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# CHANGES IN PERIWINKLE (LITTORINA LITTOREA) POPULATIONS FOLLOWING THE BAN ON TBT-BASED ANTIFOULINGS ON SMALL BOATS.

P Matthiessen, R Waldock, J E Thain, S Milton & S Scrope-Howe

Ministry of Agriculture, Fisheries and Food
Fisheries Laboratory
Remembrance Avenue, BURNHAM-ON-CROUCH, Essex
United Kingdom

## ABSTRACT

In 1987 the UK Government banned the use of TBT-based antifouling paints on small boats of less than 25m. Following initial control measures taken in 1986 a programme of research was started to monitor concentrations of TBT residues in estuarine waters and sediments and to observe faunistic changes in highly contaminated estuaries. As part of this programme, the size-frequency and abundance of Littorina littorea populations has been recorded in the estuaries of the rivers Crouch, Blackwater (Essex), and Hamble (Hampshire). Since the TBT ban the concentration of TBT residues in water and sediments has been steadily declining. In the Crouch L. littorea population the frequency of O-group individuals has increased markedly, and there has been a simultaneous decrease in TBT residues in L. littorea tissues. Furthermore, plankton surveys of the river Crouch show the numbers of L. littorea eggs and veliger larvae to have progressively increased, suggesting that TBT may have impaired periwinkle reproduction and/or survival of the eggs and larvae. These observations are discussed in relation to known toxic effects of TBT to other mollusc species.

## INTRODUCTION

On the 1st July 1987, the use of tributyl-tin-based antifouling paints on fish farming equipment, and on boats of less than 25 m waterline length was banned in the UK under the provisions of the Control of Pesticides Regulations 1986, within the Food and Environment Protection Act 1985 (Anon. 1986). This ban followed partial controls in 1986 under the Control of Pollution Act. Use of these paints on larger vessels has not been banned, partly because it was expected that leachates from such vessels would be subject to greater dilution, but also due to the difficulties of enforcement in relation to foreign vessels. There are a variety of other (mainly pesticidal) uses for tributyltin (TBT), including wood preservation, control of aquatic snails which transmit bilharzia in the tropics, control of fouling in cooling systems and various industrial applications. However, it was expected that the ban would lead to substantial reductions in TBT concentrations in both freshwater and marine environments. The basis for this expectation was partly reductions in inputs, and partly anticipated degradation of existing residues. For example, Thain et al. (1987) measured TBT half-lives of 60-90 days in seawater at 5oC, while others have reported half-lives ranging from 6 days (Seligman et al., 1986) to at least 6 weeks (Maguire, 1986). Waldock et al. (1990) have found, however, that half-lives in marine sediment in laboratory mesocosms at 12oC are considerably longer (<1 year in the aerobic layer, 1.9 years in the anaerobic layer). Nevertheless, it was expected that TBT levels in marine ecosystems around the UK coastline would decline significantly over a period of a few years.

The UK ban on TBT-use in boats and fish farming was based on a considerable body of evidence demonstrating adverse effects on marine life, particularly molluscs (for a summary of the data, see IPCS (1990)). Some of the most striking effects were seen in Pacific oysters (Crassostrea gigas) where severe shell-thickening occurred at TBT concentrations above 20 ng/l (Thain et al., 1987). Based on these data, an ambient water quality target was set at 20 ng/l. However, subsequent research has shown that several gastropods (including the dogwhelk Nucella lapillus, the mud-snail Ilyanassa obsoleta, and the sting-winkle Ocenebra erinacea) when exposed to TBT levels as low as 1 ng/l can develop a sexual abnormality known as imposex, in which females develop a penis which can block the oviduct and impair reproduction (Gibbs and Bryan, 1986; Bryan et al., 1989; Gibbs et al., 1990). In the case of N. lapillus, this effect has caused the decline and elimination of many dogwhelk populations in the UK (Bryan et al., 1986). Such evidence led to the setting of a UK environmental quality standard (EQS) for TBT in seawater of 2 ng/l (UK Department of the Environment, 1989).

Since the UK ban on certain forms of TBT-use in 1987, the Ministry of Agriculture, Fisheries and Food (MAFF) and the Department of the Environment (DOE) have jointly funded a study of changes in estuarine ecosystems that were previously heavily contaminated with TBT. The majority of the ecological work has focussed on the River Crouch estuary in Essex, broadening the scope of research to cover as many species as possible in addition to molluscs. Ecological surveys are being conducted of species structure in benthic, epibenthic and intertidal communities in order to record the major changes which may occur as TBT contamination declines. This work is still in progress and the full results will be published in due course. The present paper describes one part of this programme which has been investigating the population structure of the common periwinkle Littorina littorea (L., 1758) (Mesogastropoda, Littorinidae). It had been noticed that almost nothing was known about the impact of TBT on this widespread coastal and estuarine species, so the opportunity was taken to study a number of populations in the Crouch estuary and elsewhere.

## MATERIALS AND METHODS

L. littorea is largely an intertidal species, although it can occur sublittorally. In muddy estuaries of the type studied here, it lives on the surface of mud flats, particularly if the mud is overlain by gravel and shell. It feeds largely on epilithic algae and vegetable detritus, although it may occasionally feed on dead animal matter. The sexes are separate, and after reaching maturity at a shell height of 10-12 mm (11-18 months), they copulate and the female lays about 500 planktonic egg capsules each containing 1-5 eggs. There are up to 10 periods of laying annually, with the main spawning period in the Crouch estuary occurring in the first half of the year. The veliger larvae are liberated after 5-6 days and remain planktonic for 4-7 weeks before settling. Adults can live for many years on sheltered shores, and can reach a shell height of 38 mm (Fretter & Graham, 1980).

The main population studied was situated on the north shore of the Crouch estuary near Creeksea (national grid reference TQ921967). A less-contaminated control population was studied at the mouth of the River Blackwater estuary at Sales Point near Bradwell in Essex (NGR: TM032090). Both populations have been surveyed 3 times annually since 1988. Less frequent surveys have also been made since 1989 at 3 sites on the River Hamble estuary in Hampshire (Lower Swanwick, NGR: SU495092; Warsash, NGR: SU488064; Hamble mouth, NGR: SU486057) which is known to be even more contaminated with TBT than the Crouch (Waldock et al., 1987; Waldock et al., 1988).

L. littorea is known to be distributed unevenly in the intertidal zone, with most individuals found in the mid-shore region (Smith & Newell, 1955), so the sampling programme was designed to cover the whole shore. The sites were visited at the time of low-water spring tides and 3 x 1 m2 quadrats searched by hand for L. littorea at each of the low, middle and upper shore stations, quadrats being divided into 4 x 0.25 m2 sub-samples. In order to check that the hand-searching was efficient, a few quadrats were dug out to a depth of a few centimetres and sieved to 0.5 mm. This revealed negligible numbers of individuals that had been missed by the hand-search. The L. littorea in each sub-sample were stored in a deep-freezer at -20oC to await measurements of shell height, and analyses of their soft tissues for monobutyl, dibutyl and tributyltin. Details of the analytical methodology have been published by Waldock et al. (1989).

Throughout most of the survey period, weekly plankton samples were taken at a point approximately 3 km downstream from the main Creeksea site on the Crouch (NGR: TQ947956). Samples were taken at slack high water by drawing a 60 mesh (0.31 mm aperture) plankton net almost horizontally through the water, 0.3m below the surface. Samples were preserved in buffered 10% formalin with rose bengal stain; L. littorea eggs and veliger larvae were later counted and identified in a sub-sample under the microscope (Scrope-Howe, S. and Thain, J. E. 1991 in prep).

## RESULTS

L. littorea size-frequency distributions are presented as a composite of the total population at each site (Figs. 1-5). In Fig. 1, it is apparent that the population at Creeksea throughout 1988 was largely composed of older individuals of >12 mm shell height, with a small number of 3-6 mm individuals appearing in October 1988. In subsequent years, the 3-11 mm size range was consistently much more frequent, and the overall population structure after 1988 was of normal appearance for a sheltered estuarine population (Fish, 1972; Moore, 1937).

By comparison, the population at Bradwell (Fig. 2) was composed of smaller (and probably younger) individuals, as expected for an open site exposed to wave action. Although the data are less complete than for Creeksea, there was no indication of abnormal population structure at any time. On the other hand, the L. littorea populations on the R. Hamble during 1989 and 1990 (Figs. 3-5) fairly closely resembled those on the Crouch in 1988, with a general dearth of individuals of less than 10 mm shell height, particularly at Lower Swanwick in July 1990, Warsash in June 1989 and July 1990, and at Hamble mouth in July 1990.

The population densities for Creeksea and Bradwell are shown in Table 1.

TABLE 1

Littorina littorea. Average numbers/Sq.M
Total population.

| Date | Creeksea Average | Standard Error | Bradwell Average | Standard Error |
|---|---|---|---|---|
| 5/88 | 30 | 6.68 | | |
| 10/88 | 38 | 6.33 | 68 | 13.49 |
| 1/89 | 58 | 10.41 | | |
| 5/89 | 73 | 19.4 | 137 | 41.33 |
| 10/89 | 62 | 13.65 | 136 | 22.37 |
| 1/90 | 62 | 12.42 | 109 | 14.83 |
| 5/90 | 57 | 13.08 | 139 | 27.13 |
| 10/90 | 57 | 10.23 | 101 | 22.06 |
| 1/91 | 76 | 16.95 | 157 | 29.00 |

Complete TBT analytical data are only available at present for Creeksea. Figures 6 and 7 show TBT concentrations in water and intertidal sediment respectively taken at 8 stations along the Crouch estuary, including Creeksea (M J Waldock, pers. comm. 1991; M E Waite et al., in press 1991). They show a steady decline in summer TBT concentrations throughout the period, with levels in water at Creeksea approaching the 2 ng/l EQS in 1990. When the L. littorea population at Creeksea was abnormal in 1988, the mean TBT level at this site was 22 ng/l in water and 0.8 mg/g (dry wt) in sediment.

Fig. 8 shows the TBT concentrations in pooled samples of whole L. littorea (excluding shell) from Creeksea and Bradwell. The Creeksea samples show a trend of declining TBT concentrations during the 1988-1990 period, and the Bradwell samples (although not sufficiently complete to reveal a trend) were consistently less contaminated. The TBT levels in Creeksea L. littorea taken when the population was abnormal in 1988 were in the range 0.12-0.17 mg/g wet wt. Levels of monobutyltin (MBT) and dibutyltin (DBT) in this period were 0.11-0.17 and 0.06-0.14 mg/g wet wt respectively. The only L. littorea samples yet analysed from the Hamble were taken in 1989 and contained MBT, DBT and TBT concentrations of 0.08-0.63, 0.09-0.37 and 0.10-0.37 mg/g wet wt respectively.

L. littorea egg and veliger larval abundances in Crouch plankton for the years 1987-89 are shown in Figs. 9 and 10 respectively. It can be seen that whereas eggs and veligers peaked at approximately 1000 and 600/m3 respectively in 1987 (the year before poor recruitment was observed in the adult population), the corresponding peak values for 1988 were 1600 and 4000/m3, and for 1989 were 2,400/m3 for both eggs and veligers. As expected, these peaks occurred during the first half of the year, although some veligers were found in all years through to November.

DISCUSSION

It is too early to be sure that the poor recruitment of L. littorea observed in 1988 at Creeksea and 1989/90 on the Hamble was due to TBT, but the circumstantial evidence points to this conclusion. In particular, the low egg and veliger abundances in 1987 on the Crouch probably explain the poor recruitment in 1988, and the residue data show that the adults in early-1988 were contaminated with TBT at a concentration (0.17 mg/g wet wt) comparable with those found in heavily-impacted Nucella lapillus populations. For example, Bryan et al. (1987) found TBT at 0.19-0.39 mg/g wet wt in N. lapillus from the Fal estuary, and Gibbs et al. (1987) reported 0.07-0.32 mg/g wet wt in the same species from Plymouth Sound and Torbay (data recalculated from dry wt values assuming that the soft tissues of N. lapillus are 80% water). The only published data on TBT levels in L. littorea (Langston et al., 1987) show that samples from heavily-contaminated Poole Harbour in 1985-87 contained 0.05-0.56 mg/g wet wt (recalculated as above).

Laboratory experiments in which L. littorea are being chronically-exposed to TBT are in progress at Burnham-on-Crouch, in an attempt to mimic the poor production of eggs and veligers seen in the field. The less-likely possibility that veligers are acutely susceptible at low environmental TBT concentrations is also being investigated. If these experiments confirm that reproduction in L. littorea is indeed affected by TBT, the precise mechanism will be of great interest because imposex has not been seen in this species. In any event, it is likely that L. littorea populations were never at risk from TBT, probably due to a combination of high individual longevity (nearly 20 years under aquarium conditions) and rapid recruitment from less-contaminated populations via the planktonic veliger. N. lapillus lacks a planktonic dispersal stage, and it will therefore take much longer for this species to recolonise an area.

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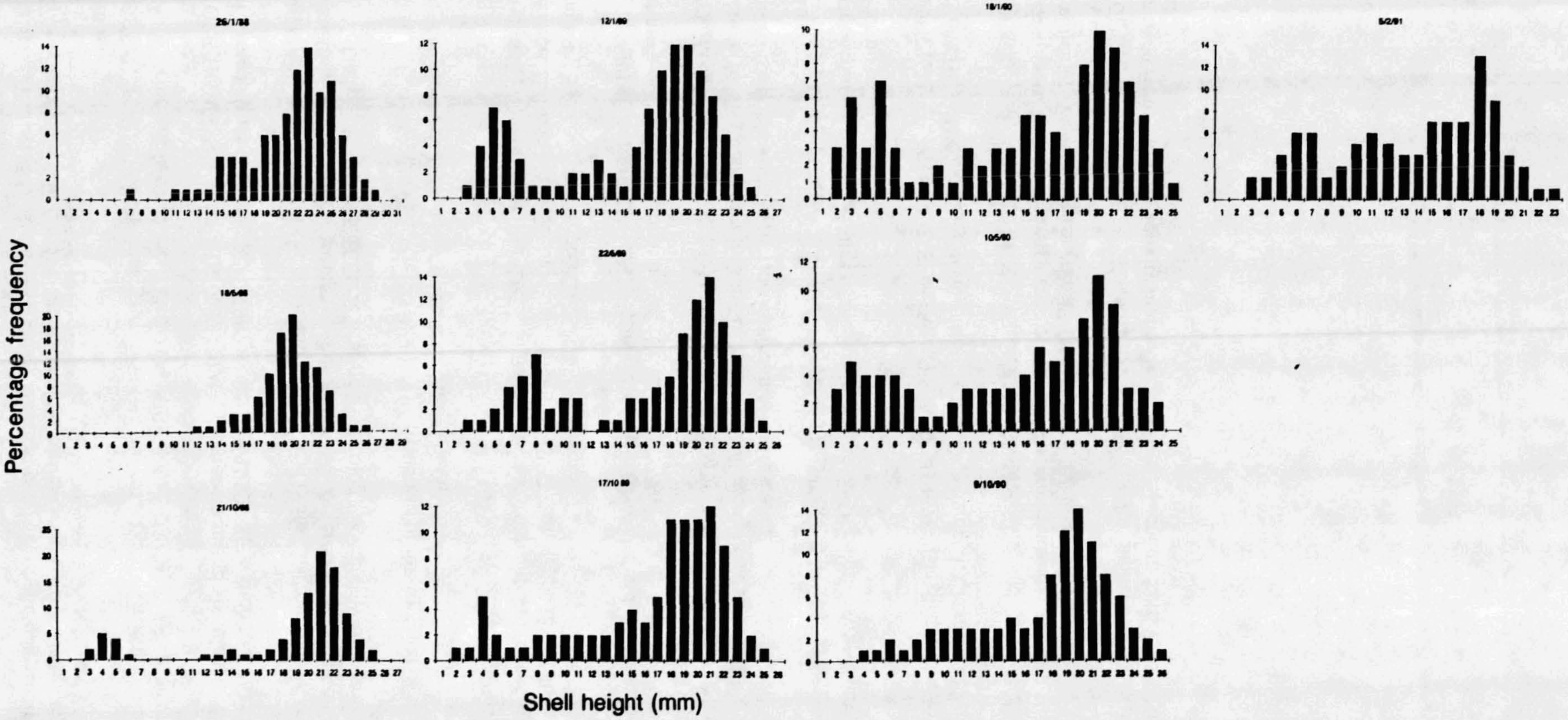


Figure 1. Frequency distribution of Littorina littorea shell height at Creeksea, R. Crouch, between January 1988 and January 1991.

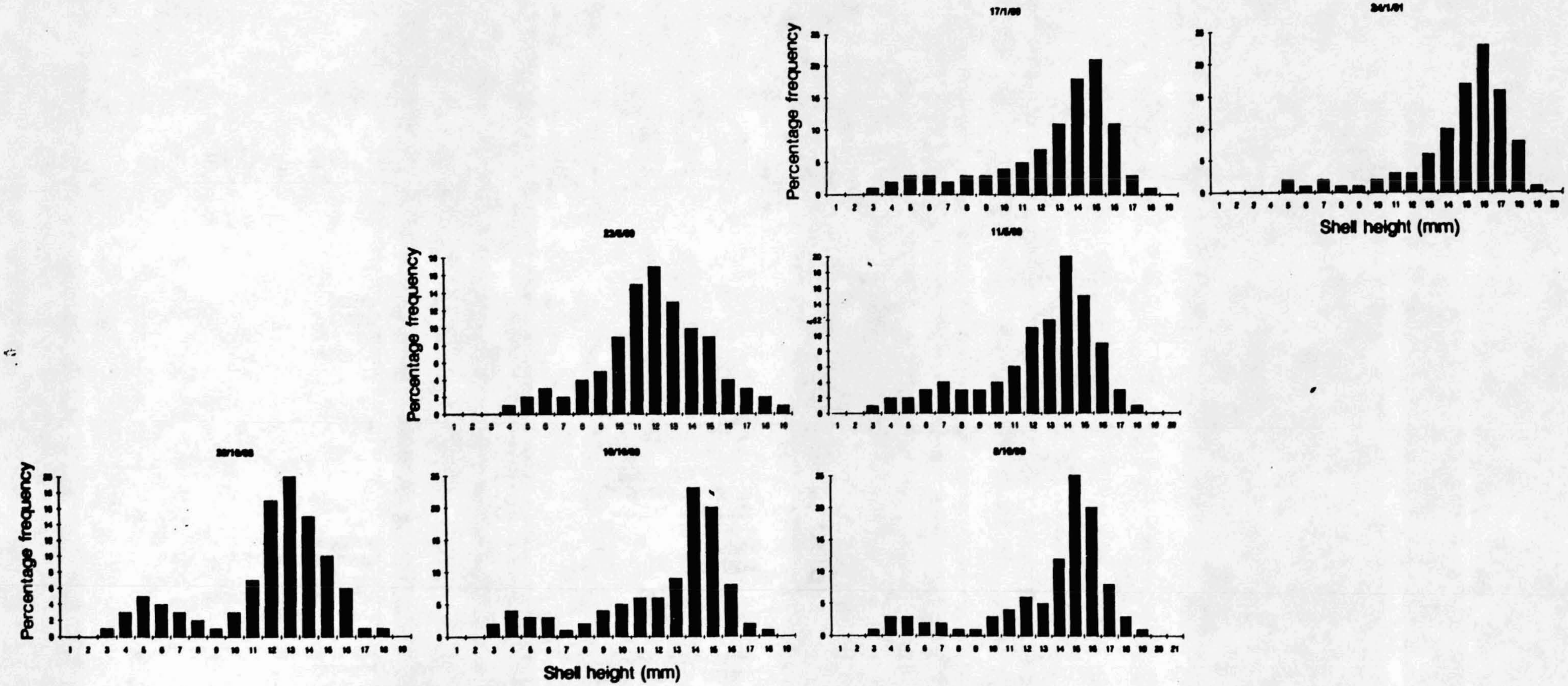


Figure 2. Frequency distribution of Littorina littorea shell height at Bradwell, R. Blackwater, between October 1988 and January 1991.

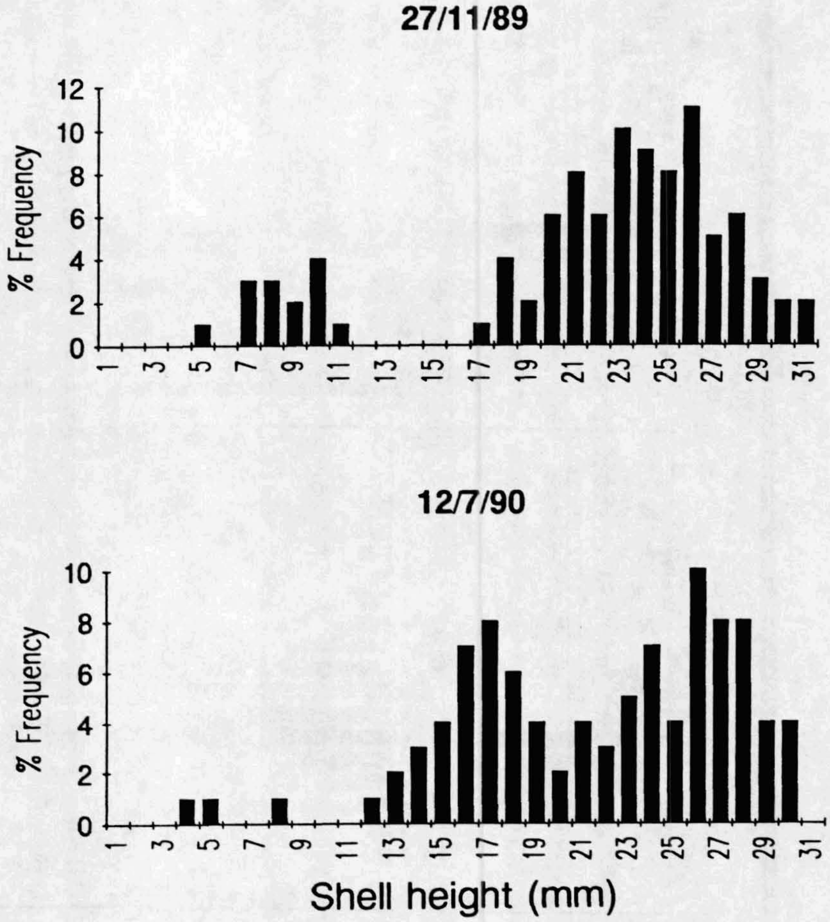


Figure 3. Frequency distribution of Littorina littorea shell height at Lower Swanwick, R. Hamble, between November 1989 and July 1990.

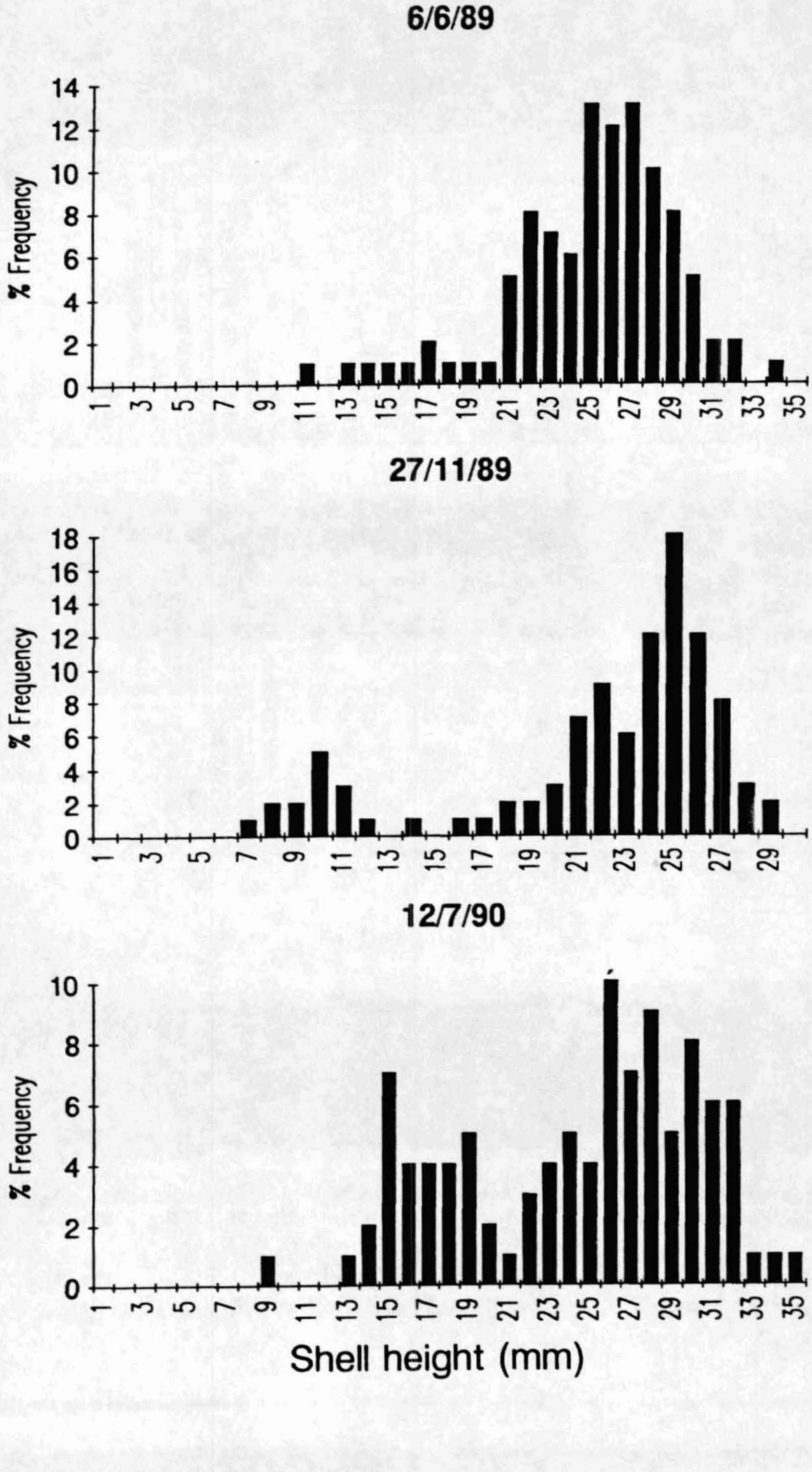


Figure 4. Frequency distribution of Littorina littorea shell height at Warsash, R. Hamble, between June 1989 and July 1990.

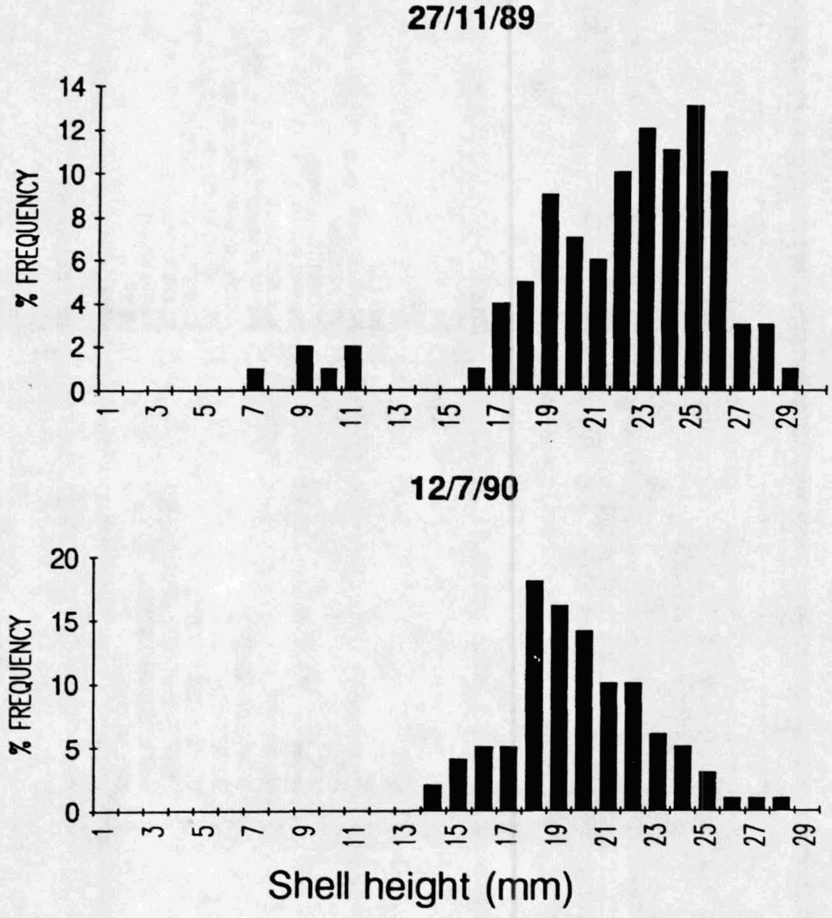


Figure 5. Frequency distribution of Littorina littorea shell height at Hamble mouth, R. Hamble, between November 1989 and July 1990. F. Fambridge; B. Bridgemarsh Island; C. Creeksea; B.N. Burnham; B.S. Bush Shore; R. Roach Mouth; H.B. Holliwell Buoy; H.P. Holliwell Point.

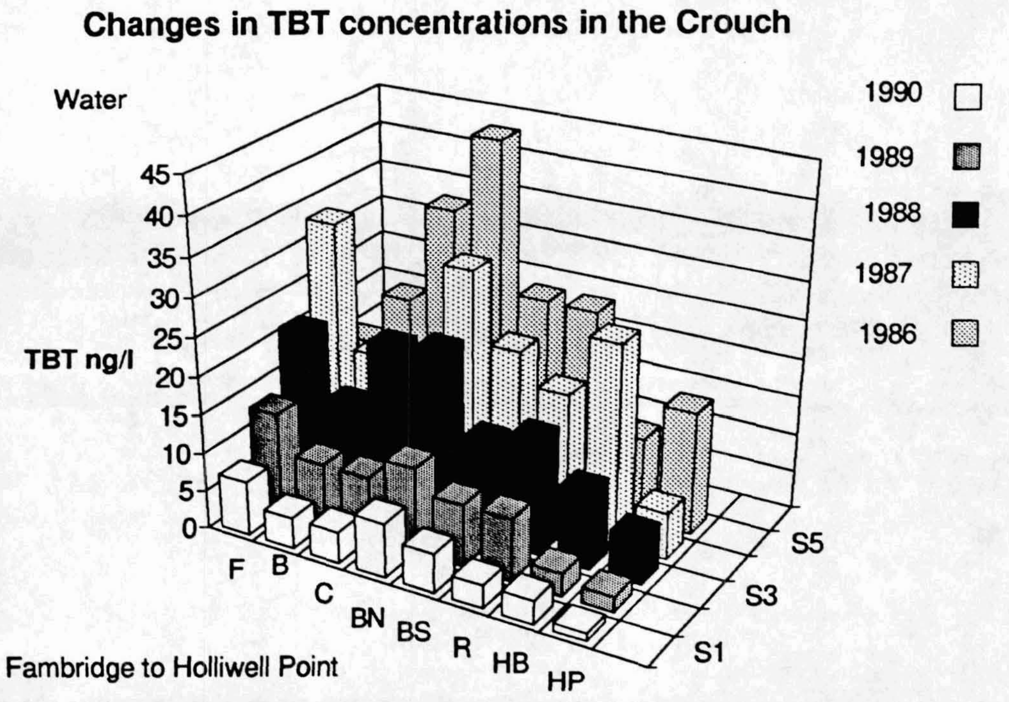


Figure 6. TBT concentrations in water at 8 stations on the R. Crouch between 1986 and 1990. Values shown are the means of samples taken between May and September (M. J. Waldock pers. com. 1991; M. E. Waite et al. in press 1991). F. Fambridge; B. Bridgemarsh Island; C. Creeksea; B.N. Burnham; B.S. Bush Shore; R. Roach Mouth; H.B. Holliwell Buoy; H.P. Holliwell Point.

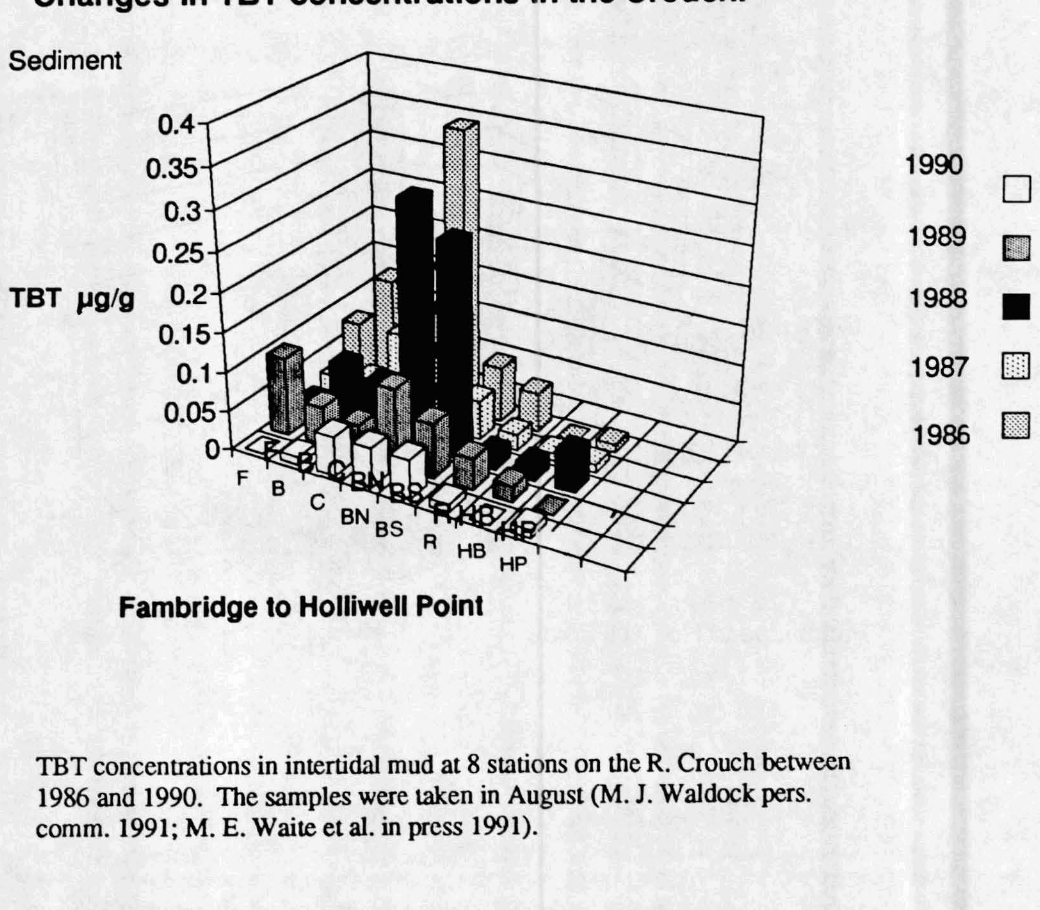


Figure 7. TBT concentrations in intertidal mud at 8 stations on the R. Crouch between 1986 and 1990. The samples were taken in August (M. J. Waldock pers. comm. 1991; M. E. Waite et al. in press 1991).

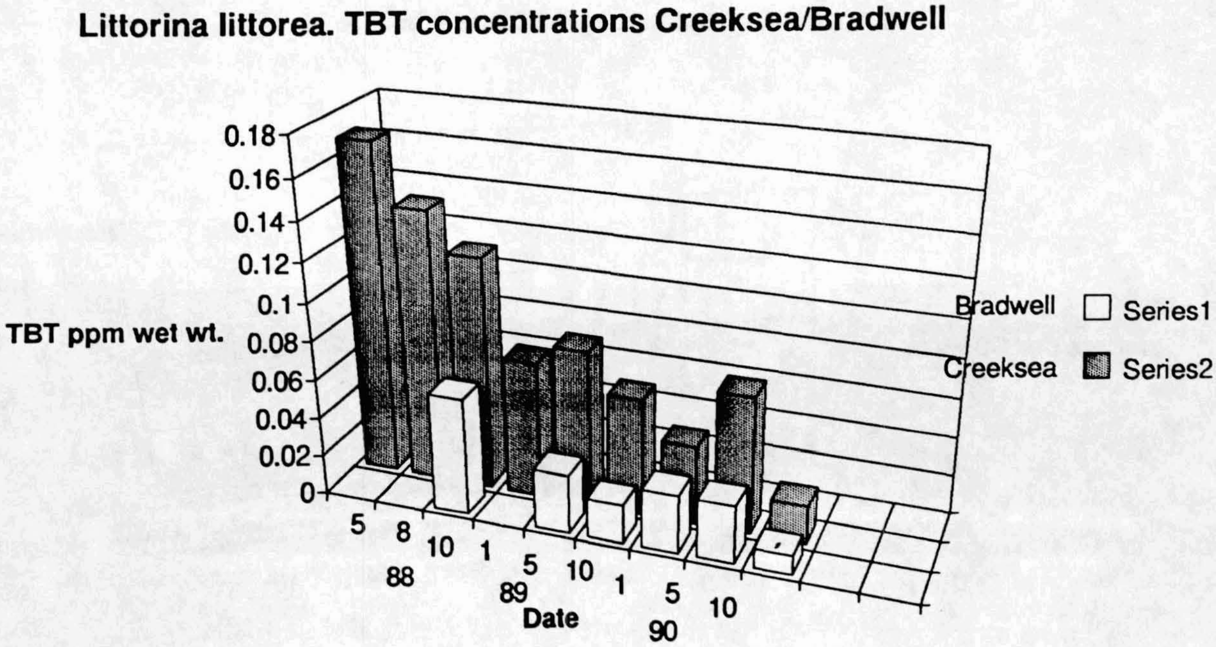


Figure 8. TBT concentrations in Littorina littorea from Creeksea and Bradwell between 1988 and 1990.

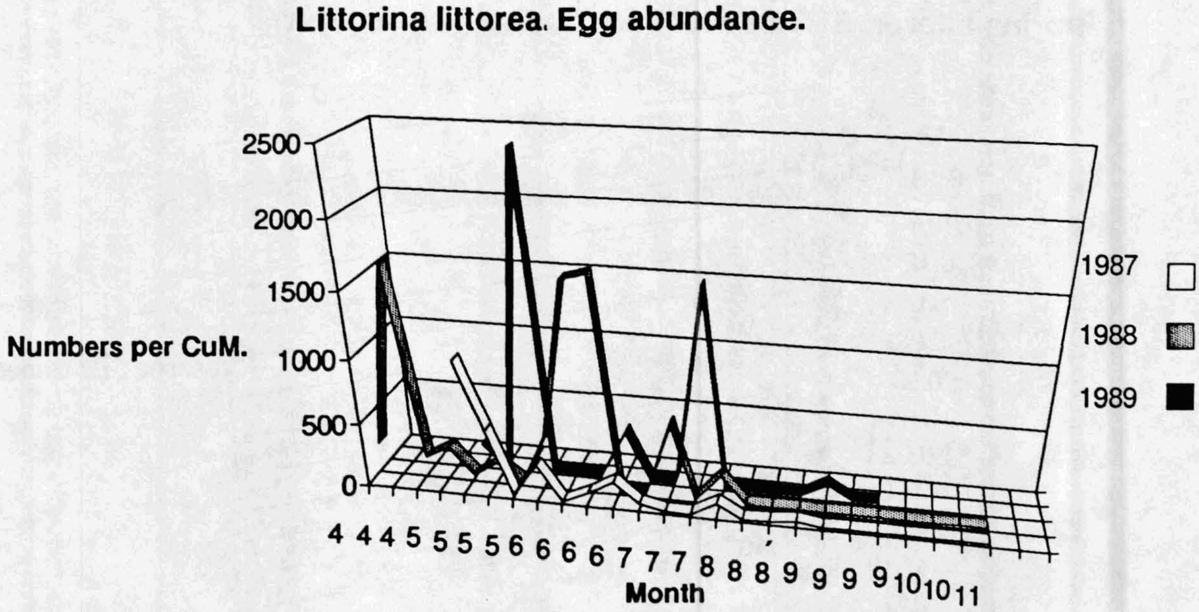


Figure 9. Abundance of Littorina littorea eggs in the Crouch plankton for the years 1987-1989.

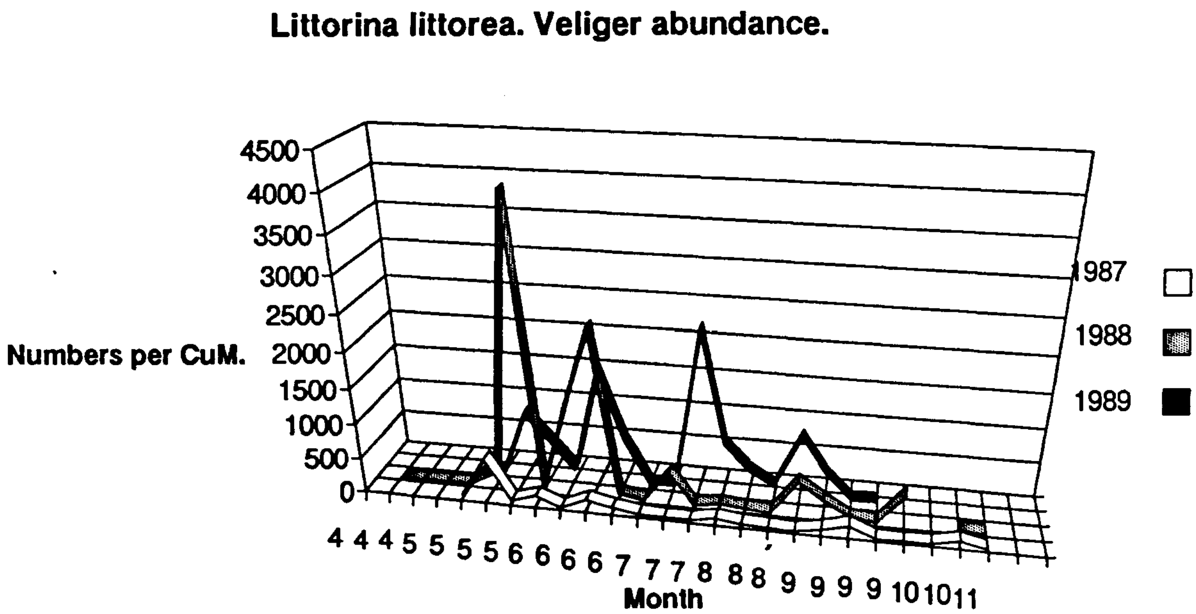


Figure 10. Abundance of Littorina littorea veliger larvae in the Crouch plankton for the years 1987-1989.