore Harbor	Jackson <u>et al.</u> (1982)
	estuarine water	<0.02-1.2 ng Sn/L	Tejo River estuary, Portugal	Andreea <u>et al.</u> (1983)
	estuarine water	n.d.-1.8 ng/L	USA, England and Portugal	Byrd and Andreea (1986)
	estuarine water	n.d.-22 ng/L	Great Bay Estuary, NH, USA	Donard <u>et al.</u> (1986a)
	estuarine water	<20-736 ng/L	Chesapeake Bay	Matthias <u>et al.</u> (1986a)
	estuarine water	n.d.-193 ng/L	England	Waldock <u>et al.</u> (1987)
	estuarine water	n.d.-102 ng/L	Chesapeake Bay marina	Matthias <u>et al.</u> (1988)
	estuarine water surface microlayer	42-212 ng/L	Great Bay Estuary, NH, USA	Donard <u>et al.</u> (1986a)
	estuarine water surface microlayer	165 ng Sn/L	Baltimore Harbor	Matthias <u>et al.</u> (1986b)

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Table 3 cont'd

Compound	Medium	Concentration	Location	Reference
	estuarine water surface microlayer	n.d.-1030 ng/L	Chesapeake Bay marina	Matthias <u>et al.</u> (1988)
	seawater	10-60 ng/L	San Diego Bay	Valkirs <u>et al.</u> (1985)
	seawater	0.32-0.82 ng Sn/L	Esquimalt Harbour, Canada	Kaye <u>et al.</u> (1986)
	seawater	n.d.-120 ng/L	San Diego Bay	Valkirs <u>et al.</u> (1986)
	seawater	n.d.-230 ng/L	California coast	Stallard <u>et al.</u> (1987)
	seawater	<1-9 ng/L	English harbours and anchorages	Waldock <u>et al.</u> (1988)
	seawater	<0.001-170 ng Sn/L	Esquimalt Harbour, Canada	Seakem (1989)
	seawater	n.d.-230 ng/L as chloride	USA harbours	Grovhog <u>et al.</u> (1989)
	seawater	<1-200 ng/L	France	Alzieu <u>et al.</u> (1989)
	seawater	n.d.-94 ng/L	European coastal areas	Quevauller and Donard (1990)
	seawater	n.d.-2774 ng/L	Mediterranean Sea coastal areas	Gabrielides <u>et al.</u> (1990)
	seawater	n.d.-19 ng Sn/L	British Columbia	Cullen <u>et al.</u> (1990)
	seawater	n.d.-109 ng/L	various locations in French Mediterranean Sea	Alzieu <u>et al.</u> (1990)
	seawater	not quantitated	Barcelona Harbour, Spain	Tolosa <u>et al.</u> (1991)
	sediment	n.d.-12 ng/g as chloride	California coast	Seidel <u>et al.</u> (1980)
	sediment	n.d.-58 ng/g	Ontario lakes and rivers	Maguire (1984)
	sediment	n.d.-100 ng/g	Toronto Harbour	Maguire and Tkacz (1985)
	sediment	n.d.-35 ng Sn/g	Detroit and St. Clair Rivers, Canada	Maguire <u>et al.</u> (1985a)
	sediment	7-45 ng/g wet wt.	San Diego Bay	Valkirs <u>et al.</u> (1985)
	sediment	n.d.-4700 ng/g	Canada	Maguire <u>et al.</u> (1986)
	sediment	3.0-30.0 ng/g	Great Bay Estuary, NH, U.S.A.	Randall <u>et al.</u> (1986b)

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Table 3 cont'd

Compound	Medium	Concentration	Location	Reference
	sediment	20-620 ng/g	Esquimalt Harbour, Canada	Kaye <u>et al.</u> (1986)
	sediment	n.d.-60 ng/g	California coast	Stallard <u>et al.</u> (1987)
	sediment	n.d.-450 ng Sn/g	Alexandria, Egypt	Aboul Dahab (1988)
	sediment	n.d.-2100 ng/g	Portuguese rivers, estuaries and sea water	Quevauviller <u>et al.</u> (1989)
	sediment	n.d.-53 ng/g wet wt.	Puget Sound, WA, USA	Krone <u>et al.</u> (1989b)
	sediment	n.d.-126 ng/g	Boston Harbor	Makkar <u>et al.</u> (1989)
	sediment	<0.6-150 ng Sn/g	marinas and waterways in Washington, U.S.A.	Krone <u>et al.</u> (1989a)
	sediment	<2-1060 ng/g	U.S.A. coastal harbours	Kram <u>et al.</u> (1989a)
	sediment	5-35 ng Sn/g	Ise Bay, Japan	Yonezawa <u>et al.</u> (1989)
	sediment	n.d.-30 ng Sn/g	Oshawa and Whitby Harbours, Ontario	Chau <u>et al.</u> (1989)
	sediment	23 ng Sn/g	Porto Vecchio Bay, Corsica, France	Astruc <u>et al.</u> (1989)
	sediment	100-6800 ng/g	France	Desauziers <u>et al.</u> (1989)
	sediment	n.d.-533 ng/g	San Diego Bay and Pearl Harbor	Kram <u>et al.</u> (1989b)
	sediment	n.d.-480 ng/g as chloride	USA harbours	Grovhoug <u>et al.</u> (1989)
	sediment (spring)	n.d.-92 ng/g	Severn Sound, Ontario	Wong and Chau (1990)
	sediment (summer)	n.d.-147 ng/g	Severn Sound, Ontario	Wong and Chau (1990)
	sediment (summer)	n.d.-62 ng/g	Severn Sound, Ontario	Wong and Chau (1990)
	sediment	36-624 ng Sn/g	Lake Maryut, Egypt	Aboul Dahab <u>et al.</u> (1990)
	sediment	<5-32 ng Sn/g	U.S.A. coastal areas	Wade <u>et al.</u> (1990)
	sediment	n.d.-60 ng Sn/g	Rhine River, Germany	Schebek <u>et al.</u> (1991)
	sediment	27-34 ng Sn/g	Germany	Schebek <u>et al.</u> (1991)

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Table 3 cont'd

Compound	Medium	Concentration	Location	Reference
	sediment	9-148 ng Sn/g	harbours in Lake Erie	Scott <u>et al.</u> (1991)
	sediment	not quantitated	Barcelona Harbour, Spain	Tolosa <u>et al.</u> (1991)
	sediment cores	n.d.-35 ng/g (as chloride), decreasing with core depth	Palace Moat, Tokyo, Japan	Tugrul <u>et al.</u> (1983)
	sediment cores	n.d.-300 ng Sn/g	Toronto Harbour	Maguire and Tkacz (1985)
	sediment cores	n.d.-55 ng/g as chloride, depending on core depth	Lake Zurich, Switzerland	Muller, M.D. (1987)
	sediment cores	n.d.-154 ng/g	marinas in Lake Lucerne, Switzerland	Fent and Hurn (1991)
	algae	present, not quantitated	Mission and San Diego Bays, CA, USA	Ishii (1982)
	seaweed	n.d.-6 ng/g	Great Bay Estuary, NH, USA	Donard <u>et al.</u> (1987)
	eelgrass	n.d.-130 ng Sn/g	Great Bay Estuary, NH, USA	Francois and Weber (1988)
	blue mussel	n.d.-600 ng Sn/g dry wt.	Swedish coastal areas	Linden (1987)
	musshells (<u>M. edulis</u>)	<5-1240 ng Sn/g dry wt.	coastal USA	Wade <u>et al.</u> (1988)
	musshells	3-169 ng/g dry wt.	Portuguese estuaries and harbours	Quevauxiller <u>et al.</u> (1989)
	musshells	n.d.-260 ng/g	San Diego Bay	Stallard <u>et al.</u> (1989)
	musshells (<u>Dreissena polymorpha</u>)	100-980 ng/g	marinas in Lake Lucerne, Switzerland	Fent and Hurn (1991)
	musshells (<u>M. galloprovincialis</u>)	not quantitated	Barcelona Harbour, Spain	Tolosa <u>et al.</u> (1991)
	mussel and oyster tissue	n.d.-320 ng/g as chloride	USA harbours	Grovhoug <u>et al.</u> (1989)
	oyster sample	580 ng Sn/g dry wt.	France	Han and Weber (1988)
	oysters (<u>C. virginica</u> or <u>O. sandwichensis</u>)	<5-920 ng Sn/g dry wt.	coastal USA	Wade <u>et al.</u> (1988)
	oysters (<u>C. gigas</u>)	n.d.-581 ng Sn/g	English estuaries	Rapsomanikis and Harrison (1988)
	oysters (adult <u>C. gigas</u>)	n.d.-11900 ng/g	English rivers	Ebdon <u>et al.</u> (1989)

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Table 3 cont'd

Compound	Medium	Concentration	Location	Reference
	oysters (seed, <u>C. gigas</u>)	n.d.-16700 ng/g	English rivers	Ebdon <u>et al.</u> (1989)
	oyster (<u>Crassostrea gigas</u>)	7-34 ng Sn/g	British Columbia	Cullen <u>et al.</u> (1990)
	caged oysters (<u>S. commercialis</u>)	0.7-75 ng Sn/g	Wapengo Lake, Australia	Batley and Scammell (1991)
	bivalves	n.d.-314 ng/g dry wt.	coastal USA	Uhler <u>et al.</u> (1989)
	herring (<u>C. harengus pallasii</u>) (two-fish composite)	60 ng Sn/g	Vancouver Harbour	Maguire <u>et al.</u> (1986)
	English sole liver	<16 ng/g	Puget Sound, WA, USA	Krone <u>et al.</u> (1989b)
	English sole muscle	<3.5 ng/g	Puget Sound, WA, USA	Krone <u>et al.</u> (1989b)
	fish muscle tissue (<u>Leuciscus cephalus</u>)	3 ng/g	marina in Lake Murten, Switzerland	Fent and Hunn (1991)
	fish (unidentified)	29-179 ng Sn/g	harbours in Lakes Ontario and Erie	Scott <u>et al.</u> (1991)

Table 3 cont'd

Compound	Medium	Concentration	Location	Reference
<u>Bu₂Sn²⁺</u>				
	dry dock wash water	10000-370000 ng/L	England	Waldock <u>et al.</u> (1988)
	landfill leachate	243 ng/L	Switzerland	Fent (1991)
	organotin production plant effluent	8570 ng Sn/L	Germany	Schebek <u>et al.</u> (1991)
	untreated wastewater	136-554 ng/L	Zurich, Switzerland	Fent and Muller (1991)
	STP influent	127-191 ng/L	Zurich, Switzerland	Fent (1989, 1991)
	STP influent	n.d.-2400 ng Sn/L	five Canadian cities	Chau <u>et al.</u> (1991)
	STP effluent	25 ng/L as chloride	Switzerland	Muller, M.D. (1987)
	STP effluent	n.d.-2000 ng Sn/L	five Canadian cities	Chau <u>et al.</u> (1991)
	tertiary STP effluent	3-13 ng/L	Zurich, Switzerland	Fent and Muller (1991)
	STP effluent (domestic effluent only)	5-8 ng Sn/L	Germany	Schebek <u>et al.</u> (1991)
	STP effluent (40% domestic - 60% industrial effluent)	3-48 ng Sn/L	Germany	Schebek <u>et al.</u> (1991)
	sewage sludge	500-7000 ng/g as chloride	Switzerland	Muller, M.D. (1987)
	sewage sludge	600 ng/g	STP influent, Zurich, Switzerland	Fent (1989, 1991)
	sewage sludge	410-1240 ng/g	Zurich, Switzerland	Fent and Muller (1991)
	sewage sludge	n.d.-305 ng Sn/g	five Canadian cities	Chau <u>et al.</u> (1991)
	fresh water	10-1600 ng/L	Lake Michigan	Hodge <u>et al.</u> (1979)
	fresh water	n.d.-7300 ng/L	Ontario lakes and rivers	Maguire <u>et al.</u> (1982)
	fresh water	n.d.-100 ng Sn/L	Toronto Harbour	Maguire and Tkacz (1985)
	fresh water	n.d.-100 ng Sn/L	Detroit and St. Clair Rivers, Canada	Maguire <u>et al.</u> (1985a)

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Table 3 cont'd

Compound	Medium	Concentration	Location	Reference
	fresh water	n.d.-1400 ng Sn/L	Canada	Maguire <u>et al.</u> (1986)
	fresh water	6-40 ng/L as chloride	Switzerland	Muller, W.D. (1987)
	fresh water	48-15700 ng/L	harbour in Antwerp, Belgium	Dirkx <u>et al.</u> (1989)
	fresh water	n.d.-221 ng/L	Rivers Bure and Yare, England	Waite <u>et al.</u> (1989)
	fresh water	n.d.-30 ng/L	California marinas	Stang and Goldberg (1989)
	fresh water	n.d.-75 ng/L	marinas in Lake Lucerne, Switzerland	Fent and Hunn (1991)
	fresh water	0.5-2 ng Sn/L	Rhine River, Germany	Schebek <u>et al.</u> (1991)
	fresh water	1-16 ng Sn/L	Germany	Schebek <u>et al.</u> (1991)
	fresh water surface microlayer	n.d.-2600000 ng/L	Ontario lakes and rivers	Maguire <u>et al.</u> (1982)
	fresh water surface microlayer	n.d.-5 ng Sn/L	St. Clair River, Canada	Maguire <u>et al.</u> (1985a)
	fresh water surface microlayer	n.d.-366000 ng Sn/L	Ontario lakes and rivers	Maguire and Tkacz (1987)
	fresh and estuarine water surface microlayer	n.d.-1156 ng/L	tributaries of Chesapeake Bay	Hall <u>et al.</u> (1987)
	fresh and estuarine water	1-67 ng Sn/L	tributaries of Chesapeake Bay	Unger <u>et al.</u> (1986)
	fresh and estuarine water	n.d.-298 ng/L	tributaries of Chesapeake Bay	Hall <u>et al.</u> (1987)
	fresh and estuarine water	5-666 ng/L	Chesapeake Bay marinas and tributaries	Hall <u>et al.</u> (1988a)
	estuarine water	4-109 ng Sn/L	various US locations	Matthias <u>et al.</u> (1986b)
	estuarine water	<20-736 ng/L	Chesapeake Bay	Matthias <u>et al.</u> (1986a)
	estuarine water	n.d.-2.3 ng/L	Tamar and Tejo River estuaries (Portugal) and Delaware River, USA	Byrd and Andreea (1986)
	estuarine water	n.d.-415 ng/L	England	Waldock <u>et al.</u> (1987)
	estuarine water	12-135 ng/L	Back Creek, Chesapeake Bay	Johnson <u>et al.</u> (1987)
	estuarine water	n.d.-233 ng/L	Chesapeake Bay marina	Matthias <u>et al.</u> (1988)

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Table 3 cont'd

Compound	Medium	Concentration	Location	Reference
	estuarine and seawater	2-270 ng/L as chloride	England and USA	Valkirs <u>et al.</u> (1987)
	estuarine and seawater	n.d.-51 ng Sn/L	Sydney Harbour and Georges River, Australia	Batley <u>et al.</u> (1989b)
	estuarine water surface microlayer	91-150 ng Sn/L	Chesapeake Bay harbours	Matthias <u>et al.</u> (1986b)
	estuarine water surface microlayer	n.d.-2020 ng/L	Chesapeake Bay marina	Matthias <u>et al.</u> (1988)
	seawater	50-150 ng/L	San Diego Bay	Valkirs <u>et al.</u> (1985)
	seawater	0.41-0.80 ng Sn/L	Esquimalt Harbour, British Columbia	Kaye <u>et al.</u> (1986)
	seawater	n.d.-460 ng/L	San Diego Bay	Valkirs <u>et al.</u> (1986)
	seawater	n.d.-460 ng/L	California coast	Stallard <u>et al.</u> (1987)
	seawater	n.d.-70 ng Sn/L	Scotland	Balls (1987b)
	seawater	<1-118 ng/L	English harbours and anchorages	Waldock <u>et al.</u> (1988)
	seawater	n.d.-750 ng/L	Italian harbours and canals	Bacci and Gaggi (1989)
	seawater	<0.001-830 ng Sn/L	Esquimalt Harbour, British Columbia	Seakem (1989)
	seawater	n.d.-300 ng/L as chloride	USA harbours	Grothoung <u>et al.</u> (1989)
	seawater	<1-194 ng/L	France	Alzieu <u>et al.</u> (1989)
	seawater	240-280 ng/L as chloride	Hakata Harbour, Japan	Nagase (1990)
	seawater	n.d.-160 ng/L	European coastal areas	Quevauviller and Donard (1990)
	seawater	n.d.-750 ng/L	Mediterranean Sea coastal areas	Gabrielides <u>et al.</u> (1990)
	seawater	n.d.-304 ng/L	various locations in French Mediterranean Sea	Alzieu <u>et al.</u> (1990)
	seawater	n.d.-58 ng Sn/L	British Columbia	Cullen <u>et al.</u> (1990)
	seawater	n.d.-74 ng/L as chloride (variation over 1 year)	Maizuru Bay, Japan	Takahashi (1991b)

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Table 3 cont'd

Compound	Medium	Concentration	Location	Reference
	seawater	3-52 ng Sn/L	La Spezia Gulf, Italy	Chiavarini <u>et al.</u> (1991)
	seawater	n.d.-400 ng/L as chloride	bays in Japan	Takahashi (1991a)
	seawater surface microlayer	28-116 ng Sn/L	British Columbia	Cullen <u>et al.</u> (1990)
	sediment	n.d.-350 ng/g	Ontario lakes and rivers	Maguire (1984)
	sediment	470 ng/g as chloride	river in Osaka, Japan	Hattori <u>et al.</u> (1984)
	sediment	n.d.-300 ng Sn/g	Toronto Harbour	Maguire and Tkacz (1985)
	sediment	n.d.-36 ng Sn/g	Detroit and St. Clair Rivers, Canada	Maguire <u>et al.</u> (1985a)
	sediment	7-56 ng/g	San Diego Bay	Valkirs <u>et al.</u> (1985)
	sediment	n.d.-8500 ng Sn/g	Canada	Maguire <u>et al.</u> (1986)
	sediment	n.d.-15.0 ng/g	Great Bay Estuary, NH, USA	Randall <u>et al.</u> (1986b)
	sediment	120-1400 ng Sn/g	Esquimalt Harbour, British Columbia	Kaye <u>et al.</u> (1986)
	sediment	1.3-4.0 ng/g as chloride	Japan	Tsuda <u>et al.</u> (1986)
	sediment	n.d.-27 ng/g	California coast	Stallard <u>et al.</u> (1987)
	sediment	10-570 ng Sn/g	Poole Harbour, England	Langston <u>et al.</u> (1987)
	sediment	n.d.-2200 ng/g	Chesapeake Bay marina	Matthias <u>et al.</u> (1988)
	sediment	n.d.-425 ng Sn/g	Alexandria, Egypt	About Dahab (1988)
	sediment	n.d.-9600 ng/g	Portugese rivers, estuaries and	Quevauviller <u>et al.</u> (1989)
	sediment	<4-950 ng Sn/g	marinas and waterways in Washington, USA	Krone <u>et al.</u> (1989a)
	sediment	<2-701 ng/g	USA coastal harbours	Kram <u>et al.</u> (1989a)
	sediment	5-35 ng Sn/g	Ise Bay, Japan	Yonezawa <u>et al.</u> (1989)
	sediment	n.d.-18 ng Sn/g	Oshawa and Whitby Harbours	Chau <u>et al.</u> (1989)
	sediment	13 ng Sn/g	Porto Vecchio Bay, Corsica, France	Astruc <u>et al.</u> (1989)

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Table 3 cont'd

Compound	Medium	Concentration	Location	Reference
	sediment	n.d.-5500 ng/g	English rivers	Ashby and Craig (1989)
	sediment	n.d.-367 ng/g	San Diego Bay and Pearl Harbor	Kram <u>et al.</u> (1989b)
	sediment	n.d.-170 ng/g	Puget Sound, WA, USA	Krone <u>et al.</u> (1989b)
	sediment	n.d.-1500 ng/g as chloride	USA harbours	Grothoung <u>et al.</u> (1989)
	sediment	n.d.-316 ng/g	Boston Harbor	Makkar <u>et al.</u> (1989)
	sediment	200-4900 ng Sn/g	France	Desauziers <u>et al.</u> (1989)
	sediment (spring)	n.d.-265 ng/g	Severn Sound, Ontario	Wong and Chau (1990)
	sediment (summer)	n.d.-344 ng/g	Severn Sound, Ontario	Wong and Chau (1990)
	sediment (summer)	n.d.-70 ng/g	Severn Sound, Ontario	Wong and Chau (1990)
	sediment	23-559 ng Sn/g	Lake Maryut, Egypt	Aboul Dahab <u>et al.</u> (1990)
	sediment	<5-63 ng Sn/g	USA coastal areas	Wade <u>et al.</u> (1990)
	sediment	10-50 ng Sn/g	Rhine River, Germany	Schebek <u>et al.</u> (1991)
	sediment	15-44 ng Sn/g	Germany	Schebek <u>et al.</u> (1991)
	sediment	n.d.-97 ng Sn/g	harbours in Lake Erie	Scott <u>et al.</u> (1991)
	sediment cores	n.d.-700 ng Sn/g	Toronto Harbour	Maguire and Tkacz (1985)
	sediment cores	n.d.-140 ng/g as chloride, depending upon depth	Lake Zurich, Switzerland	Muller, M.D. (1987)
	sediment cores	0.1-144 ng/g	marinas in Lake Lucerne, Switzerland	Fent and Hunn (1991)
	seaweed (<i>F. vesiculosus</i>)	20-290 ng Sn/g dry wt.	Poole Harbour, England	Langston <u>et al.</u> (1987)
	polychaetes (<i>N. diversicolor</i>)	0.03-1.6 ng Sn/g dry wt.	Poole Harbour, England	Langston <u>et al.</u> (1987)
	snails (<i>L. littorea</i>)	0.23-3.22 ng Sn/g dry wt.	Poole Harbour, England	Langston <u>et al.</u> (1987)
	dogwhelk (<i>N. lapillus</i>)	n.d.-750 ng/g dry wt.	southwest English rivers	Bryan <u>et al.</u> (1986)
	dogwhelk (<i>N. lapillus</i>)	n.d.-1000 ng Sn/g dry wt.	Fal River Estuary	Bryan <u>et al.</u> (1987)

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Table 3 cont'd

Compound	Medium	Concentration	Location	Reference
	dogwhelk (<u>N. lapillus</u>)	5-21 ng Sn/g dry wt.	Pennyghael, Scotland	Bryan <u>et al.</u> (1987)
	neogastropods (<u>N. lamellosa</u> , <u>N. canaliculata</u> and <u>N. emarginata</u>)	n.d.- approx. 350 ng Sn/g	coastal British Columbia	Bright and Ellis (1990)
	eelgrass	n.d.-190 ng Sn/g dry wt.	Great Bay Estuary, NH, USA	Francois and Weber (1988)
	blue mussel	n.d.-980 ng Sn/g dry wt.	Swedish coastal areas	Linden (1987)
	mussels	90-540 ng/g	San Diego Bay	Stallard <u>et al.</u> (1989)
	mussels (<u>M. edulis</u>)	<5-870 ng Sn/g dry wt.	coastal USA	Wade <u>et al.</u> (1988)
	mussels	n.d.-82 ng/g dry wt.	Portugese estuaries and harbours	Quevaufiller <u>et al.</u> (1989)
	caged mussels (<u>E. complanata</u>)	n.d.-5 ng Sn/g	Oshawa and Whitby Harbours, Ontario	Chau <u>et al.</u> (1989)
	mussels (<u>Dreissena polymorpha</u>)	470-2430 ng/g	marinas in Lake Lucerne, Switzerland	Fent and Kunn (1991)
	mussel and oyster tissue	n.d.-2100 ng/g as chloride	USA harbours	Grovhou <u>et al.</u> (1989)
	oyster sample	840 ng Sn/g dry wt.	France	Han and Weber (1988)
	oysters (<u>C. virginica</u> or <u>O. sancti-chensis</u>)	<5-1280 ng Sn/g	coastal USA	Wade <u>et al.</u> (1988)
	oysters (<u>C. gigas</u>)	n.d.-402 ng Sn/g	English estuaries	Rapsomanikis and Harrison (1988)
	oyster (<u>S. commercialis</u>)	<1-19 ng Sn/g	Australian estuaries	Batley <u>et al.</u> (1989a)
	oyster (<u>C. gigas</u>)	35 ng Sn/g	Georges River estuary, Australia	Batley <u>et al.</u> (1989a)
	oyster (adult <u>C. gigas</u>)	n.d.-11300 ng/g	English rivers	Ebdon <u>et al.</u> (1989)
	oyster (seed <u>C. gigas</u>)	n.d.-16100 ng/g	English rivers	Ebdon <u>et al.</u> (1989)
	oyster shell (<u>C. gigas</u>)	20000 ng Sn/g	Denman I., British Columbia	Cullen <u>et al.</u> (1989)
	oyster (<u>Crassostrea gigas</u>)	17.5-178 ng Sn/g	British Columbia	Cullen <u>et al.</u> (1990)

Table 3 cont'd

Compound	Medium	Concentration	Location	Reference
	caged oysters (<u>S. commercialis</u>)	5-15 ng Sn/g	Wapengo Lake, Australia	Batley and Scammell (1991)
	bivalve (<u>S. plana</u>)	0.27-7.29 ng Sn/g dry wt.	Poole Harbour, England	Langston et al. (1987)
	bivalve (<u>M. arenaria</u>)	1.78-9.95 ng Sn/g dry wt.	Poole Harbour, England	Langston et al. (1987)
	bivalves	2-1708 ng/g dry wt.	coastal USA	Uhlir et al. (1989)
	herring (<u>C. harengus pallasi</u>) (two-fish composite)	50 ng Sn/g	Vancouver Harbour	Maguire et al. (1986)
	fish	n.d.-5.3 ng/g as chloride	Japan	Tsuda et al. (1986)
	fish (yellowtails) from fish farms	20-160 ng/g	Japan	Sasaki et al. (1988a)
	fish (yellowtails) from fish farms	n.d.-3740 ng/g as chloride (more in liver than muscle)	Japan	Sasaki et al. (1988b)
	English sole liver	n.d.-25 ng/g	Puget Sound, WA, USA	Krone et al. (1989b)
	English sole muscle	n.d.-11 ng/g	Puget Sound, WA, USA	Krone et al. (1989b)
	fish	n.d.-56 ng/g as chloride	Japan	Ishizaka et al. (1989a)
	fish muscle tissue (<u>Leuciscus cephalus</u>)	20 ng/g	marina in Lake Murten, Switzerland	Fent and Hunn (1991)
	fish (unspecified)	4-109 ng Sn/g	harbours in Lakes Ontario and Erie	Scott et al. (1991)

*All concentrations in sediment and sludge are by dry weight, and in biota are by wet weight, unless otherwise noted. Concentrations are reported in various ways by the original authors: (i) as the cation (e.g., $\text{Bu}_2\text{Sn}^{2+}$), (ii) unspecified, hence reported as the cation, (iii) as the chloride, and (preferably) (iv) in terms of Sn (e.g., ng Sn/L). Reporting concentrations in terms of Sn is preferable because it allows easy calculations of mass balances of homologous species and comparisons of toxicity. Note however that the errors involved in comparing different units reported here are quite small compared to experimental errors in determining concentrations of organotin species in environmental media. Abbreviation used: STP - sewage treatment plant.

Table 4. Toxicity of mono- and di-methyltin, n-butyltin and n-octyltin compounds.

Compound	Species	Parameter	Concentration/Dose	Reference
<u>MeSn³⁺</u>				
trichloride	bacteria (<u>Microtox</u> ⁿ)	EC ₅₀	10 mg/L as chloride	Dooley and Ken's (1987)
trichloride	yeasts	48-hr IC ₅₀	0.5-14 mg/L	Cooney <u>et al.</u> (1989)
trichloride	alga (<u>A. falcatus</u>)	IC-50	23 mg/L	Wong <u>et al.</u> (1982)
trichloride	diatom (<u>S. costatum</u>)	EC ₅₀	0.08 mg/L	Walsh <u>et al.</u> (1985)
trichloride	diatom (<u>S. costatum</u>)	EC ₅₀	0.34 mg/L	Walsh <u>et al.</u> (1985)
trichloride	water flea (<u>D. magna</u>)	24 hr EC ₅₀	90 mg/L	Vighi and Calamari (1985)
trichloride	rat	oral LD ₅₀	1370 mg/kg	Lanigan and Weinberg (1976)
tris(isooctyl mercaptoacetate)	bacterium (<u>P. phosphoreum</u>)	EC ₅₀	2.3 mg/L	Steinhauser <u>et al.</u> (1985)
tris(isooctyl mercaptoacetate)	water flea (<u>D. magna</u>)	EC ₅₀	> 2.9 mg/L	Steinhauser <u>et al.</u> (1985)
tris(isooctyl mercaptoacetate)	rat	oral LD ₅₀	920 mg/kg	Lanigan and Weinberg (1976)

Table 4 cont'd

Compound	Species	Parameter	Concentration/Dose	Reference
<u>Me₂Sn²⁺</u>				
dichloride	bacteria (<u>Microtox</u> ^b)	EC ₅₀	2.6 mg/L as chloride	Dooley and Kenis (1987)
dichloride	yeasts	48-hr LC ₅₀	3-11 mg/L	Cooney <u>et al.</u> (1989)
dichloride	alga (<u>A. falcatus</u>)	IC-50	21 mg/L	Wong <u>et al.</u> (1982)
dichloride	diatom (<u>S. costatum</u>)	EC ₅₀	> 0.5 mg/L	Walsh <u>et al.</u> (1985)
dichloride	diatom (<u>S. costatum</u>)	EC ₅₀	> 0.5 mg/L	Walsh <u>et al.</u> (1985)
dichloride	water flea (<u>D. magna</u>)	24 hr EC ₅₀	88 mg/L	Vighi and Calamari (1985)
dichloride	mud crab larva (<u>R. harrisi</u>)	LC ₅₀	20 mg/L	Laughtin <u>et al.</u> (1985)
dichloride	red killifish (<u>O. latipes</u>)	48-hr LC ₅₀	6 mg/L	Nagase <u>et al.</u> (1991)
dichloride	loach (<u>M. fossilis</u>)	MATC for various development stages	100-1000 mg/L	Daniil'chenko (1977)
dichloride	rat	oral LD ₅₀	75 mg/kg	Lanigan and Weinberg (1976)
bis(isooctyl mercaptoacetate)	bacterium (<u>P. phosphoreum</u>)	EC ₅₀	0.1 mg/L	Steinhauser <u>et al.</u> (1985)
bis(isooctyl mercaptoacetate)	water flea (<u>D. magna</u>)	EC ₅₀	> 0.13 mg/L	Steinhauser <u>et al.</u> (1985)
bis(isooctyl mercaptoacetate)	alga (<u>S. suspicatus</u>)	EC ₁₀	> 0.07 mg/L	Steinhauser <u>et al.</u> (1985)
bis(isooctyl mercaptoacetate)	rat	oral LD ₅₀	1210 mg/kg	Lanigan and Weinberg (1976)

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Table 4 cont'd

Compound	Species	Parameter	Concentration/Dose	Reference
<u>BuSn³⁺</u>				
trichloride	bacteria (Microtox [®])	EC ₅₀	13 mg/L as chloride	Dooley and Kenis (1987)
trichloride	yeasts	48-hr IC ₅₀	5-17 mg/L	Cooney et al. (1989)
trichloride	alga (<u>A. falcatus</u>)	IC-50	25.0 mg/L	Wong et al. (1982)
trichloride	water flea (<u>D. magna</u>)	24 hr EC ₅₀	49 mg/L	Vighi and Calamari (1985)
trichloride	red killifish (<u>O. latipes</u>)	48-hr LC ₅₀	38 mg/L	Nagase et al. (1991)
trichloride	mouse	oral LD ₅₀	1400 mg/kg	Pelikan and Cerny (1970b)
trichloride	rat	oral LD ₅₀	2300 mg/kg	Lanigan and Weinberg (1976)
trichloride	rat	oral LD ₅₀	2200 mg/kg	Smith (1978)
trichloride	rat	oral LD ₅₀	2300 mg/kg	Smith (1978)
trichloride	rat	oral LD ₅₀	2200 - 2395 mg/kg	Luijten and Klimmer (1978)
tris(2-ethylhexyl mercaptoacetate)	mouse	oral LD ₅₀	1520 mg/kg	Pelikan and Cerny (1970b)
tris(isooctyl mercaptoacetate)	rat	oral LD ₅₀	1063 mg/kg	Lanigan and Weinberg (1976)

Table 4 cont'd

Compound	Species	Parameter	Concentration/Dose	Reference
<u>Bu₂Sn²⁺</u>				
diacetate	diatom (<u>S. costatum</u>)	EC ₅₀	0.04 mg/L	Walsh <u>et al.</u> (1985)
diacetate	diatom (<u>I. pseudonana</u>)	EC ₅₀	0.13 mg/L	Walsh <u>et al.</u> (1985)
diacetate	diatom (<u>S. costatum</u>)	LC ₅₀	> 0.5 mg/L	Walsh <u>et al.</u> (1985)
diacetate	red killifish (<u>O. latipes</u>)	48-hr LC ₅₀	3 mg/L	Nagase <u>et al.</u> (1991)
diacetate	mouse	oral LD ₅₀	110 mg/kg	Smith (1978)
dichloride	bacterium (<u>P. phosphoreum</u>)	EC ₅₀	0.2 mg/L	Steinhauser <u>et al.</u> (1985)
dichloride	bacterium (<u>P. putida</u>)	EC ₁₀	2.5 mg/L	Steinhauser <u>et al.</u> (1985)
dichloride	bacteria (Microtox [®])	EC ₅₀	0.3 mg/L as chloride	Dooley and Kenis (1987)
dichloride	yeasts	48-hr IC ₅₀	3.7-22 mg/L	Cooney <u>et al.</u> (1989)
dichloride	alga (<u>A. falcatus</u>)	IC ₅₀	6.8 mg/L	Wong <u>et al.</u> (1982)
dichloride	diatom (<u>S. costatum</u>)	EC ₅₀	0.04 mg/L	Walsh <u>et al.</u> (1985)
dichloride	diatom (<u>I. pseudonana</u>)	EC ₅₀	0.2 mg/L	Walsh <u>et al.</u> (1985)
dichloride	diatom (<u>S. costatum</u>)	LC ₅₀	> 0.5 mg/L	Walsh <u>et al.</u> (1985)
dichloride	zooplankton (adult) (<u>Calanus</u> sp.)	no effect over 7 d	0.08-1.1 mg/L	Thain <u>et al.</u> (1987)
dichloride	water flea (<u>D. magna</u>)	24 hr EC ₅₀	0.9 mg/L	Vighi and Calamari (1985)
dichloride	mosquito larva (<u>C. pipiens</u>)	24-h LC ₅₀	0.38 mg/L	Gras and Rioux (1965)
dichloride	mosquito larva (<u>C. pipiens</u>)	24-h LC ₅₀	0.69 mg/L	Gras and Rioux (1965)

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Table 4 cont'd

Compound	Species	Parameter	Concentration/Dose	Reference
dichloride	polychaetes (juvenile) (<u>N. diversicolor</u>)	no effect over 10 d	0.1 mg/L	Thain <u>et al.</u> (1987)
dichloride	oyster (<u>C. gigas</u>) (spat)	49-d LC ₅₀	0.1 mg/L	Thain <u>et al.</u> (1987)
dichloride	mud crab larva (<u>R. harrisi</u>)	LC ₅₀	0.9 mg/L	Laughlin <u>et al.</u> (1985)
dichloride	crab (adult) (<u>C. maenus</u>)	no effect over 28 d	0.5 mg/L	Thain <u>et al.</u> (1987)
dichloride	shrimp (adult) (<u>C. crangon</u>)	no effect over 28 d	0.5-0.75 mg/L	Thain <u>et al.</u> (1987)
dichloride	flatfish (juvenile) (<u>S. soles</u>)	no effect over 21 d	0.5 mg/L	Thain <u>et al.</u> (1987)
dichloride	sturgeon poacher (adult) (<u>Agonus cataphractus</u>)	no effect over 18 d	0.5 mg/L	Thain <u>et al.</u> (1987)
dichloride	red killifish (<u>O. latipes</u>)	48-hr LC ₅₀	6 mg/L	Nagase <u>et al.</u> (1991)
dichloride	golden orfe	LC ₅₀	0.6 mg/L	Steinhauser <u>et al.</u> (1985)
dichloride	rat	oral LD ₁₀₀	100 mg/kg	Barnes and Magee (1958)
dichloride	rat	oral LD ₅₀	219 mg/kg	Smith (1978)
dichloride	rat	oral LD ₁₀₀	100 mg/kg	Smith (1978)
dichloride	rat	oral LD ₅₀	126 mg/kg	Smith (1978)
dichloride	rat	oral LD ₅₀	112-182 mg/kg	Smith (1978)
dichloride	rat	oral LD ₅₀	58 - 219 mg/kg	Luijten and Klimmer (1978)
dichloride	rat	oral LD ₅₀	126 mg/kg	Lanigan and Weinberg (1976)
dichloride	rat	inhalation LC ₅₀	73 mg/L/hr (vapour)	Smith (1978)

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Table 4 cont'd

Compound	Species	Parameter	Concentration/Dose	Reference
dichloride	rat	inhalation LC ₅₀	>1.5 mg/L/hr (vapour)	Smith (1978)
difluoride	diatom (<i>S. costatum</i>)	EC ₅₀	0.06 mg/L	Walsh <u>et al.</u> (1985)
difluoride	diatom (<i>I. pseudonana</i>)	EC ₅₀	0.28 mg/L	Walsh <u>et al.</u> (1985)
oxide	red killifish (<i>O. latipes</i>)	48-hr LC ₅₀	1 mg/L	Nagase <u>et al.</u> (1991)
oxide	rat	oral LD ₅₀	487 mg/kg	Smith (1978)
oxide	rat	oral LD ₅₀	520 mg/kg	Smith (1978)
oxide	rat	oral LD ₅₀	ca. 500 mg/kg	Smith (1978)
oxide	rat	oral LD ₅₀	68-800 mg/kg	Luijten and Klimmer (1978)
oxide	rat	intraperitoneal LD ₅₀	39.9 mg/kg	Robinson (1969)
maleate	red killifish (<i>O. latipes</i>)	48-hr LC ₅₀	12 mg/L	Nagase <u>et al.</u> (1991)
bis(methyl maleate)	rat	oral LD ₅₀	62-1330 mg/kg	Luijten and Klimmer (1978)
bis(n-butyl maleate)	rat	oral LD ₅₀	120 mg/kg	Luijten and Klimmer (1978)
dilaurate	bacterium (<i>P. phosphoreum</i>)	EC ₅₀	0.57 mg/L	Steinhauser <u>et al.</u> (1985)
dilaurate	mosquito larva (<i>C. pipiens</i>)	24-h LC ₅₀	0.71 mg/L	Gras and Rioux (1965)
dilaurate	mosquito larva (<i>C. pipiens</i>)	24-h LC ₅₀	8.74 mg/L	Gras and Rioux (1965)
dilaurate	water flea (<i>D. magna</i>)	EC ₅₀	0.66 mg/L	Steinhauser <u>et al.</u> (1985)
dilaurate	golden orfe	LC ₅₀	2 mg/L	Steinhauser <u>et al.</u> (1985)
dilaurate	red killifish (<i>O. latipes</i>)	48-hr LC ₅₀	1 mg/L	Nagase <u>et al.</u> (1991)

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Table 4 cont'd

Compound	Species	Parameter	Concentration/Dose	Reference
dilaurate	rat	oral LD ₅₀	200-300 mg/kg	Smith (1978)
dilaurate	rat	oral LD ₅₀	175 mg/kg	Smith (1978)
dilaurate	rat	oral LD ₅₀	1600 mg/kg	Smith (1978)
dilaurate	rat	intraperitoneal LD ₁₀₀	85 mg/kg	Stoner et al. (1955)
bis(isooctyl mercaptoacetate)	rat	oral LD ₅₀	510 mg/kg	Lenigan and Weinberg (1976)
bis(isooctyl mercaptoacetate)	rat	oral LD ₅₀	510 mg/kg	Smith (1978)
bis(isooctyl mercaptoacetate)	rat	oral LD ₅₀	1037 mg/kg	Smith (1978)
bis(isooctyl mercaptoacetate)	rat	oral LD ₅₀	500 mg/kg	Smith (1978)

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Table 4, cont'd

Compound	Species	Parameter	Concentration/Dose	Reference
<u>OctSn³⁺</u>				
trichloride	rat	oral LD ₅₀	3800 mg/kg	Lanigan and Weinberg (1976)
trichloride	rat	oral LD ₅₀	2400 mg/kg	Smith (1978)
trichloride	rat	oral LD ₅₀	3800 mg/kg	Smith (1978)
trichloride	rat	oral LD ₅₀	2100-4700 mg/kg	Luijten and Klijmer (1978)
tris(2-ethylhexyl mercaptoacetate)	mouse	oral LD ₅₀	1500 mg/kg	Pelikan and Cerny (1970a)
tris(isooctyl mercaptoacetate)	rat	oral LD ₅₀	3400 mg/kg	Lanigan and Weinberg (1976)

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Table 4 cont'd

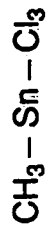
Compound	Species	Parameter	Concentration/Dose	Reference
<u>Oct₂Sn²⁺</u>				
diacetate	rat	intraperitoneal LD ₅₀	> 800 mg/kg	Robinson (1969)
dichloride	bacterium (<u>P. phosphoreum</u>)	EC ₅₀	0.002 mg/L	Steinhauser <u>et al.</u> (1985)
dichloride	mosquito larva (<u>C. pipiens</u>)	24-h LC ₅₀	> 2.5 mg/L	Gras and Rioux (1965)
dichloride	water flea (<u>D. magna</u>)	EC ₅₀	0.005 mg/L	Steinhauser <u>et al.</u> (1985)
dichloride	rat	oral LD ₅₀	7000 mg/kg	Lanigan and Weinberg (1976)
dichloride	rat	thymus atrophy	> 50 mg/kg (oral)	Seinen and Willems (1976)
maleate	mouse	oral LD ₅₀	4600 mg/kg	Smith (1978)
maleate	rat	oral LD ₅₀	4500 mg/kg	Smith (1978)
maleate	rat	oral LD ₅₀	4000-8500 mg/kg	Luijten and Klimmer (1978)
dilaurate	rat	oral LD ₅₀	> 6000 mg/kg	Luijten and Klimmer (1978)
bis(lauryl mercaptide)	mouse	oral LD ₅₀	4000 mg/kg	Pelikan and Cerny (1970a)
bis(isooctyl mercaptoacetate)	rat	oral LD ₅₀	1975 mg/kg	Smith (1978)
bis(isooctyl mercaptoacetate)	rat	oral LD ₅₀	2100 mg/kg	Smith (1978)
bis(isooctyl mercaptoacetate)	rat	oral LD ₅₀	1200 mg/kg	Smith (1978)
bis(isooctyl mercaptoacetate)	rat	dermal LD ₅₀	2250 mg/kg	Smith (1978)
bis(isooctyl mercaptoacetate)	mouse	oral LD ₅₀	2010 mg/kg	Pelikan and Cerny (1970a)

Abbreviations used:

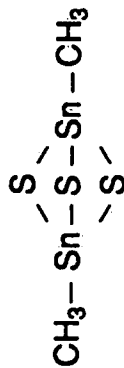
EC₅₀, IC₅₀ - concentration which effectively inhibits 50% of function/growth (in some cases used interchangeably)LC₅₀ - concentration which kills 50% of test organismsLD₅₀ - dose which kills 50% of test organisms

MATC - maximum acceptable toxicant concentration

Figure 1. Structures of monomethyltin compounds: methyltin trichloride (1), methyltin sulfide (2), methyltin tris(laurylmercaptide) (3), methyltin tris(isooctyl mercaptoacetate) (4), methyltin tris(2-ethylhexyl mercaptoacetate) (5), methyltin tris(2-mercaptoethyl oleate) (6), methyltin tris(lauroyloxyethylmercaptide) (7), methyltin tris(2-tetradecanoyloxyethylmercaptide) (8), methyltin tris(2-linoleoyloxyethylmercaptide) (9), and methyltin tris(decyloxyloxyethylmercaptide) (10).



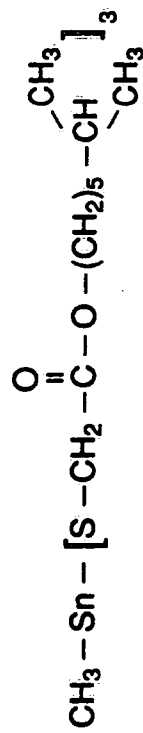
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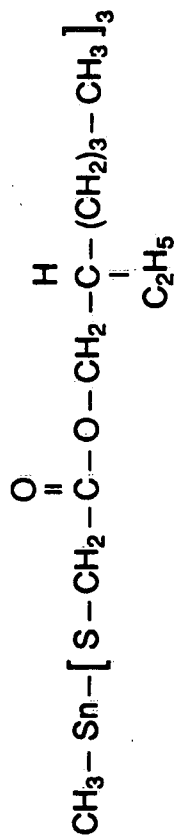
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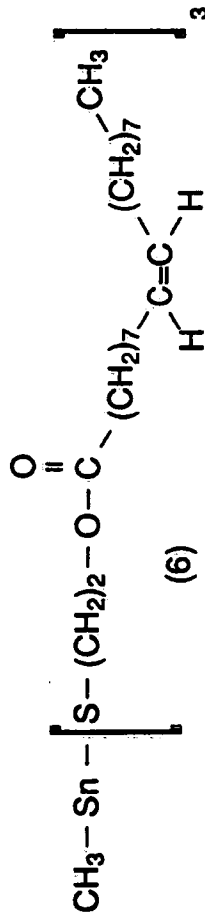
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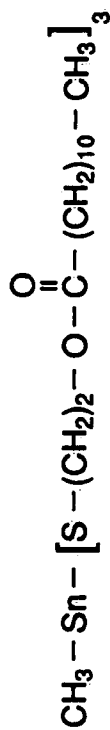
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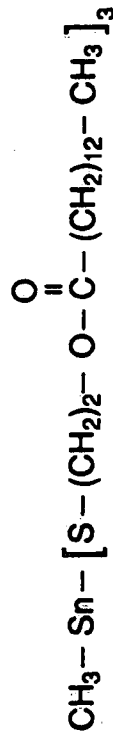
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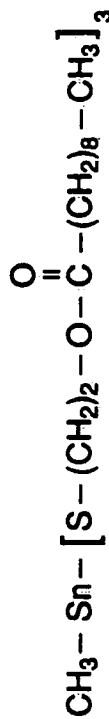
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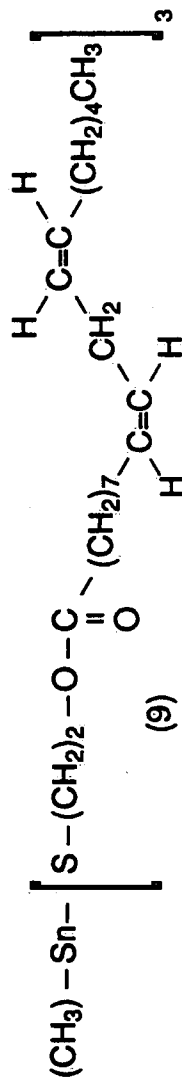
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(8)

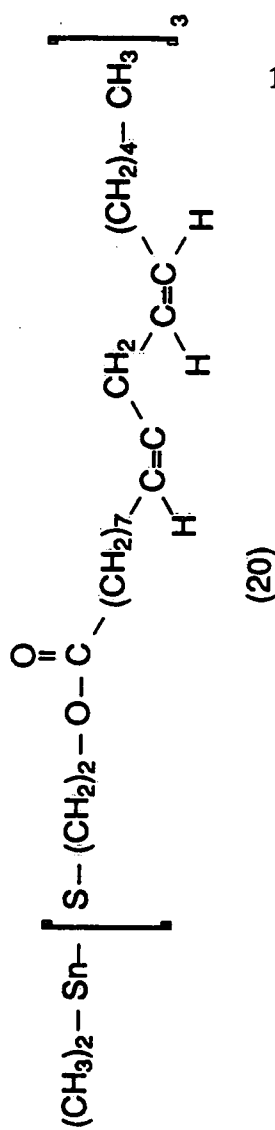
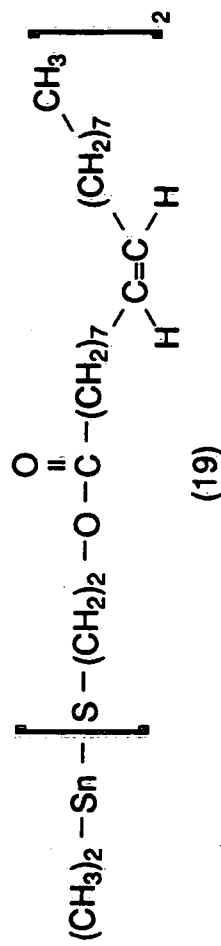
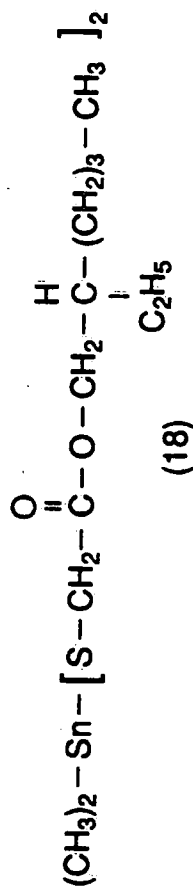
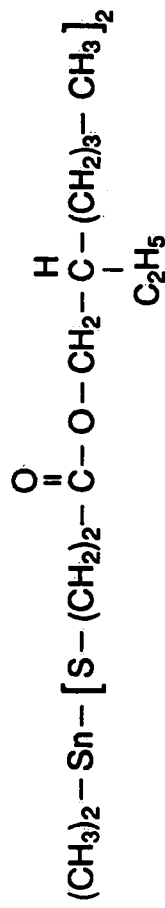
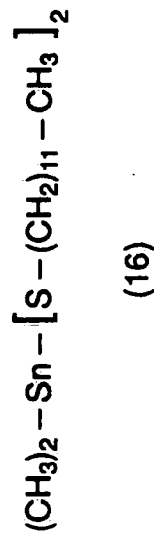
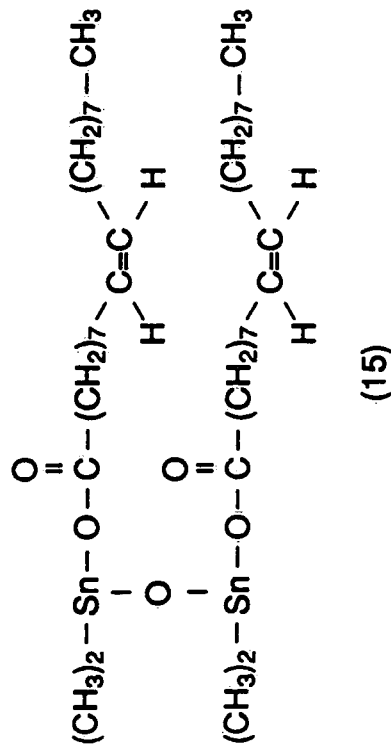
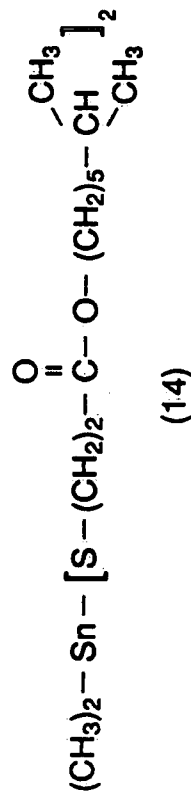
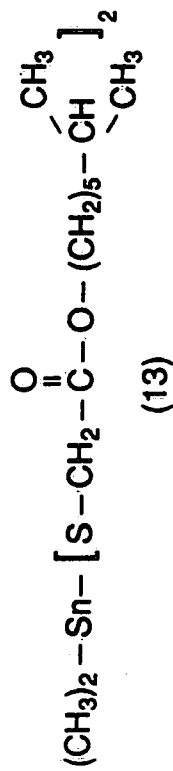


(10)



(9)

Figure 2. Structures of dimethyltin compounds: dimethyltin dichloride (11), dimethyltin sulfide (12), dimethyltin bis(isooctyl mercaptoacetate) (13), dimethyltin bis(isooctyl mercaptopropionate) (14), 1,1,3,3-tetramethyl-1,3-bis(oleoyloxy)distannoxane (15), dimethyltin bis(laurylmercaptide) (16), dimethyltin bis(2-ethylhexyl mercaptopropionate) (17), dimethyltin bis(2-ethylhexyl mercaptoacetate) (18), dimethyltin bis(2-oleoyloxyethylmercaptide) (19), dimethyltin bis(2-linoleoyloxyethylmercaptide) (20), dimethyltin bis(decyloxyloxyethylmercaptide) (21), dimethyltin bis(lauroyloxyethylmercaptide) (22), and dimethyltin bis(neodecanoate) (23).



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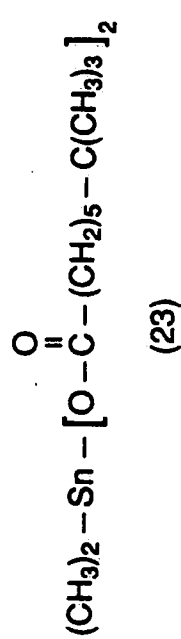
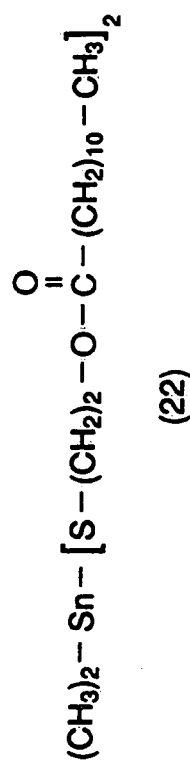
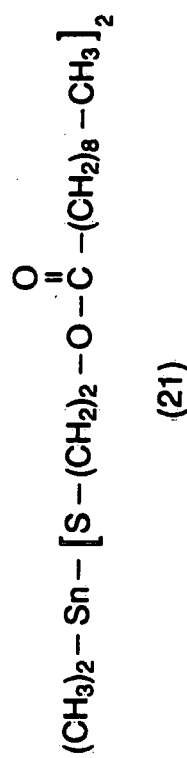


Figure 3. Structures of mono-n-butyltin compounds: n-butyltin trichloride (24), n-butylstannous acid (25), n-butylchlorotin dihydroxide (26), n-butyltin sulfide (27), n-butyltin tris(2-ethylhexanoate) (28), n-butyltin tris(isooctyl mercaptoacetate) (29), n-butylthiostannous acid (30), n-butyltin tris(2-ethylhexyl mercaptoacetate) (31), n-butyltin tris(isooctyl mercaptopropionate) (32), n-butyltin tris(2-oleoyloxyethylmercaptide) (33), n-butyltin tris(2-decanoyloxyethylmercaptide) (34), n-butyltin tris(2-tetradecanoyloxyethylmercaptide) (35), n-butyltin tris(2-linoleoyloxyethylmercaptide) (36), n-butyltin tris(2-lauroyloxyethylmercaptide) (37).

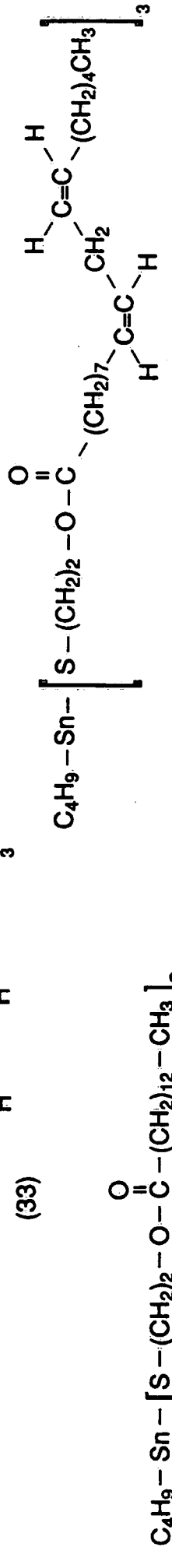
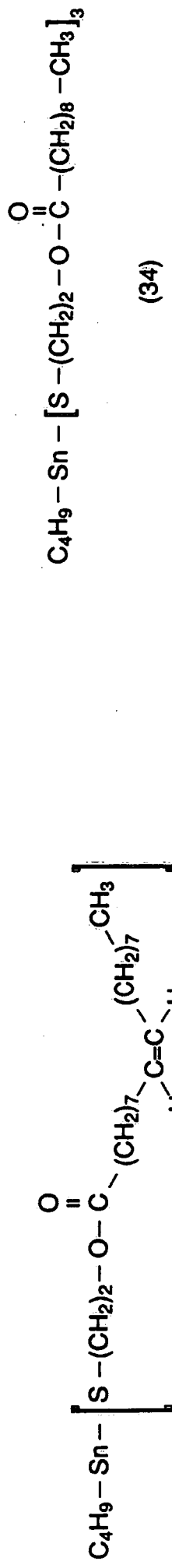
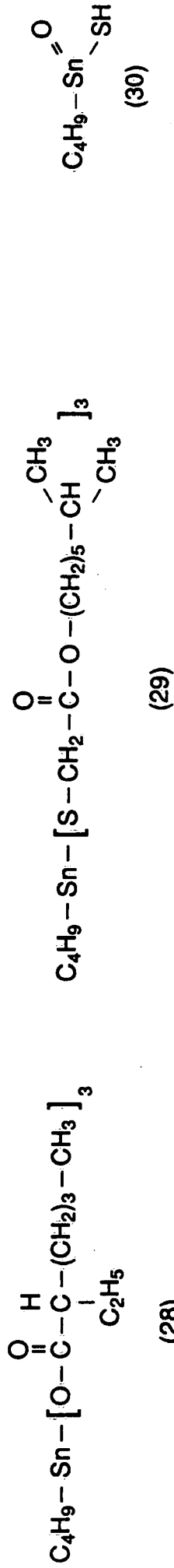
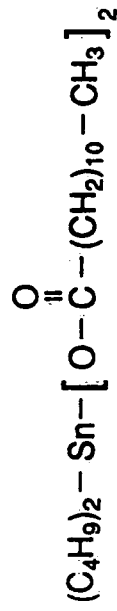
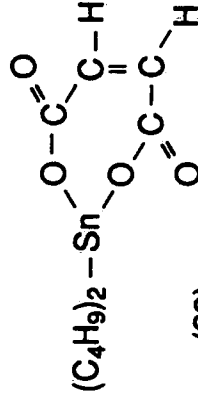


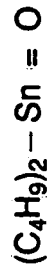
Figure 4a. Structures of di-n-butyltin compounds: di-n-butyltin dilaurate (38), di-n-butyltin maleate (39), di-n-butyltin oxide (40), di-n-butyltin diacetate (41), di-n-butyltin bis(laurylmercaptide) (42), di-n-butyltin bis(2-ethylhexanoate) (43), di-n-butyltin sulfide (44), di-n-butyltin distearate (45), di-n-butyltin bis(methyl maleate) (46), di-n-butyltin bis(2-ethylhexyl maleate) (47), di-n-butyltin bis(n-butyl maleate) (48), di-n-butyltin bis(acetylacetonate) (49), di-n-butyltin bis(isooctyl maleate) (50), di-n-butyltin bis(isooctyl mercaptoacetate) (51), and di-n-butyltin bis(isooctyl mercaptopropionate) (52).



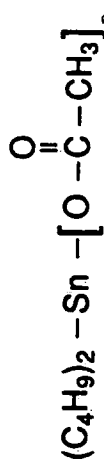
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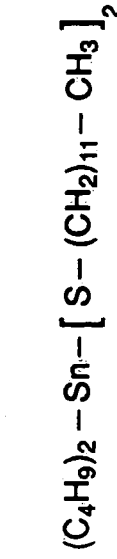
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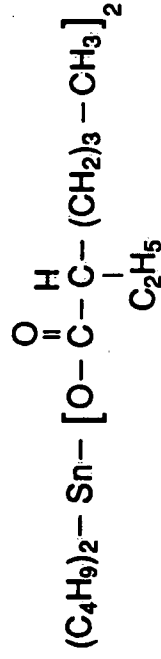
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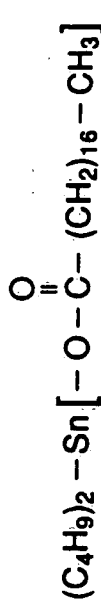
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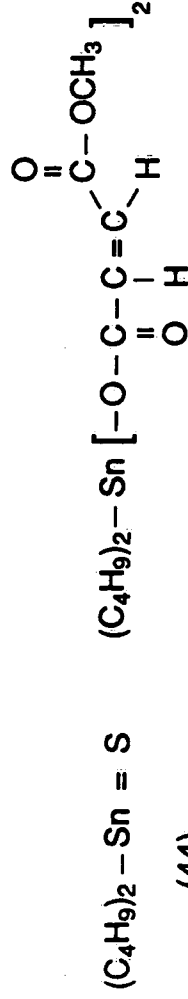
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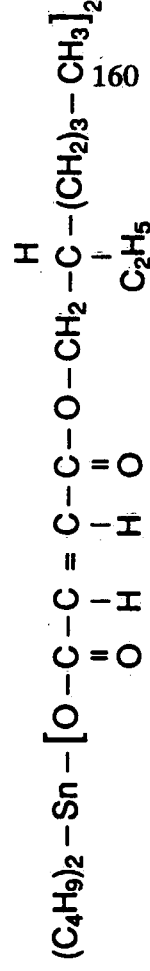
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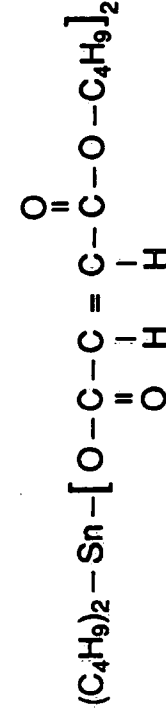


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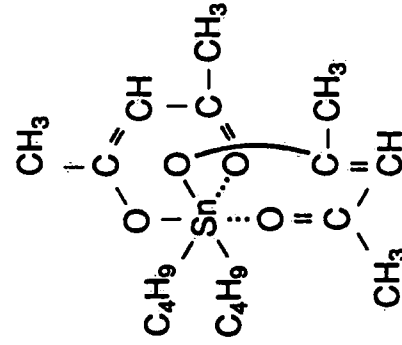


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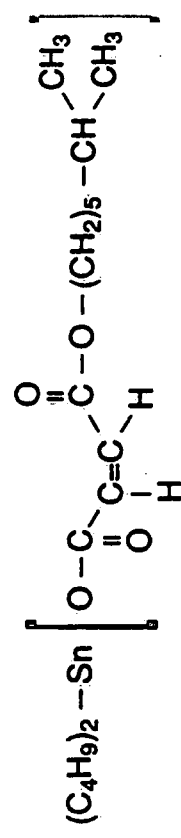
(47)



(48)



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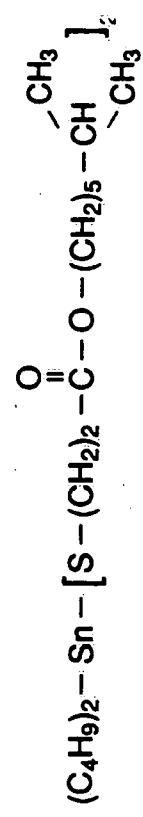
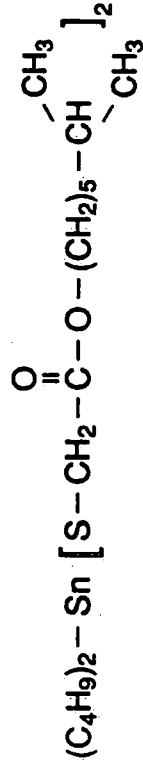
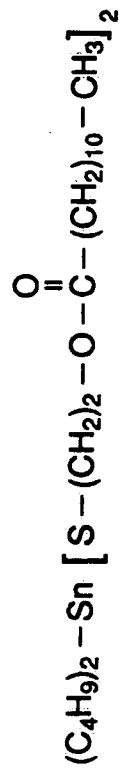
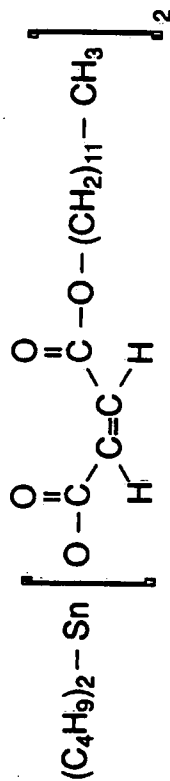


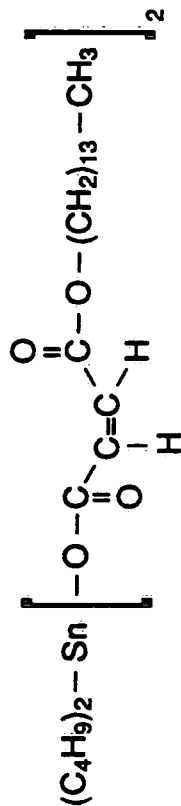
Figure 4b. Structures of di-n-butyltin compounds, cont'd: di-n-butyltin bis(2-mercaptoethyl laurate) (53), di-n-butyltin bis(lauryl maleate) (54), di-n-butyltin bis(tetradecyl maleate) (55), di-n-butyltin bis(2-oleoyloxyethylmercaptide) (56), di-n-butyltin bis(2-linoleoyloxyethylmercaptide) (57), di-n-butyltin bis(pentadecyl maleate) (58), and di-n-butyltin bis(2-decanoyloxyethylmercaptide) (59).



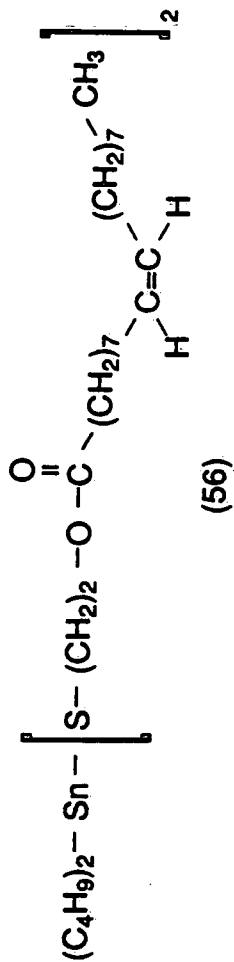
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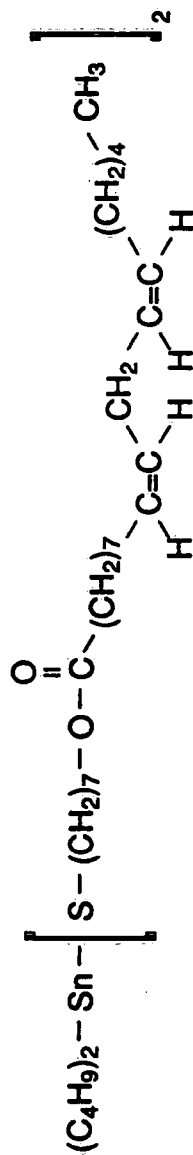
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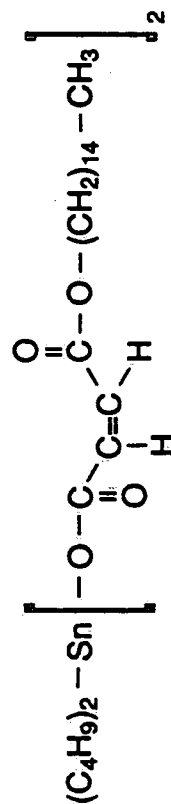
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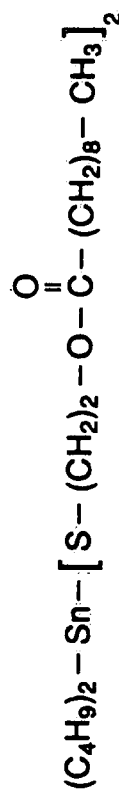
(56)



(57)

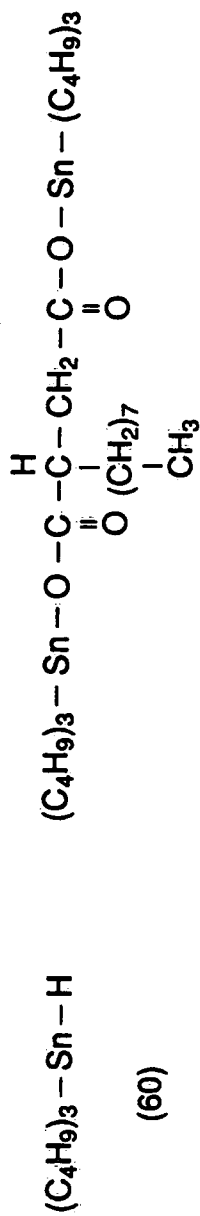


(58)



(59)

Figure 5. Structures of tri-n-butyltin hydride (60), stannane, [(2-octyl-1,4-dioxo-1,4-butanediyl) bis(oxy)]bis[tributyl- (61), n-octyltin tris(isooctyl mercaptoacetate) (62), and n-octyltin tris(2-ethylhexyl mercaptoacetate) (63).



(61)

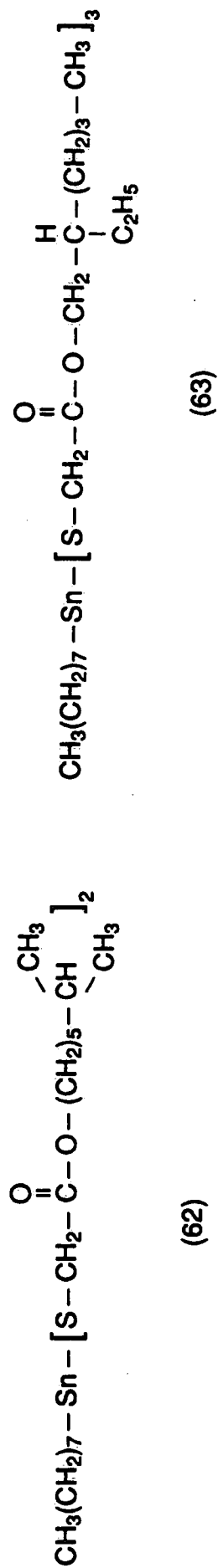
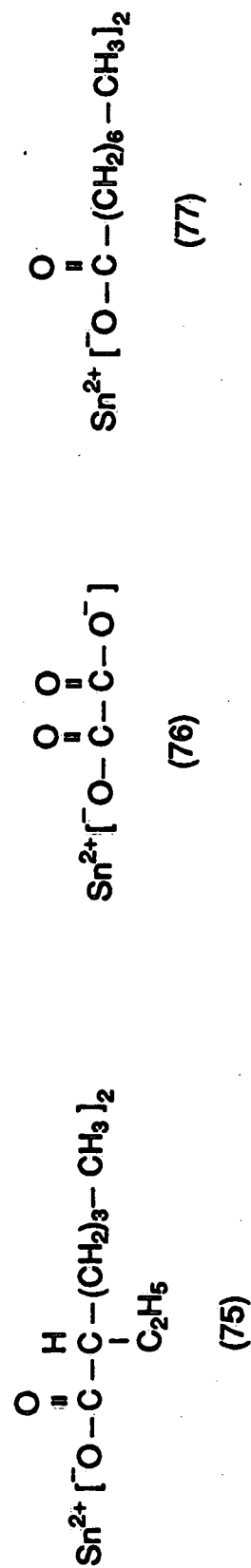
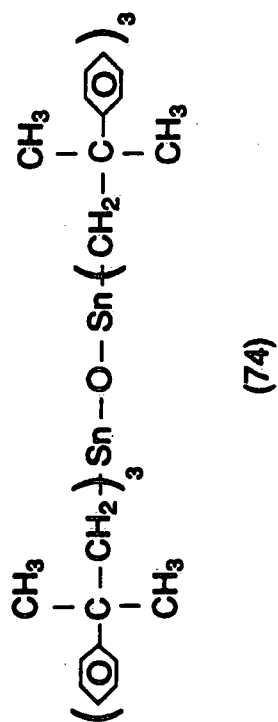
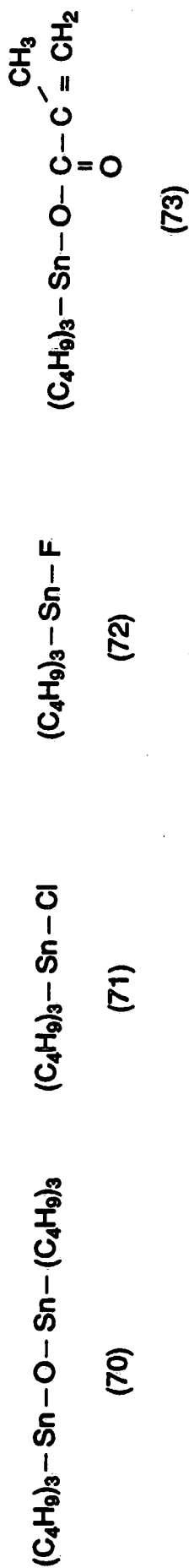


Figure 6. Structures of di-n-octyltin compounds: di-n-octyltin dilaurate (64), di-n-octyltin bis(2-ethylhexyl mercaptoacetate) (65), di-n-octyltin maleate (66), di-n-octyltin bis(laurylmercaptide) (67), di-n-octyltin bis(isooctylmercaptoacetate) (68), and di-n-octyltin bis(neodecanoate) (69).

Figure 7. Structures of bis(tri-n-butyltin) oxide (70), tri-n-butyltin chloride (71), tri-n-butyltin fluoride (72), tri-n-butyltin methacrylate (73), hexakis(β , β -dimethylphenylethyl)stannoxane (74), stannous 2-ethylhexanoate (75), stannous oxalate (76), and stannous n-octanoate (77).



APPENDIX

**CHEMICAL ABSTRACTS SERVICE REGISTRY NUMBERS, MOLECULAR FORMULAS,
MOLECULAR WEIGHTS AND SYNONYMS OF CEPA NON-PESTICIDAL
ORGANOTIN COMPOUNDS**

Tables A1 - A77 give the Chemical Abstracts Service Registry Numbers, molecular formulas, molecular weights and synonyms for those organotin compounds identified in Table 1 of the review. Some physical and chemical data are given for a few of the chemicals, where these were available.

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- Mitchum, D.L. and Moore, T.D. (1969). Efficacy of di-n-butyltin oxide on an intestinal fluke, *Crepidostomum farionis*, in golden trout. *Progressive Fish Culturist* 31, 143-148.
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Table A1. Methyltin trichloride

Chemical Abstracts Service Registry Number 993-16-8

Molecular Formula $\text{CH}_3\text{Cl}_3\text{Sn}$

Molecular Weight 240.08

Synonyms:

stannane, trichloromethyl- (8CI,9CI)
methyltrichlorostannane
methyltrichlorotin
monomethyltin trichloride
stannane, trichloromethyl-
stannane, methyltrichloro-
tin, methyl, trichloride
tin, trichloromethyl-
trichloromethylstannane
trichloromethyltin

Table A2. Methyltin sulfide

Chemical Abstracts Service Registry Number 33397-79-4

Molecular Formula $\text{C}_2\text{H}_6\text{S}_3\text{Sn}_2$

Molecular Weight 363.64

Synonyms:

distannathiane, dimethyldithioxo-
distannathiane, 1,3-dimethyl-1,3-dithioxo-
tin, trithiobis[methyl-

Table A3. Methyltin tris(laurylmercaptide)

Chemical Abstracts Service Registry Number 52165-03-4

Molecular Formula $C_{37}H_{78}S_3Sn$

Molecular Weight 737.92

Synonyms:

stannane, tris(dodecylthio)methyl- (9CI)
methyltin tris(n-dodecylmercaptide)

Table A4. Methyltin tris(isooctyl mercaptoacetate)

Chemical Abstracts Service Registry Number 54849-38-6

Molecular Formula $C_{31}H_{60}O_6S_3Sn$

Molecular Weight 743.70

Synonyms:

acetic acid, 2,2',2''-[(methylstannylidyne)tris(thio)]tris-,
triisooctyl ester
monomethyltin tris(isooctyl thioglycolate)
monomethyltin tris(isooctyl mercaptoacetate)
methyltin S,S',S''-tris(isooctyl mercaptoacetate)
methyltin tris(isooctyl thioglycolate)

Table A5. Methyltin tris(2-ethylhexyl mercaptoacetate)

Chemical Abstracts Service Registry Number 57583-34-3

Molecular Formula $C_{31}H_{60}O_6S_3Sn$

Molecular Weight 743.70

Synonyms:

8-oxa-3,5-dithia-4-stannatetradecanoic acid, 10-ethyl-4-[[2-
[(2-ethylhexyl)oxy]-2-oxoethyl]thio]-4-methyl-7-oxo-,
2-ethylhexyl ester
methyltin tris(2-ethylhexyl thioglycolate)

Table A6. Methyltin tris(2-mercaptoethyl oleate)

Chemical Abstracts Service Registry Number 59118-79-5

Molecular Formula $C_{61}H_{114}O_6S_3Sn$

Molecular Weight 1158.47

Synonyms:

9-octadecenoic acid (Z)-, (methylstannylidyne)tris(thio-2,1-ethanediyl)ester
monomethyltin tris(2-mercaptoethyl oleate)

Table A7. Methyltin tris(lauroyloxyethylmercaptide)

Chemical Abstracts Service Registry Number 67859-62-5

Molecular Formula $C_{43}H_{84}O_6S_2Sn$

Molecular Weight 879.96

Synonyms:

dodecanoic acid, (methylstannylidyne)tris
(thio-2,1-ethanediyl) ester

Table A8. Methyltin tris(2-tetradecanoyloxyethylmercaptide)

Chemical Abstracts Service Registry Number 68928-38-1

Molecular Formula $C_{49}H_{96}O_6S_3Sn$

Molecular Weight 996.19

Synonyms:

tetradecanoic acid, (methylstannylidyne)tris(thio-2,1-ethanediyl) ester

Table A9. Methyltin tris(2-linoleoyloxyethylmercaptide)

Chemical Abstracts Service Registry Number 68928-40-5

Molecular Formula $C_{61}H_{108}O_6S_3Sn$

Molecular Weight 1152.42

Synonyms:

9,12-octadecadienoic acid (Z,Z)-, (methylstannylidyne)tris(thio-2,1-ethanediyl) ester

Table A10. Methyltin tris(decyloyloxyethylmercaptide)

Chemical Abstracts Service Registry Number 68928-50-7

Molecular Formula $C_{37}H_{72}O_6S_3Sn$

Molecular Weight 827.86

Synonyms:

decanoic acid, (methylstannylidyne)tris
(thio-2,1-ethanediyl) ester

Table A11. Dimethyltin dichloride

Chemical Abstracts Service Registry Number 753-73-1

Molecular Formula $C_2H_6Cl_2Sn$

Molecular Weight 219.67

Synonyms:

stannane, dichlorodimethyl- (8CI,9CI)
dichlorodimethylstannane
dichlorodimethyltin
dimethyldichlorostannane
dimethyldichlorotin
tin, dimethyl-, dichloride

Properties:

solubility in seawater 20 g/L (Blunden and Chapman, 1986)

Table A12. Dimethyltin sulfide

Chemical Abstracts Service Registry Number 13269-74-4

Molecular Formula C_2H_6SSn

Molecular Weight 180.82

Synonyms:

stannane, dimethylthioxo-
tin, dimethylthioxo-

Table A13. Dimethyltin bis(isooctyl mercaptoacetate)

 Chemical Abstracts Service Registry Number 26636-01-1
Molecular Formula $C_{22}H_{44}O_4S_2Sn$

Molecular Weight 555.41

Synonyms:

acetic acid, 2,2'-[(dimethylstannylene)bis(thio)]bis-, diisooctyl ester

Advastab TM 181

diisooctyl [(dimethylstannylene)dithio]diacetate

dimethyltin bis(isooctyl thioglycolate)

dimethyltin S,S'-bis(isooctyl mercaptoacetate)

bis[[[(isooctyloxy)carbonyl]methyl]thio]dimethyltin

T 40 (ester)

Table A14. Dimethyltin bis(isooctyl mercaptopropionate)

 Chemical Abstracts Service Registry Number 42378-34-7
Molecular Formula $C_{24}H_{47}O_4S_2Sn$

Molecular Weight 582.46

Synonyms:

propanoic acid, 3,3'-[(dimethylstannylene)bis(thio)]bis-, diisooctyl ester (9CI)

dimethyltin bis(isooctyl 3-mercaptopropionate)

Table A15. 1,1,3,3,-Tetramethyl-1,3-bis(oleoyloxy)distannoxane

 Chemical Abstracts Service Registry Number 43136-18-1
Molecular Formula $C_{40}H_{78}O_5Sn_2$

Molecular Weight 876.44

Synonyms:

distannoxane, 1,1,3,3-tetramethyl-1,3-bis[(1-oxo-9-octadecenyl)oxy]-, (Z,Z)- (9CI)

1,3-bis(oleoyloxy)-1,1,3,3-tetramethyldistannoxane

Table A16. Dimethyltin bis(laurylmercaptide)

Chemical Abstracts Service Registry Number 51287-84-4

Molecular Formula $C_{26}H_{56}S_2Sn$

Molecular Weight 551.55

Synonyms:

stannane, bis(dodecylthio)dimethyl- (9CI)
 bis(dodecylthio)dimethylstannane
 dimethyltin bis(dodecyl mercaptide)
 dimethyltin S,S'-bis(dodecyl mercaptide)
 dimethyltin, bis(dodecylmercaptide)-

Table A17. Dimethyltin bis(2-ethylhexyl mercaptopropionate)

Chemical Abstracts Service Registry Number 57057-50-8

Molecular Formula $C_{26}H_{47}O_4S_2Sn$

Molecular Weight 582.46

Synonyms:

10-oxa-4,6-dithia-5-stannahexadecanoic acid, 12-ethyl-5,5-dimethyl-9-oxo, 2-ethylhexyl ester (9CI)
 dimethyltin bis(2-ethylhexyl β -mercaptopropionate)
 dimethyltin bis(2-ethylhexyl 3-mercaptopropionate)

Table A18. Dimethyltin bis(2-ethylhexyl mercaptoacetate)

Chemical Abstracts Service Registry Number 57583-35-4

Molecular Formula $C_{22}H_{44}O_4S_2Sn$

Molecular Weight 555.41

Synonyms:

8-oxa-3,5-dithia-4-stannatetradecanoic acid, 10-ethyl-4,4-dimethyl-7-oxo-, 2-ethylhexyl ester
 methyltin tris(2-ethylhexyl thioglycolate)

Table A19. Dimethyltin bis(2-oleoyloxyethylmercaptide)

Chemical Abstracts Service Registry Number 67859-63-6

Molecular Formula $C_{42}H_{80}O_4S_2Sn$

Molecular Weight 831.92

Synonyms:

9-octadecenoic acid (Z)-, (dimethylstannylene)bis(thio-2,1-ethanediyl) ester (9CI)

Table A20. Dimethyltin bis(2-linoleoyloxyethylmercaptide)

Chemical Abstracts Service Registry Number 67859-64-7

Molecular Formula $C_{42}H_{76}O_4S_2Sn$

Molecular Weight 827.89

Synonyms:

9,12-octadecadienoic acid (Z,Z)-, (dimethylstannylene) bis(thio-2,1-ethanediyl) ester

Table A21. Dimethyltin bis(decyloxyethylmercaptide)

Chemical Abstracts Service Registry Number 67874-41-3

Molecular Formula $C_{22}H_{52}O_4S_2Sn$

Molecular Weight 563.48

Synonyms:

decanoic acid, (dimethylstannylene)bis(thio-2,1-ethanediyl) ester

Table A22. Dimethyltin bis(lauroyloxyethylmercaptide)

Chemical Abstracts Service Registry Number 68928-42-7

Molecular Formula $C_{30}H_{60}O_4S_2Sn$

Molecular Weight 667.63

Synonyms:

dodecanoic acid, (dimethylstannylene)bis
(thio-2,1-ethanediyl) ester

Table A23. Dimethyltin bis(neodecanoate)

Chemical Abstracts Service Registry Number 68928-76-7

Molecular Formula $C_{22}H_{44}O_4Sn$

Molecular Weight 491.28

Synonyms:

stannane, dimethylbis[(1-oxoneodecyl)oxy]-

Table A24. n-Butyltin trichloride

Chemical Abstracts Service Registry Number 1118-46-3

Molecular Formula $C_4H_9Cl_3Sn$

Molecular Weight 282.17

Synonyms:

stannane, butyltrichloro- (8CI,9CI)
butyltrichlorostannane
butyltrichlorotin
monobutyltin trichloride
stannane, trichlorobutyl-
tin, n-butyl-, trichloride
trichlorobutylstannane
trichlorobutyltin

Uses:

Glass coating for scratch resistance. In the process of heating after deposition on the glass, it is converted to a thin layer of SnO_2 .

Table A25. n-Butylstannoic acid

Chemical Abstracts Service Registry Number 2273-43-0

Molecular Formula $C_4H_{10}O_2Sn$

Molecular Weight 208.81

Synonyms:

stannane, butylhydroxyoxo-
1-butane stannoic acid
butane stannoic acid
butylstannonic acid
butyltin hydroxide oxide
butylhydroxytin oxide
n-butyltin hydroxide oxide

Table A26. n-Butylchlorotin dihydroxide

Chemical Abstracts Service Registry Number 13355-96-9

Molecular Formula $C_4H_{11}ClO_2Sn$

Molecular Weight 245.27

Synonyms:

stannane, butylchlorodihydroxy-

Table A27. n-Butyltin sulfide

Chemical Abstracts Service Registry Number 15666-29-2

Molecular Formula $C_8H_{18}S_3Sn_2$

Molecular Weight 457.97

Synonyms:

distannathiane, dibutyldithio-

monobutyltin sulfide

distannathiane, 1,3-dibutyl-1,3-dithio-
butyl thiostannoic anhydride

Table A28. n-Butyltin tris(2-ethylhexanoate)

Chemical Abstracts Service Registry Number 23850-94-4

Molecular Formula $C_{28}H_{54}O_6Sn$

Molecular Weight 605.43

Synonyms:

stannane, butyltris[(2-ethyl-1-oxohexyl)oxy]-

stannane, butyltris[(2-ethylhexanoyl)oxy]-

monobutyltin tris(2-ethylhexanoate)

Table A29. n-Butyltin tris(isooctyl mercaptoacetate)

Chemical Abstracts Service Registry Number 25852-70-4

Molecular Formula $C_{34}H_{66}O_6S_3Sn$

Molecular Weight 785.78

Synonyms:

acetic acid, 2,2',2''-[(butylstannylidyne)tris(thio)]tris,
triisooctyl ester
acetic acid, [(butylstannylidyne)trithio]tri-,
triisooctyl ester
butyltin tris(isooctyl thioglycolate)
butylstannane tris(isooctyl mercaptoacetate)
monobutyltin tris(isooctyl thioglycolate)

Table A30. n-Butylthiostannonic acid

Chemical Abstracts Service Registry Number 26410-42-4

Molecular Formula $C_4H_{10}OSSn$

Molecular Weight 224.88

Synonyms:

stannane, butylmercaptooxo- (8CI)

Table A31. n-Butyltin tris(2-ethylhexyl mercaptoacetate)

Chemical Abstracts Service Registry Number 26864-37-9

Molecular Formula $C_{34}H_{66}O_6S_3Sn$

Molecular Weight 785.78

Synonyms:

8-oxa-3,5-dithia-4-stannatetradecanoic acid, 4-butyl-10-ethyl-4-[[2-[(2-ethylhexyl)oxy]-2-oxoethyl]thio]-7-oxo-, 2-ethylhexyl ester (9CI)
 [(butylstannylidyne)trithio]triacetic acid, tris(2-ethylhexyl) ester
 acetic acid, [(butylstannylidyne)trithio]tri-, tris(2-ethylhexyl) ester
 monobutyltin tris(2-ethylhexyl thioacetate)
 monobutyltin tris(2-ethylhexyl thioglycolate)
 tris(2-ethylhexyl) 2,2',2''-[(butylstannylidyne)tris(thio)]tris-[acetate]

Table A32. n-Butyltin tris(isooctyl mercaptopropionate)

Chemical Abstracts Service Registry Number 36118-60-2

Molecular Formula $C_{37}H_{72}O_6S_3Sn$

Molecular Weight 827.86

Synonyms:

propanoic acid, 3,3',3''-[(butylstannylidyne)tris(thio)]tris-, triisooctyl ester (9CI)
 butyltin tris(isooctyl 3-mercaptopropionate)
 monobutyltin tris(isooctyl 3-mercaptopropionate)

Table A33. n-Butyltin tris(2-oleoyloxyethylmercaptide)

Chemical Abstracts Service Registry Number 67361-76-6

Molecular Formula $C_{64}H_{120}O_6S_3Sn$

Molecular Weight 1200.55

Synonyms:

9-octadecenoic acid (Z)-, (butylstannylidyne)tris
(thio-2,1-ethanediyl) ester (9CI)

Table A34. n-Butyltin tris(2-decanoyloxyethylmercaptide)

Chemical Abstracts Service Registry Number 67874-51-5

Molecular Formula $C_{37}H_{78}O_6S_3Sn$

Molecular Weight 833.91

Synonyms:

decanoic acid, (dimethylstannylidyne)tris
(thio-2,1-ethanediyl) ester (9CI)

Table A35. n-Butyltin tris(2-tetradecanoyloxyethylmercaptide)

Chemical Abstracts Service Registry Number 68928-34-7

Molecular Formula $C_{52}H_{102}O_6S_3Sn$

Molecular Weight 1038.27

Synonyms:

tetradecanoic acid, (butylstannylidyne)tris
(thio-2,1-ethanediyl) ester (9CI)

Table A36. n-Butyltin tris(2-linoleoyloxyethylmercaptide)

Chemical Abstracts Service Registry Number 68928-37-0

Molecular Formula $C_{64}H_{114}O_6S_3Sn$

Molecular Weight 1194.50

Synonyms:

9,12-octadecadienoic acid (Z,Z)-, (butylstannylidyne)tris
(thio-2,1-ethanediyl) ester

Table A37. n-Butyltin tris(2-dodecanoyloxyethylmercaptide)

Chemical Abstracts Service Registry Number 68928-52-9

Molecular Formula $C_{46}H_{90}O_6S_3Sn$

Molecular Weight 954.11

Synonyms:

dodecanoic acid, (butylstannylidyne)tris
(thio-2,1-ethanediyl) ester (9CI)

Table A38. Di-n-butyltin dilaurate

Chemical Abstracts Service Registry Number 77-58-7

Molecular Formula $C_{32}H_{64}O_4Sn$

Molecular Weight 631.55

Synonyms:

stannane, dibutylbis[(1-oxododecyl)oxy]-
butynorate
DBTL
dibutylbis(lauroyloxy)tin
stabilizer D 22
tin dibutyl dilaurate

Trade Names (Martec, 1979b; World Health Organization, 1980):

Advastab DBTL
Butynorate
Clear 1
Davainex
DBTDL
DBTL
Ferro 820
Mark 1038
Niax D-22
Stabilizer D-22
Stanclere DBTL
Stavinor 1200 SN
T 12 (catalyst); T 12 (VAN)
Thermolite 12
Tinostat
TVS Tin Lau
Wormal CC-1

Properties (from Martec, 1979b unless otherwise indicated):

soft crystals or oily liquid depending upon the type and purity of lauric acid used in preparation; yellow colour; combustible soluble in benzene and acetone, but not in water or methanol. solubility in seawater 6 mg/L (Blunden and Chapman, 1986)
m.p. 22-27 °C at 760 mm Hg
b.p. 205°C at 10 mm Hg
s.g. 1.04 at 25°C
viscosity 42 centipoise at 25°C
flash point (open cup) 226.7°C
refractive index 1.471 at 20°C
log K_{ow} 5.0-6.5 (est.) (Enviro Control, 1980)
v.p. 7.5×10^{-4} mm (Enviro Control, 1980)

Uses:

Anthelmintic and growth stimulant for poultry, toxicidiostat for young turkeys, catalyst for room temperature vulcanization of silicone elastomers, and catalyst in polyurethane foams (polyester type) (Martec, 1979a; Enviro Control, 1980; Epstein et al., 1991). In 1976 the total production of dibutyltin dilaurate was 18% of the total U.S. production of alkyltin compounds (Enviro Control, Inc. 1980). Production was predicted to rise to 8 million pounds by 1986. Ninety-three per cent of dibutyltin dilaurate is used in catalytic applications. Organ distribution and clearance times given for poultry (Enviro Control, 1980).

Table A39. Di-n-butyltin maleate

Chemical Abstracts Service Registry Number 78-04-6

Molecular Formula $C_{12}H_{20}O_4Sn$

Molecular Weight 346.98

Synonyms:

1,3,2-dioxastannepin-4,7-dione, 2,2-dibutyl- (8CI,9CI)

Advastab DBTM

Advastab T 290

Advastab T 340

dibutyl(maleoyldioxy)tin

dibutylstannylene maleate

Irgastab T 290

Nuodex V 1525

Stavinor 1300 SN

tin, dibutyl(maleoyldioxy)-

Table A40. Di-n-butyltin oxide

Chemical Abstracts Service Registry Number 818-08-6

Molecular Formula $C_8H_{18}OSn$

Molecular Weight 248.92

Synonyms:

stannane, dibutyloxo-
dibutyloxostannane
dibutylstannane oxide
dibutyloxotin

Properties (from Martec, 1979b unless otherwise indicated):

white, amorphous powder, pentacoordinate structure based on Mossbauer studies; combustible

"insoluble in any solvents"

m.p. 110-115 °C at 760 mm Hg

decomposition temperature 110-115 °C at 760 mm Hg

s.g. 1.58 at 25 °C

Uses:

Precursor for dibutyltin dilaurate and esterification catalyst (Martec, 1979a). Dibutyltin oxide has been used for the treatment of intestinal flukes in trout (Mitchum and Moore, 1969).

Table A41. Di-n-butyltin diacetate

Chemical Abstracts Service Registry Number 1067-33-0

Molecular Formula $C_{12}H_{24}O_4Sn$

Molecular Weight 351.01

Synonyms:

stannane, bis(acetyloxy)dibutyl-
diacetoxydibutyltin
stannane, diacetoxydibutyl-
Ba 2726
diacetoxydibutylstannane
T 1 (catalyst)
T 1

Properties (from Martec, 1979b unless otherwise indicated):

clear, colourless liquid, combustible, slight acetic acid odour
m.p. 10°C at 760 mm Hg
b.p. 142-145 °C at 10 mm Hg
decomposition temperature 142-145°C at 760 mm Hg
s.g. 1.32 at 25°C
flash point (open cup) 143°C
refractive index 1.482 at 20°C.
solubility in seawater 6 mg/L (Blunden and Chapman, 1986)

Uses:

Catalyst for flexible foams (Martec, 1979a)

Table A42. Di-n-butyltin bis(laurylmercaptide)

Chemical Abstracts Service Registry Number 1185-81-5

Molecular Formula $C_{32}H_{68}S_2Sn$

Molecular Weight 635.72

Synonyms:

stannane, dibutylbis(dodecylthio)-
dibutylbis(dodecylthio)tin
Mellite 39
dibutyltin bis(dodecyl mercaptide)
bis(dodecylthio)dibutyltin
dibutylbis(dodecylthio)stannane
Advastab TM 918

Trade Names (Martec, 1979b; World Health Organization, 1980):

Advastab TM-918, TM-98
CC-10
Ferro 822
Mark A
Mellite 39, 139
Thermolite 20

Properties (from Martec, 1979b unless otherwise indicated):

liquid, clear pale yellow colour
soluble in toluene and heptane
s.g. 1.006 at 25°C
viscosity 22 centipoises at 25°C
refractive index 1.498 at 25°C

Properties:

Feed additive and polyurethane foam catalyst (Martec, 1979a).

Table A43. Di-n-butyltin bis(2-ethylhexanoate)

Chemical Abstracts Service Registry Number 2781-10-4

Molecular Formula $C_{24}H_{48}O_4Sn$

Molecular Weight 519.34

Synonyms:

stannane, dibutylbis[(2-ethyl-1-oxohexyl)oxy]- (9CI)
 stannane, dibutylbis[(2-ethylhexanoyl)oxy]- (8CI)
 di-n-butyltin di-2-ethylhexanoate
 dibutyltin bis(α -ethylhexanoate)
 dibutyltin bis(2-ethylhexoate)
 dibutyltin di-2-ethylhexanoate
 stannane, bis[(2-ethylhexanoyl)oxy]dibutyl-
 tin, dibutyl-, di(2-ethylhexoate)
 2-ethylhexanoic acid, dibutyltin salt

Table A44. Di-n-butyltin sulfide

Chemical Abstracts Service Registry Number 4253-22-9

Molecular Formula $C_8H_{18}SSn$

Molecular Weight 264.99

Synonyms:

stannane, dibutylthioxo-
 tin dibutyl mercaptide
 tin, dibutylthioxo-
 BTS 71
 BTS 80
 Stann BK 1100L

Table A45. Di-n-butyltin distearate

Chemical Abstracts Service Registry Number 5847-55-2

Molecular Formula $C_{44}H_{88}O_4Sn$

Molecular Weight 799.88

Synonyms:

stannane, dibutylbis[(1-oxooctadecyl)oxy]-
 stannane, dibutylbis(stearoyloxy)-
 tin, dibutylbis(stearoyloxy)-
 dibutyltin stearate

Table A46. Di-n-butyltin bis(methyl maleate)

Chemical Abstracts Service Registry Number 15546-11-9

Molecular Formula $C_{18}H_{28}O_8Sn$

Molecular Weight 499.11

Synonyms:

5,7,12-trioxa-6-stannatrideca-2,9-dienoic acid,
 6,6-dibutyl-4,8,11-trioxo-, methyl ester, (Z,Z)- (9CI)
 dibutyltin (methyl maleate)
 dibutyltin bis(monomethyl maleate)
 Stan-Guard 156
 stannane, bis(methoxymaleoyloxy)dibutyl-
 stannane, dibutyl bis(methyl maleate)
 stannane, dibutylbis[(3-carboxyacryloyl)oxy]-,
 dimethyl ester, (Z,Z)-

Table A47. Di-n-butyltin bis(2-ethylhexyl maleate)

 Chemical Abstracts Service Registry Number 15546-12-0
Molecular Formula $C_{32}H_{56}O_8Sn$

Molecular Weight 687.49

Synonyms:

5,7,12-trioxa-6-stannaoctadeca-2,9-dienoic acid,
 6,6-dibutyl-14-ethyl-4,8,11-trioxo-, 2-ethylhexyl
 ester, (Z,Z)- (9CI)
 stannane, dibutylbis[(3-carboxyacryloyl)oxy]-,
 bis(2-ethylhexyl) ester, (Z,Z)- (8CI)
 bis[(2-ethylhexyloxy)maleoyloxy]di(n-butyl)stannane
 dibutyltin bis(mono-2-ethylhexyl maleate)
 dibutyltin di(2-ethylhexyl maleate)
 stannane, bis[(2-ethylhexyloxy)maleoloxy]dibutyl-
 tin, bis(hydrogen maleato)dibutyl-,
 bis(2-ethylhexyl ester)

Table A48. Di-n-butyltin bis(n-butyl maleate)

 Chemical Abstracts Service Registry Number 15546-16-4
Molecular Formula $C_{24}H_{40}O_8Sn$

Molecular Weight 575.27

Synonyms:

5,7,12-trioxa-6-stannahehexadeca-2,9-dienoic acid,
 6,6-dibutyl-4,8,11-trioxo, butyl ester, (Z,Z)-
 stannane, dibutylbis[(3-carboxyacryloyl)oxy],
 dibutyl ester, (Z,Z)-
 dibutyltin bis(monobutyl maleate)
 maleic acid dibutyltin salt (2:1) dibutyl ester
 B 5 (stabilizer)
 BT 53A

Table A49. Di-n-butyltin bis(acetylacetonate)

Chemical Abstracts Service Registry Number 22673-19-4

Molecular Formula $C_{18}H_{32}O_4Sn$

Molecular Weight 431.14

Synonyms:

tin, dibutylbis(2,4-pentanedionate-O,O')-

Table A50. Di-n-butyltin bis(isooctyl maleate)

Chemical Abstracts Service Registry Number 25168-21-2

Molecular Formula $C_{32}H_{56}O_8Sn$

Molecular Weight 687.49

Synonyms:

2-butenic acid, 4,4'-[(dibutylstannylene)bis(oxy)]bis[4-oxo-,
diisooctyl ester, (Z,Z)-
stannane, dibutylbis[(3-carboxyacryloyl)oxy]-,
diisooctyl ester, (Z,Z)-

Table A51. Di-n-butyltin bis(isooctyl mercaptoacetate)

Chemical Abstracts Service Registry Number 25168-24-5

Molecular Formula $C_{28}H_{56}O_4S_2Sn$

Molecular Weight 639.57

Synonyms:

acetic acid, 2,2'-[(dibutylstannylene)bis(thio)] bis-,
 diisooctyl ester
 acetic acid, [(dibutylstannylene)dithio]di-,
 diisooctyl ester
 dibutyltin S,S'-bis(isooctyl mercaptoacetate)
 diisooctyl [(dibutylstannylene)dithio]diacetate
 Thermolite 31
 dibutyltin bis(isooctyl thioglycolate)

Trade Names (Martec, 1979a; World Health Organization, 1980):

Advastab TM-180, TM-220
 BTS 70
 Ferro 803, 807, 814, 820, 832, 835, 837, 840, 871, 873, 876A,
 877
 Irgastab 17M
 Mark 292, 534B, 649A, No. 11, CC-54, CC-78
 Nuostabe V-1562, V-1902
 T 101, 101 (accelerator)
 Thermolite 31, 66, 73, 310

Uses:

Stabilizer for PVC used in siding, eavestroughs and soffits
 (Martec, 1979b)

Table A52. Di-n-butyltin bis(isooctyl mercaptopropionate)

Chemical Abstracts Service Registry Number 26761-46-6

Molecular Formula $C_{30}H_{60}O_4S_2Sn$

Molecular Weight 667.63

Synonyms:

propanoic acid, 3,3'-[(dibutylstannylene)bis(thio)]bis-,
 diisooctyl ester (9CI)
 dibutylbis[[(isooctyloxy)carbonyl]ethyl]thio]tin
 dibutyltin bis(isooctyl β -mercaptopropionate)
 dibutyltin bis(isooctyl 3-mercaptopropionate)
 propionic acid, 3,3'-[(dibutylstannylene)dithio]di-,
 diisooctyl ester

Table A53. Di-n-butyltin bis(2-mercaptoethyl laurate)

Chemical Abstracts Service Registry Number 28570-24-3

Molecular Formula $C_{36}H_{52}O_4S_2Sn$

Molecular Weight 731.63

Synonyms:

lauric acid, diester with 2,2'-[(dibutylstannylene)di-
 thio]diethanol (8CI)

Table A54. Di-n-butyltin bis(lauryl maleate)

Chemical Abstracts Service Registry Number 33466-31-8

Molecular Formula $C_{40}H_{72}O_8Sn$

Molecular Weight 799.71

Synonyms:

5,7,12-trioxa-6-stannatetracos-2,9-dienoic acid,
6,6-dibutyl-4,8,11-trioxo-, dodecyl ester, (Z,Z)-
stannane, dibutylbis[(3-carboxyacryloyl)oxy]-,
didodecyl ester, (Z)-
dibutyltin bis(monolauryl maleate)
dibutyltin bis(dodecyl maleate)

Table A55. Di-n-butyltin bis(tetradecyl maleate)

Chemical Abstracts Service Registry Number 60659-60-1

Molecular Formula $C_{44}H_{80}O_8Sn$

Molecular Weight 885.81

Synonyms:

5,7,12-trioxa-6-stannahexacos-2,9-dienoic acid,
6,6-dibutyl-4,8,11-trioxo-, tetradecyl ester, (Z,Z)-

Table A56. Di-n-butyltin bis(2-oleoyloxyethylmercaptide)

Chemical Abstracts Service Registry Number 67361-77-7

Molecular Formula $C_{48}H_{92}O_4S_2Sn$

Molecular Weight 916.08

Synonyms:

9-octadecenoic acid (Z)-, (dibutylstannylene)
bis(thio-2,1-ethanediyl) ester (9CI)

Table A57. Di-n-butyltin bis(2-linoleoyloxyethylmercaptide)

Chemical Abstracts Service Registry Number 67859-61-4

Molecular Formula $C_{48}H_{88}O_4S_2Sn$

Molecular Weight 912.05

Synonyms:

9,12-octadecadienoic acid (Z,Z)-, (dibutylstannylene)
bis(thio-2,1-ethanediyl) ester

Table A58. Di-n-butyltin bis(pentadecyl maleate)

Chemical Abstracts Service Registry Number 68299-23-0

Molecular Formula $C_{46}H_{84}O_8Sn$

Molecular Weight 883.87

Synonyms:

2-butenic acid, 4,4'-[(dibutylstannylene)
bis(oxy)]bis[4-oxo-, diisopentadecyl ester, (Z)- (9CI)

Table A59. Di-n-butyltin bis(2-decanoyloxyethylmercaptide)

Chemical Abstracts Service Registry Number 68928-47-2

Molecular Formula $C_{32}H_{64}O_4S_2Sn$

Molecular Weight 695.68

Synonyms:

decanoic acid, (dibutylstannylene)bis(thio-
2,1-ethanediyl) ester (9CI)

Table A60. Tri-n-butyltin hydride

Chemical Abstracts Service Registry Number 688-73-3

Molecular Formula $C_4H_{10}Sn$

Molecular Weight 176.81

Synonyms:

stannane, tributyl-
tributylstannic hydride
tributylstannane
tributylstannyl hydride
tri-n-butylstannane

Table A61. Stannane, [(2-octyl-1,4-dioxo-1,4-butanediyl)bis
(oxy)]bis[tributyl- (9CI)]

Chemical Abstracts Service Registry Number 67701-37-5

Molecular Formula $C_{36}H_{74}O_4Sn_2$

Molecular Weight 808.37

Table A62. n-Octyltin tris(isooctyl mercaptoacetate)

 Chemical Abstracts Service Registry Number 26401-86-5
Molecular Formula $C_{38}H_{71}O_6S_3Sn$

Molecular Weight 838.87

Synonyms:

acetic acid, 2,2',2''-[(octylstannylidyne)tris(thio)]tris-,
 triisooctyl ester (9CI)
 acetic acid, [(octylstannylidyne)trithio]tri-,
 triisooctyl ester
 monooctyltin tris(isooctyl mercaptoacetate)
 monooctyltin tris(isooctyl thioglycolate)
 octyltin tris(isooctyl thioglycolate)

Table A63. n-Octyltin tris(2-ethylhexyl mercaptoacetate)

 Chemical Abstracts Service Registry Number 27107-89-7
Molecular Formula $C_{38}H_{74}O_6S_3Sn$

Molecular Weight 841.89

Synonyms:

8-oxa-3,5-dithia-4-stannatetradecanoic acid, 10-ethyl-
 4-[[2-[(2-ethylhexyl)oxy]-2-oxoethyl]thio]-4-octyl-
 7-oxo, 2-ethylhexyl ester
 acetic acid, [(octylstannylidyne)trithio]tri-,
 tris(2-ethylhexyl) ester
 tris(2-ethylhexyl) [(octylstannylidyne)trithio] triacetate
 octyltin tris(2-ethylhexyl thioglycolate)

Table A64. Di-n-octyltin dilaurate

 Chemical Abstracts Service Registry Number 3648-18-8
Molecular Formula $C_{40}H_{80}O_4Sn$

Molecular Weight 743.77

Synonyms:

stannane, dioctylbis[(1-oxododecyl)oxy]- (9CI)
 bis(lauroyloxy)dioctylstannane
 dioctyldilauroyloxytin
 dioctyltin didodecanoate
 lauric acid, dioctyltin deriv.
 stannane, bis(dodecanoyloxy)dioctyl-
 stannane, bis(lauroyloxy)dioctyl-
 stannane, di(dodecanoyloxy)dioctyl-
 stannane, dioctylbis(lauroyloxy)-

Table A65. Di-n-octyltin bis(2-ethylhexyl mercaptoacetate)

Chemical Abstracts Service Registry Number 15571-58-1

Molecular Formula $C_{36}H_{72}O_4S_2Sn$

Molecular Weight 751.79

Synonyms:

8-oxa-3,5-dithia-4-stannatetradecanoic acid, 10-ethyl-
 4,4-dioctyl-7-oxo-, 2-ethylhexyl ester
 acetic acid, [(dioctylstannylene)dithio]di-,
 bis(2-ethylhexyl) ester
 bis(2-ethylhexyl) [(dioctylstannylene) dithio]diacetate
 dioctyltin bis(2-ethylhexyl thioglycolate)
 bis(2-ethylhexyl thioglycolato)dioctyltin
 Advastab 17MOK
 Advastab 17MOL
 17MOK

Table A66. Di-n-octyltin maleate

Chemical Abstracts Service Registry Number 16091-18-2

Molecular Formula $C_{20}H_{36}O_4Sn$

Molecular Weight 459.20

Synonyms:

1,3,2-dioxastannapin-4,7-dione, 2,2-dioctyl
Thermolite 813
Estabex U 18
dioctylstannylene maleate
Mellite 825
LIV 1176

Properties:

s.g. 1.33 at 25 °C

Uses:

Stabilizer for rigid PVC pipe for potable water supplies (Martec, 1979a)

Use in Canada in Food Packaging Materials

A person may sell food, other than milk, skim milk, partly skimmed milk, sterilized milk, malt beverages and carbonated non-alcoholic beverage products, in a package that has been manufactured from a polyvinyl chloride formulation containing one or both of the octyltin chemicals di(n-octyl)tin S,S'-bis(isooctylmercaptoacetate) and di(n-octyl)tin maleate-polymer if the proportion of such chemicals, either singly or in combination, does not exceed a total of 3 per cent of the resin, and the food in contact with the package contains not more than 1 part per million total octyltin (Canada Department of National Health and Welfare, 1975). Di(n-octyl)tin maleate polymer shall be the octyltin chemical made from di(n-octyl)tin dichloride and shall have the formula $((C_8H_{17})_2SnC_4H_2O_4)_n$ (where n is between 2 and 4 inclusive), and a saponification number of 225 and 255, and shall contain 25.2 to 26.6 per cent of tin.

Table A67. Di-n-octyltin bis(laurylmercaptide)

Chemical Abstracts Service Registry Number 22205-30-7

Molecular Formula $C_{40}H_{84}S_2Sn$

Molecular Weight 747.93

Synonyms:

stannane, bis(dodecylthio)dioctyl- (8CI,9CI)
bis(dodecylthio)dioctyltin
di-n-octyltin bis(dodecyl mercaptide)
dioctyltin bis(dodecyl mercaptide)
dioctyltin didodecylmercaptide
dioctyltin dilauryl mercaptan salt
tin, bis(mercapto)dioctyl-, bis(dodecyl) ester

Table A68. Di-n-octyltin bis(isooctyl mercaptoacetate)

Chemical Abstracts Service Registry Number 26401-97-8

Molecular Formula $C_{36}H_{72}O_4S_2Sn$

Molecular Weight 751.79

Synonyms:

acetic acid, 2,2'-[(dioctylstannylene)bis(thio)]bis-,
diisooctyl ester (9CI)

acetic acid, [(dioctylstannylene)dithio]di-,
diisooctyl ester (8CI)

Advastab 17mo

bis[[[(isooctyloxy)carbonyl]methyl]thio]dioctyltin

di-n-octyltin-S,S'-bis(isooctyl mercaptoacetate)

di-n-octyltin diisooctylthioglycolate

dioctyltin bis(diisooctyl thioglycolate)

Trade Names (Martec, 1979b and World Health Organization, 1980):

Irgastab 17 MOK

Mark OTM

Mellite 831C

Octyl 11

Thermolite 831

Properties (from Martec 1979b unless otherwise noted):

clear yellow liquid

soluble in esters, ethers, ketones, alcohols, aliphatic and
aromatic hydrocarbons, chlorinated hydrocarbons and other organic
solvents

s.g. 1.085 at 25°C

refractive index 1.5005 at 25°C

Spiliopoulos et al. (1989): show mass spectrum of dioctyltin
bis(isooctyl mercaptoacetate) at 240 °C by direct inlet mass
spectrometry.

Uses:

Use in Canada in Food Packaging Materials

A person may sell food, other than milk, skim milk,
partly skimmed milk, sterilized milk, malt beverages and carbonated
non-alcoholic beverage products, in a package that has been
manufactured from a polyvinyl chloride formulation containing one
or both of the octyltin chemicals di(n-octyl)tin S,S'-

bis(isooctylmercaptoacetate) and di(n-octyl)tin maleate-polymer if the proportion of such chemicals, either singly or in combination, does not exceed a total of 3 per cent of the resin, and the food in contact with the package contains not more than 1 part per million total octyltin (Canada Department of National Health and Welfare, 1975). Di(n-octyl)tin maleate polymer shall be the octyltin chemical made from di(n-octyl)tin dichloride and shall have the formula $((C_8H_{17})_2SnC_4H_2O_4)_n$ (where n is between 2 and 4 inclusive), and a saponification number of 225 and 255, and shall contain 25.2 to 26.6 per cent of tin.

Table A69. Di-n-octyltin bis(neodecanoate)

Chemical Abstracts Service Registry Number 68299-15-0

Molecular Formula $C_{36}H_{72}O_4Sn$

Molecular Weight 687.66

Synonyms:

stannane, dioctylbis[(1-oxoneodecyl)oxy]- (9CI)

Table A70. Bis(tri-n-butyltin) oxide

Chemical Abstracts Service Registry Number 56-35-9

Molecular Formula $C_{24}H_{54}OSn_2$

Molecular Weight 596.08

Synonyms:

distannoxane, hexabutyl

BioMeT TBTO

Butinox

oxybis[tributyltin]

6-oxa-5,7-distannaundecane, 5,5,7,7-tetrabutyl-

Lastanox T

Table A71. Tri-n-butyltin chloride

Chemical Abstracts Service Registry Number 1461-22-9

Molecular Formula $C_{12}H_{27}ClSn$

Molecular Weight 325.49

Synonyms:

stannane, tributylchloro-
tributylchlorotin
chlorotributylstannane
tributylstannyl chloride
monochlorotributyltin
chlorotributyltin

Table A72. Tri-n-butyltin fluoride

Chemical Abstracts Service Registry Number 1983-10-4

Molecular Formula $C_{12}H_{27}FSn$

Molecular Weight 309.04

Synonyms:

stannane, tributylfluoro-
tributylfluorostannane
fluorotributyltin
tri-n-butylstannyl fluoride
BioMet TBTF

Table A73. Tri-n-butyltin methacrylate

Chemical Abstracts Service Registry Number 2155-70-6

Molecular Formula $C_{16}H_{32}O_2Sn$

Molecular Weight 375.12

Synonyms:

stannane, tributyl[(2-methyl-1-oxo-2-propenyl)oxy]-
 stannane, tributyl(methacryloyloxy)-
 tributylstannyl methacrylate
 tributyl(methacryloxy)stannane
 tin tributylmethacrylate
 (methacryloyloxy)tributylstannane

Table A74. Bis(trineophyltin) oxide

Chemical Abstracts Service Registry Number 13356-08-6

Molecular Formula $C_{60}H_{78}OSn_2$

Molecular Weight 1052.67

Synonyms:

distannoxane, hexakis(2-methyl-2-phenylpropyl)- (9CI)
 distannoxane, hexakis(β,β -dimethylphenyl)- (8CI)
 Vendex
 bis[tris(β,β -dimethylphenethyl)tin]oxide
 bis[tris(2-methyl-2-phenylpropyl)tin]oxide
 di[tri-(2,2-dimethyl-2-phenylethyl)tin]oxide
 Fenbutatin oxide
 hexakis(β,β -dimethylphenethyl)distannoxane
 hexakis(2-methyl-2-phenylpropyl)distannoxane
 Neostannox

Table A75. Stannous 2-ethylhexanoate

Chemical Abstracts Service Registry Number 301-10-0

Molecular Formula $C_{16}H_{30}O_4Sn$

Molecular Weight 406.71

Synonyms:

hexanoic acid, 2-ethyl-, tin(2⁺) salt
stannous 2-ethylhexoate
tin(II) 2-ethylhexanoate
Nuocure 28
stannous octoate
tin octoate

Table A76. Stannous oxalate

Chemical Abstracts Service Registry Number 814-94-8

Molecular Formula C_2O_4Sn

Molecular Weight 206.71

Synonyms:

ethanedioic acid, tin(2⁺) salt (1:1) (9CI)
oxalic acid, tin(2⁺) salt (1:1) (8CI)
tin oxalate
tin(II) oxalate

Table A77. Stannous n-octanoate

Chemical Abstracts Service Registry Number 1912-83-0

Molecular Formula $C_{16}H_{30}O_4Sn$

Molecular Weight 406.71

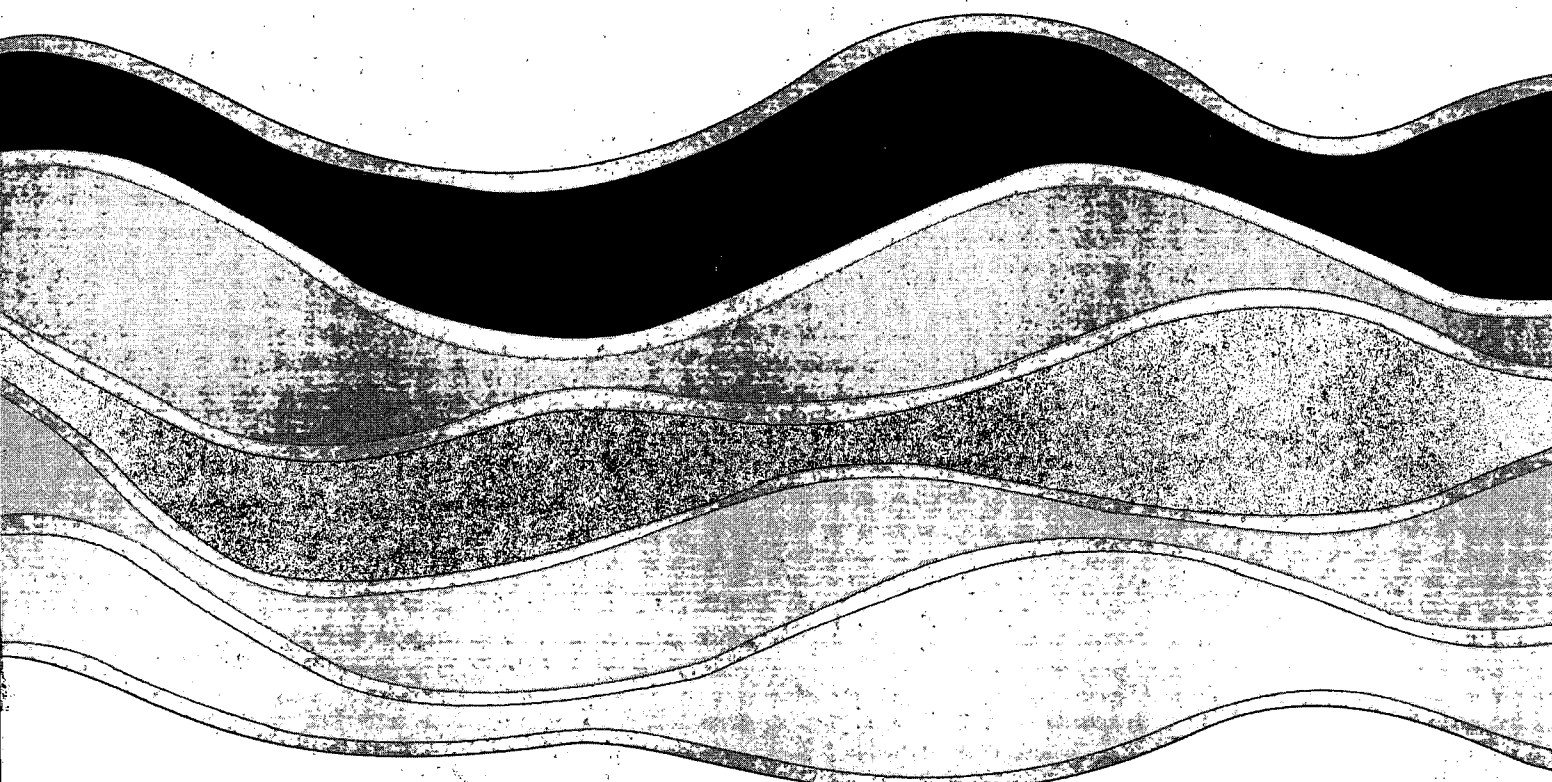
Synonyms:

octanoic acid, tin(2+) salt
stannous octanoate
tin octanoate
stannous dioctanoate
tin(II) octanoate
stannous caprylate

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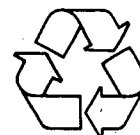
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