13. Tributyltin (TBT) antifoulants: a tale of ships, snails and imposex

David Santillo, Paul Johnston and William J. Langston

13.1. Introduction

There is little doubt that, without some form of control, the accumulation of marine fouling communities on vessels and man-made structures at sea increases drag and, in the case of vessels, fuel consumption, with substantial consequences in terms of economics and emissions. There is no doubt, also, that tributyltin (TBT) compounds (and the less widely used triphenyltins) are extremely effective and relatively economical as antifouling biocides, contributing to the rapid take-up of organotin-based paints by the shipping industry and small boat owners in the 1970s. These two arguments have formed the basis of the defence of TBT antifouling formulations since the first undesirable consequences of their use became apparent in the late 1970s, and indeed are still cited today by proponents of their continued suitability (for example, see Evans, 2000; Abel, 2000). What is missing from such evaluations, of course, is proper consideration not only of the quantifiable financial losses suffered by the aquaculture industry and imposed upon harbour authorities as a result of the widespread application of TBT paints, but also of the broader environmental 'costs' which led initially to restrictions on use and which now underlie the decision for a global phase-out.

The TBT story presents, in some ways, a rather unusual case study. Firstly, the widespread use of TBT paints is a relatively recent development, with its origins in the late 1960s. Secondly, initial concerns about adverse effects were raised during the period in which its use became most popular, concerns that led rapidly to some national and regional restrictions. Thirdly, the adverse effect most commonly associated with TBT, that of imposex in marine gastropod molluscs resulting from interference with steroid hormone metabolism, represented a highly sensitive, chemical-specific phenomenon. This factor contributed greatly to the early acceptance of a direct causal relationship and, in turn, of the need for controls. The paint manufacturers, in concert with other interested parties, mounted challenges to restrictions imposed in the 1980s, but the strength of the evidence for severe effects on biota, including regional extermination of some species, could hardly be denied.

The prohibition of the use of TBT paints on vessels under 25 metres in length, effective in France from 1982, in the United Kingdom from 1987 and more widely from the end of the 1980s and early 1990s, did much to improve the situation within marinas and sheltered harbours where use on leisure craft had predominated. Some regional recovery of affected mollusc populations has since been recorded. Through the late 1980s and 1990s, however, a picture of more widespread TBT contamination and population-level effects emerged, coincident with improved monitoring and understanding of the properties and environmental distribution of organotins. Vos *et al.* (2000) estimate that imposex has now been documented in the wild for as many as 150 species of marine prosobranch snails worldwide. Evidence relating the prevalence of imposex to density of shipping traffic, along with poor recovery of affected populations in some areas and widespread accumulation of butyltin residues in marine mammals, led to renewed calls for prohibitions to be extended to use on all vessels, irrespective of size. As we stand on the brink of just such a prohibition, developed under the auspices of the International Maritime Organization's Marine Environmental Protection Committee (MEPC), what can we learn from the process of its evolution?

13.2. The emergence of the TBT problem

Organotin compounds were first developed as moth-proofing agents in the 1920s, and only later used more widely as bactericides and fungicides (Moore *et al.*, 1991). Dibutyltin and tributyltin compounds have been produced since the late 1940s (Laughlin and Linden, 1985), although use of TBT in marine antifouling paints dates only from the 1960s (Balls, 1987; ten Hallers-Tjabbes, 1997) and then initially as a booster biocide in copper-based formulations. As a result of its superior effectiveness over copper (Wade *et al.*, 1988a), the use of TBT paints by private and commercial users accelerated greatly in the 1970s, during which these formulations captured a major proportion of the antifouling market (Evans, 2000). 'Free association' paints, which released the biocide rapidly at first but demanded frequent reapplication, were gradually replaced by 'self-polishing copolymer' formulations which ensured a more constant release of biocide and reduced repainting frequency.

At the same time as its explosive increase in use, the first observations were made of TBT effects on non-target organisms. While toxicity to fouling organisms was intentional, its propensity for wider impacts on the marine environment had been grossly underestimated. An early focus on acute effects, especially mortality (Laughlin and Linden, 1987), failed to identify sub-lethal consequences of prolonged exposure in some taxa. For example, imposex (the development of male sexual structures in females) can be initiated in some gastropod molluscs by TBT in the low ng/l (parts per trillion) range (Bryan *et al.*, 1986; Alzieu, 1998), concentrations also known to cause shell deformity and larval mortality (Alzieu *et al.*, 1986).

The phenomenon of imposex was first described by Smith (1971), from studies of the American mud-snail (*Nassarius obsoletus*) in the vicinity of harbours on the US east coast. Around the same time, Blaber (1970) recorded the appearance of a penis in female dogwhelks (*Nucella lapillus*) in Plymouth Sound, United Kingdom, at much greater prevalence close to the harbour than further away. Despite the severity of the phenomenon, however, the causal agent remained unknown. Only as analytical capabilities improved in the late 1970s and early 1980s was the connection with shipping made and the extent of damage already done recognised.

Two regional case histories were instrumental in identifying low-dose effects of TBT and initiating development of the first regional controls; the collapse of the shellfish industry in Arcachon Bay (Atlantic coast of France) and the reporting of widespread imposex in dogwhelks from southern UK coastal waters.

13.3. Arcachon Bay

Until the mid-1970s, Arcachon Bay had been an important area for oyster (*Crassostrea gigas*) culture, with production of 10 000–15 000 tonnes per year (Evans, 2000) covering substantial areas of the tidal mud flats. The bay was also popular with leisure craft, with vessel numbers increasing from 7 500 in the mid-1970s to 15 000 at the start of the 1980s. Estimated inputs of TBT to the bay peaked at around 8 kg per day (Ruiz *et al.*, 1996).

Imposex was first observed in the bay in 1970, affecting the predatory gastropod *Ocenebra erinacea* (oyster drill), leading rapidly to its near extirpation from the bay (Gibbs, 1993). TBT was identified as the responsible agent only in the early 1980s.

Had the adverse effects been limited to the loss of this species, considered a pest within the shellfish industry for its damage to oyster stocks, little if any action may have followed. However, this early warning was quickly followed by failure of the oyster stocks themselves. Despite a normal spawning event in summer 1976, few of the larvae survived. Larval settlement largely failed through the late 1970s and into the 1980s, resulting in massive financial losses by the shellfish industry. By 1981, annual oyster production had fallen to only 3 000 tonnes (Ruiz *et al.*, 1996). In addition to reproductive failure, adult oysters were rendered unsaleable by shell deformation leading, in severe cases, to 'ball-shaped' specimens (Alzieu *et al.*, 1989).

Such observations pre-dated analytical techniques sensitive enough to describe in detail environmental distributions of TBT. Alzieu *et al.* (1986) provided the first reliable survey of organotins in the waters of Arcachon Bay, while sediment data were not available until the 1990s (Sarradin *et al.*, 1994). Nevertheless, the severity of impacts on the ecology of Arcachon Bay, manifest in heavy financial losses, was sufficient to stimulate relatively swift action by the French government. Acting on the best information available linking the oyster collapse to the presence of TBT, France was the first country to introduce legislation prohibiting the application of TBT paints to small (< 25 metre) vessels, in 1982 (Michel and Averty, 1999a). These controls undoubtedly markedly reduced TBT inputs from marinas throughout France. In the case of Arcachon, implementation was probably aided by the local provenance of many boat-owners and their interest in preserving a local industry. Their effectiveness in addressing more widespread TBT contamination remains questionable, however (Michel and Averty, 1999a).

13.4. UK harbours and coastal waters

Following Blaber's (1970) observations in Plymouth Sound, other research began to identify imposex as a more widespread problem. Frequency of imposex in *N. lapillus* was greatest close to ports and harbours, although evidence was also growing for the spread of the phenomenon through coastal waters of the southern United Kingdom (Bryan *et al.*, 1986 and 1987). In addition, Waldock and Thain (1983) linked the failure of attempts to introduce Pacific oyster culture into the United Kingdom in the early 1980s to TBT exposure rather than to fine sediment particles as previously thought. Reproductive failures and shell deformities in the United Kingdom showed many parallels with those recorded in Arcachon Bay.

Based on emergent concerns, the UK government introduced controls on the sale of TBT paints for use on small vessels in 1985. These included controls on retail sales, guidelines for handling paints and the setting of an environmental quality target (EQT) of 20 ng/l TBT (8 ng/l as tin), a level considered at the time to be sufficiently protective of marine biota (Waldock *et al.*, 1987). TBT concentrations in many coastal waters consistently exceeded the EQT. Moreover, as understanding developed through the 1980s, it became clear that concentrations well below the EQT could cause severe effects.

Sterilisation of female dogwhelks occurs at TBT concentrations as low as 3–5 ng/l TBT (Gibbs *et al.*, 1988), with almost all females affected once concentrations reach 10 ng/l and more fundamental changes at higher concentrations (including testes development and suppression of egg production).

During the 1980s, butyltin concentrations exhibited strong seasonal trends relating to boating activity and maintenance. Summer-time concentrations in marinas regularly exceeded 100 ng/l and occasionally 1 000 ng/l (Waldock *et al.*, 1987). Langston *et al.* (1987) reported similar concentrations in marina waters within Poole Harbour (southern United Kingdom). These marinas regularly served 5 000 leisure craft, engaged in hull maintenance and berthing. Even beyond the marinas, TBT concentrations above 100 ng/l were sometimes encountered. The overlap between field concentrations and those known to have severe developmental effects highlights the extent of the problem faced, and illustrates the inadequacy of the EQT as a protective instrument.

Evidence continued to accumulate during the 1980s. Bryan *et al.*'s (1986) study of dogwhelks in southwest England was particularly influential, concluding: 'that the incidence of 'imposex'... is widespread, that all populations are affected to some degree... Populations close to centres of boating and shipping activity show the highest degrees of imposex'.

The study also noted that the rapid increase in imposex in Plymouth Sound strongly coincided with contemporary increases in TBT applications, and revealed a strong correlation with tissue concentrations of organotin residues. The significance of this study was profiled in September the same year in the journal *Marine Pollution Bulletin* ('TBT linked to dogwhelk decline', 1986), in many ways marking wider acceptance of the central role of TBT in observed declines.

The geographical extent of the TBT problem in Europe had been recognised by the late 1980s (Bailey and Davies, 1988a and 1989; Gibbs *et al.*, 1991). Balls (1987) reported that use of TBT paints on salmon-farming cages had resulted in contamination of a Scottish sea loch, and noted the generally higher prevalence of imposex within sea lochs used for aquaculture. Cleary and Stebbing (1987) highlighted the additional concern of accumulation of organotins in the lipid-rich sea-surface microlayer, a threat considered particularly relevant to buoyant fish larvae (because of resulting exposure during sensitive developmental periods) and intertidal organisms (exposed to the microlayer during tidal ebb and flow) (Cleary *et al.*, 1993). Concerns over persistence and impacts in sediments were also emerging at this time (Langston *et al.*, 1987; Langston and Burt, 1991).

Faced with the accumulation of research, the UK government considered options for further retail restrictions and, in January

1987, reduced the maximum permissible content of organotins in applied paint, helped by technological developments (Side, 1987). These limits were rapidly superseded, however, by the introduction in May 1987 of a total ban on the retail sale of TBT paint (Waldock *et al.*, 1987 and 1988) for use on vessels under 25 metres and on fish cages (earlier voluntary measures having failed adequately to address this latter application). At this time, work by the UK Water Research Centre was under way to recommend environmental quality standards (EQS) for organotin compounds in water (Zabel *et al.*, 1988), involving consideration of toxicity and environmental behaviour of a range of organotins. It became apparent that for TBT, in particular, no-effect levels could not be achieved in the environment without use restrictions. Two years later, and in recognition of the low-dose effects, the 1985 EQT was replaced by an EQS of only 2 ng/l (Cleary, 1991).

In June 1987, the Paris Commission (PARCOM), responsible for administering the Paris Convention (1978), recognised that the use of TBT paints was causing 'serious pollution in the inshore areas of the Convention waters (NE Atlantic)' (PARCOM, 1987). PARCOM Recommendation 87/1 called for a harmonised ban on retail sales for application to pleasure boats and fish cages and, furthermore, for consideration of restrictions for seagoing vessels and underwater structures.

Though strong in intent, it quickly became apparent that PARCOM could not achieve restrictions within the commercial shipping sector. As this issue had already been referred to the International Maritime Organization (IMO), PARCOM instead turned its attention to inputs from docking activities, especially hull maintenance operations, in PARCOM Recommendation 88/1 (PARCOM, 1988). The effectiveness of such measures remains difficult to evaluate, though it is indisputable that shipyards and docks remain important point sources of organotins.

13.5. A global pollutant

Concerns over the impacts of TBT soon extended worldwide, leading to a series of national and regional measures towards the end of the 1980s. The annual Oceans symposia, organised by the US Marine Technology Society, did much to disseminate the growing body of research (Wade *et al.*, 1988b; Krone *et al.*, 1989). The NOAA (US National Oceanic and Atmospheric Administration) Mussel Watch programme highlighted the ubiquitous distribution of butyltins in bivalves from US coasts (Wade *et al.*, 1988a and b; Uhler *et al.*, 1989), while work in New Zealand confirmed accumulation in sediments (King *et al.*, 1989). Prohibitions addressing small vessels were adopted in the United States in 1988, followed in 1989 by similar measures in Canada, New Zealand and Australia, and by European legislation in 1991 harmonising controls throughout the EU (Evans, 2000).

13.6. Effectiveness of controls on small vessels

Retail restrictions undoubtedly resulted in a major shift away from the use of TBT paint on leisure craft (although the extent of illegal application remains unknown) and substantial reductions in inputs. In some regions at least, this was manifest in partial recovery of mollusc populations. In Arcachon Bay, monitoring of the effectiveness of regulations in reducing water column concentrations of butyltins was limited by the high detection limits of early analytical methods. While Alzieu *et al.* (1986) were able to demonstrate that concentrations fell from 900 ng/l tin in 1983 to below 100 ng/l by 1985, it took until the late 1980s to confirm that concentrations had fallen below 10 ng/l in most parts of the bay. Hydrodynamic disturbance has largely prevented reconstruction of sediment concentration trends (Ruiz *et al.*, 1996).

Recovery of the oyster beds, and resumption of commercial operations, in the mid-1980s provided the first indirect evidence of the effectiveness of the ban. Indeed, the recovery of the bay is frequently cited as an illustration of how the limited controls of the 1980s 'solved' the TBT problem (Nicholson and Evans, 1997; Evans, 2000), a view which has met substantial disagreement.

Evidence for partial recovery of severely affected mollusc populations certainly exists from several parts of Europe. For example, Minchin (1995) reported recovery of the flame shell (*Lima hians*) and associated flora in Mulroy Bay (Ireland), previously decimated through use of TBT on salmon cages between 1981 and 1985. In the sheltered inlet of Sullom Voe, Shetland Islands, and adjacent Yell Sound, imposex in dogwhelks was significantly less prevalent in

1995 than in 1991 (Harding *et al.*, 1997) although overall prevalence remained high. Evans *et al.* (2000a) suggest that there has been a more general recovery in UK dogwhelk populations over the last decade.

Other studies provide less cause for optimism. Minchin *et al.* (1997) stressed that recovery of dogwhelk populations in Ballybegs (Ireland) had been slower than expected, and recoveries in Sullom Voe and Yell Sound occurred against a continuing high background of imposex, and continued absence of dogwhelks from worst affected areas. In Canada, St-Jean *et al.* (1999) noted that butyltin concentrations in blue mussels (*Mytilus edulis*) and sediments from the southern Gulf of St Lawrence remained remarkably high eight years after retail restrictions. Despite initial declines in French coastal waters, dissolved TBT concentrations have since stabilised and remain at levels often well in excess of those known to cause adverse effects in some species (Michel and Averty, 1999a). In some areas of Arcachon Bay, concentrations remained high enough to cause imposex in sensitive species 10 years after the 1982 regulations (Ruiz *et al.*, 1996). Similar concerns have been voiced for UK estuarine and coastal locations (Cleary, 1991; Langston *et al.*, 1994).

The underlying causes of continued contamination and failure of ecosystems fully to recover undoubtedly differ from location to location. In some cases (e.g. Huet *et al.*, 1996), isolated but significant illegal use of TBT paints on small vessels has been implicated, while other researchers have stressed the importance of release from historically contaminated sediments (Waldock *et al.*, 1990; Langston *et al.*, 1994). High larval sensitivities (Gibbs, 1993) and long life histories (Rees *et al.*, 1999) undoubtedly also contribute to the slow rate of recovery of some species.

Potentially the most significant single contributor to continued widespread presence of TBT in the marine environment, however, and currently the most fiercely debated, is use on seagoing vessels.

13.7. The significance of seagoing vessels

At the time of the first national and regional prohibitions, the use of TBT paints on commercial and military ships was viewed as a lesser concern because, it was argued, those vessels spent most of their operational lives on the open seas. Nevertheless, these ships use inshore waters and port facilities on a regular basis. The scale of TBT inputs from hull maintenance of large vessels has been recognised for some time. Waldock *et al.* (1988) recorded TBT in wash water from a naval frigate at approximately 1 million times the lowest biologically effective concentrations, and estimated total input from cleaning a single vessel at 100 g of freely available TBT. Including TBT bound to paint-chips, which might act as a long-term reservoir of butyltins to the environment, led to estimates of almost 1 kg TBT per vessel per cleaning operation. Guidelines to control inputs from such operations have been in use in many countries for some time, although their effectiveness is difficult to evaluate.

Studies surrounding the Sullom Voe oil terminal (Bailey and Davies, 1988b) provide evidence for significant inputs of organotins from normal transport operations of TBT-coated ships. Although uses on small service vessels and marker buoys contributed to early inputs, more recent inputs arise predominantly from movements of oil tankers themselves. Significant contributions from heavy commercial traffic have also been highlighted in the Gulf of St Lawrence (St-Jean *et al.*, 1999).

While ongoing inputs from commercial shipping to coastal waters are now generally accepted, the significance of such sources to remote coastal and offshore areas remains the subject of intense debate (ten Hallers-Tjabbes, 1997; Evans, 2000). This is despite the increasing body of evidence supporting a link between density of shipping traffic and occurrence of biological effects (predominantly imposex) in offshore waters. Among the first to highlight imposex in the common whelk (*Buccinum undatum*) from the open North Sea and describe a positive correlation with shipping intensity were ten Hallers-Tjabbes *et al.* (1994). Similar correlations have since been reported in the Strait of Malacca, connecting the Bay of Bengal with the South China Sea (Swennen *et al.*, 1997; Hashimoto *et al.*, 1998), and in remote parts of the Galician coast (Ruiz *et al.*, 1998). Cadee *et al.* (1995) speculated that organotins might be responsible for local destruction of whelks in the Dutch Wadden Sea. More recently, Davies *et al.* (1998) estimated total annual inputs of 68 tonnes TBT to the North Sea from shipping, with continuous leaching from painted hulls when in port or under way the major contributor.

Nicholson and Evans (1997) provided further evidence supporting the pioneering work of ten Hallers-Tjabbes *et al.* (1994) but questioned the significance of what they termed 'mild' imposex against a background of overexploitation by shell fisheries. There is little doubt, however, that detection of population-level effects in offshore regions was an important factor underlying calls for the extension of restrictions in the mid-1990s.

During the same period, the truly global nature of the TBT problem was acknowledged (Ellis and Pattisina, 1990; Kannan *et al.*, 1995a, b and c). Though data remain limited for areas outside Europe and North America, recent studies in Japan and the Philippines (Harino *et al.*, 1998a, b, c and 1999; Prudente *et al.*, 1999) have gone some way towards redressing the balance, confirming (as suspected) the occurrence of similar relationships.

Recent research has also highlighted the widespread accumulation of organotins in organisms higher in the food chain, including cetaceans. Iwata *et al.* (1995) were among the first to determine concentrations in marine mammals, suggesting that the high levels recorded (up to 10 parts per million (ppm) in porpoise liver) reflected a low potential for metabolism of these compounds. Tanabe *et al.* (1998) later extended the data set, including species from North Pacific and Asian coastal waters. Kannan *et al.* (1996) described butyltin accumulation in dolphin, tuna and sharks from the Mediterranean, noting markedly different ratios of TBT breakdown products within the different taxa. First reports of butyltin residues in European cetaceans and seals, including in pelagic cetaceans that feed in remote offshore waters, were published by Ariese *et al.* (1998) and Law *et al.* (1998 and 1999).

Although uncertainty remains, evidence suggests that accumulation of butyltins in top predators might adversely effect the immune system. Kannan *et al.* (1997) reported high levels of TBT and its breakdown products (primarily dibutyltin) in bottlenose dolphins associated with mortality events on the US Atlantic and Gulf coasts. Similar correlations have been reported for Californian southern sea otters (*Enhydra lutris nereis*) exhibiting various infectious diseases (Kannan *et al.*, 1998).

Human intake of organotins from seafood, especially from fish farmed in TBT-treated cages, has long been recognised (Davies and McKie, 1987; Ishizaka *et al.*, 1989). Only more recently, however, has such intake been evaluated. Cardwell *et al.* (1999) reported that residues were widely detectable in US seafood but that estimated intakes were significantly below levels judged to be of concern for human health. In contrast, Belfroid *et al.* (2000) concluded that average intakes of TBT from seafood could lead to exceedance of tolerable daily intakes based on more sensitive immunotoxicological end-points for some products retailed in North America, Europe and Asia, while stressing that, for the majority of countries, data were simply unavailable.

13.8. Progress towards a global phase-out

Despite uncertainties regarding 'far-field' effects, contamination of the marine environment with organotins is clearly a persistent and pervasive problem. In 1995, and in spite of a decision by MEPC in 1994 that no further controls were necessary, ministers at the fourth Conference on the Protection of the North Sea in Esbjerg agreed: 'to undertake concerted action within IMO aiming at a worldwide phase-out of the use of TBT on all ships' (MINDEC, 1995).

Their preference for global action within IMO, in common with the position taken by the OSPAR (Convention for the Protection of the Marine Environment of the Northeast Atlantic) Commission, recognised the limits of regional measures in addressing a transboundary problem. On the basis of emerging research, MEPC agreed in 1996 to look again at the need for a global TBT ban (ten Hallers-Tjabbes, 1997). The possibility of such a ban was explicitly recognised at MEPC's 40th session (MEPC, 1997), marking a sea change in the thinking of that committee. Draft mandatory regulations were duly adopted the following year (MEPC, 1998).

The deadlines now incorporated in the draft International Convention on the Control of Harmful Anti-fouling Systems (2003 for phase-out of application of organotin paints, 2008 for their presence on ship hulls) were adopted under IMO Assembly Resolution A.895(21) in November 1999, and formal adoption of the convention is expected in 2001.

13.9. The question of alternatives

With the advent of such a progressive measure, attention is focusing increasingly on the availability of effective and economically viable alternatives to TBT (Evans *et al.*, 2000b). It is clear that some of the preparations in use prior to the widespread introduction of TBT (including those containing mercury or arsenic compounds) are not acceptable alternatives. Copper-based systems generally require booster biocides in order to be effective, biocides which themselves may present additional problems. Those containing the triazine herbicide Irgarol 1051, used extensively in some areas as a replacement for TBT on leisure craft, are commonly cited examples. Widespread occurrence of this herbicide in certain estuaries (Scarlett *et al.*, 1997), coupled with direct effects on plant growth in the field (Dahl and Blanck, 1996; Scarlett *et al.*, 1999), in some ways mirror early findings with respect to TBT.

Some commentators use such examples as justification against substitution of TBT in commercial shipping applications (Abel, 2000; Abbott *et al.*, 2000). This is a somewhat negative argument. The more appropriate question is, can fouling control be achieved without recourse to such highly toxic, persistent and bioaccumulative substances?

Research into use of natural antifouling chemicals is developing rapidly (Clare, 1998), though commercial application may remain some time away. Perhaps the most promising contemporary solution is biocide-free, 'non-stick' coatings which simply present a physical barrier to settlement. Such coatings have been commercially available for many years, and retain a significant market share for leisure craft. Their viability and performance on larger vessels remain under evaluation, although applicability for fast-moving craft has been demonstrated. Although significant technical issues remain to be resolved, advances towards antifouling mechanisms which do not release hazardous substances to the sea would seem desirable.

13.10. Late lessons from the TBT story

The above discussions indicate the degree to which the hazards of TBT were initially underestimated. It was initially assumed, for example, based on acute toxicity tests that concentrations in the ug/l (micrograms per litre) range were necessary to initiate biological effects and that, as a consequence, an EQT of 20 ng/l would be sufficiently protective. It quickly became apparent that, even if the EQT could be met (and it was greatly exceeded in some areas), severe biological effects could be expected. Had the endocrine-mediated mechanisms underlying imposex (Matthiessen and Gibbs, 1998) been identified earlier, the potential for low-dose effects would have been clear. Of course, much of our knowledge of organotin toxicity has been obtained through hindsight and is likely to continue to develop (Langston, 1996; Bouchard *et al.*, 1999; Morcillo and Porte, 2000). In the meantime, it may be hoped that the lessons gained will allow earlier identification and, wherever possible, avoidance of future problems in advance.

The persistence of organotins, permitting their accumulation and wider dissemination, was also underestimated. Early predictions that TBT would degrade rapidly in surface waters (see reviews by Simmonds, 1986 and Lee *et al.*, 1989) failed to account for the high lipophilicity and sediment-binding properties of organotins in seawater. Half-life estimates in the order of days for eutrophic surface waters must be contrasted with up to several years for residues in nutrient-poor waters (Michel and Averty, 1999b) and marine sediments (de Mora *et al.*, 1989), especially anaerobic sediments. The persistence of TBT, and toxicity to sediment communities, raises the prospect of delayed recovery of damaged ecosystems (Langston *et al.*, 1994; Dahllof *et al.*, 1999) and represents a substantial legacy to be borne by authorities responsible for dredging operations. This problem is only now receiving formal recognition within international conventions (London Convention 1972, OSPAR Convention 1992) regulating the dumping of dredge materials at sea.

Organotin bioaccumulation was also underestimated, and the specific role of biofilms in enhancing bioaccumulation and toxicity has only recently been recognised (Labare *et al.*, 1997). Accumulation in top predators was simply not envisaged.

Hindsight is a wonderful thing, of course. Nevertheless, increasing awareness of pathways and interactions within the marine environment, coupled with an appreciation of their complexity and indeterminate nature (Santillo *et al.*, 1998), should equip us to reduce or even avoid the potential for future TBT-style scenarios involving other persistent organic pollutants.

Two key factors contributed to early ignorance of the geographical extent of the TBT problem: low sensitivity of analytical methods and absence, in most regions, of adequate baseline data on the distribution and ecology of non-commercial species. The first of these was subject to an unavoidable process of methods development, partly overcome by the high specificity of imposex as a biomarker for TBT exposure. The second limitation illustrates a more general concern which could have been avoided through greater focus on the collection of 'baseline' data. Where such data were available (e.g. southwest England), they were invaluable in the early detection of effects on non-commercial species. Though frequently underrated, baseline studies play a vital role in the early detection of adverse trends and may consequently serve the application of precaution. Equally important is the need for continuing long-term monitoring of TBT-affected areas, as this will provide us with unique insights into recovery of pollutant-damaged environments.

The continuation of the TBT problem in coastal waters of Japan (Iwata *et al.*, 1995; Harino *et al.*, 1998a and b), despite a national ban on all applications of TBT for marine antifouling, clearly illustrates the transboundary nature of the problem. It would seem that universal, global restrictions are the only way to address the totality of the TBT problem and to achieve environmental quality standards in all coastal systems.

Finally, though the selection of alternatives to hazardous chemicals is never a simple task, this does not justify inaction. Several of the existing TBT 'alternatives' carry their own problems, though thankfully, on a global scale, none has yet proved as damaging as TBT. In selecting appropriate alternatives, those which do not rely on the release of hazardous substances to the marine environment should, perhaps, be viewed most favourably. Broader consideration of problems may give rise to more beneficial solutions than simple 'chemical for chemical' substitution.

13.11. Conclusions: precaution or retrospective action?

The use of organotin antifoulants has and continues to result in widespread and sometimes severe environmental effects. Unusually in the field of ecotoxicology, the evidence linking cause and effect (with respect to TBT-induced imposex) is irrefutable. Vos *et al.* (2000) refer to the phenomenon as 'the best example of endocrine disruption in invertebrates that is causally linked to an environmental pollutant'.

It would be difficult to argue, therefore, that any of the actions to address TBT to date have been precautionary, resulting as they have from extensive documentation of ecological impacts. Actions have undoubtedly contributed towards remediating the most severe problems, but this is not precaution. The 1987 UK restrictions, for example, stemmed from a realisation that environmental concentrations were already well above those known to cause chronic effects and had already resulted in widespread declines in gastropod populations. It could be argued, similarly, that agreement of the IMO convention which will bring about elimination of organotin antifoulants has come only after the consequences of continued use have been well documented. Although the significance of this global, legally binding treaty should not be underestimated, it too is fundamentally retrospective in action.

Complete phase-out of organotin paints from the global shipping fleet by 2008 will mark the closure of an important chapter in the TBT story. Persistence in sediments and long-lived biota will remain to be addressed, but widespread inputs from shipping should, at least, be a thing of the past. Organotin inputs will continue, of course, through their use as additives in a wide range of consumer products. Whether efforts to address these emerging challenges will draw on the lessons of the past remains to be seen.

TBT: early warnings and actions — Table 13.1.

Source: EEA

Early 1970s	Rapid increase in the use of TBT antifouling paints on vessels of all sizes and first reports of imposex in marine snails (Blaber, 1970; Smith, 1971)
1976–81	Repeated failure of larval settlement leads to near collapse of oyster fishery, Arcachon Bay, France
1982	France introduces legislation prohibiting the use of TBT paints on small vessels
1985	First controls introduced in United Kingdom limiting concentrations of TBT in paints
1986	Bryan *et al.* (1986) report widespread imposex in dogwhelks on southern coast of United Kingdom, linked to TBT
January 1987	United Kingdom announces further restrictions on TBT content of applied antifouling paint
May 1987	United Kingdom introduces ban on retail sale of TBT paint for use on vessels < 25 m and on fish cages
June 1987	PARCOM Recommendation 87/1 calls for similar ban over entire convention area (Northeast Atlantic)
1988	United States introduces restrictions. Waldock *et al.* (1988) highlight significance of inputs from shipyards
1989	Restrictions introduced in Canada, Australia and New Zealand
1991	Harmonised ban on retail sale of TBT paint introduced at European Union level
1994	Early reports of imposex in whelks from offshore areas of North Sea linked to shipping activity
1995	Ministerial declaration of fourth North Sea conference (Esbjerg) commits to working for global phase-out of TBT paint within IMO
1997	Concept of global phase out of organotin containing paints agreed at MEPC's 40th session
1998	Draft mandatory regulations aimed at such a phase-out adopted. OSPAR (Convention for the Protection of the Marine Environment of the Northeast Atlantic) prioritises organotins for action to cease all releases. Cessation of all releases of organotins to marine environment, under OSPAR's hazardous substances strategy in 2020
November 1999	Deadlines for phase-out adopted under IMO Assembly Resolution A.895(21)
2001	Text of International Convention on the Control of Harmful Anti-fouling Systems to be finalised. In 2003 worldwide prohibition on new application of organotin antifoulants to all vessels and in 2008 the existing organotin antifouling coatings will be replaced on all vessels worldwide

13.12. References

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