RESPONSES OF A HYDROID TO SURFACE WATER SAMPLES FROM THE RIVER TAMAR AND PLYMOUTH SOUND IN RELATION TO METAL CONCENTRATIONS

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(Figs. 1-8)

The River Tamar and its tributaries drain a highly metalliferous area; and increased metal levels in river water might be expected to affect biological water quality in the estuary. In order to detect possible effects we have used sensitive responses to stress of the hydroid Campanularia flexuosa as an index of quality of water samples from the River Tamar, Plymouth Sound and Cattewater. No water samples caused inhibition of colonial growth rate, but in each experiment there were significant variations in other more sensitive responses. It is these variations that we have related to metal distributions, although of the metals measured, only copper and cadmium occasionally occur in concentrations that could be biologically significant, and at the same time show any correlation with the hydroid responses. The limitation of the survey to water of relatively high salinity does not permit a firm conclusion about the origin of these metals, but there are some indications that local inputs may be important.

INTRODUCTION

This paper is the first dealing with a study of water quality in the River Tamar, Plymouth Sound and the Cattewater (Fig. 1). Here we consider the inorganic chemistry of water samples in relation to the responses of cultured Campanularia flexuosa to the same samples in the laboratory. The technique was developed in work with metals (Stebbing, 1976) and it is sensitive to the levels of toxic substances that sometimes occur in coastal waters contaminated with industrial effluents (Stebbing, 1979). The rationale for this approach stems from the need to estimate the biological impact of contaminants in the marine environment and from our inability to do so at present using physico-chemical data alone. This is mainly due to the numerous factors that alter the biological availability of toxicants in the environment (Bryan, 1976; Burton, 1979) and the complex effects caused by mixtures of toxicants (Stebbing & Santiago-Fandino, 1983). Sensitive organisms, used to bioassay water samples, provide a biological measure of water quality that integrates the complex effects of mixtures of toxicants and the many factors that may enhance or reduce their toxicity.

Although mining for metals in the area drained by the River Tamar ceased many years ago, the Tamar valley is highly metalliferous and drainage water from spoil heaps and adits enters the river and its tributaries. That these inputs are still important in determining the distribution of metals in organisms and sediments has been demonstrated on a number of occasions in recent years. It has been shown that metal levels in water from the River Tamar tend to increase as one passes upstream, particularly over the first 18 Km above Devil’s Point.

Fig. 1. Map of the lower reaches of the River Tamar, the Sound and Cattewater. The positions of sampling stations are given and the 2 m depth contour is indicated by a dotted line. Control samples (C) were collected 13 km offshore to the south near the Eddystone Rock.

(Bryan & Hummerstone, 1973) and these increases are reflected in the concentrations of metals in the sediments and organisms living in the sediment (Nereis – Bryan & Hummerstone, 1971; Scrobicularia – Bryan & Uysal, 1978) or attached on the surface (Fucus – Bryan & Hummerstone, 1973). It was natural therefore to suppose that metals might be one of the determinants of biological water quality in the River Tamar. In this paper variations in the responses of Campanularia cultured in water samples from the Cattewater, Plymouth Sound and River Tamar are related to the concentrations of metals (Cd, Cu, Fe, Mn, Ni, Pb, Zn) in the samples. This has led us to consider the form and availability of metals to organisms.
HYDROID RESPONSES TO METALS IN RIVER TAMAR

MATERIALS AND METHODS

Sampling and field measurements

Water samples were taken near the surface (0.5-1 m) at stations indicated on the chart of Plymouth Sound, the lower reaches of the River Tamar and the Cattewater (Fig. 1). By sampling at intervals of approximately 400 m it was intended to provide good definition of whatever variations in water quality might be detected. It was hoped that the localized effects of the numerous minor inputs to the system would be avoided by sampling along the mid-line of the river, and that variations in bioassay responses would reflect changes in the biological quality of the main water mass. Samples were taken at or soon after the predicted time of high water of spring tides.

The upstream limits were constrained by water salinity because the most sensitive of the hydroid indices shows a response to reduction of salinity below 30% (Stebbing, 1981a). The upstream stations in the Tamar and Cattewater were selected, therefore, to ensure that salinity would be close to or above this level; nevertheless, it did fall below 30% at a few stations, notably Nos. 17 and 18 in July 1980 (see Fig. 6).

Water samples for bioassay were collected using a submersible pump with Teflon coated components and plastic hosing. The samples were pumped into clean 20 l glass aspirators that had been washed with deionized water and sea water; they were filtered on the same day using Whatman GF/F papers to remove the larger biota and suspended sediments. A rapid filtering system driven by a peristaltic pump was used and water samples totalling 30 l were filtered in 15 min or less. A PVC pressure filter holder (filter size 142 mm) with teflon coated metal components and neoprene hosing was used. The first 1-2 l of each sample were used to rinse the system and were discarded.

At the time of collection, temperature and salinity measurements were made at each station using MC5 or Braystoke salinometers, and measurements of suspended solids were made with a Partech turbidity meter. Turbidity, temperature and salinity data are given in Fig. 6.

Hydroid culture and responses

The hydroid clone used here was derived from a colony of Campanularia flexuosa isolated from the River Tamar in 1973. As the hydroid has only reproduced asexually since then it is assumed the laboratory population is genetically homogeneous. The bioassay techniques are described in earlier papers (Stebbing, 1976, 1979) but will be summarized here. Colonies are grown on glass plates held in Perspex racks with seven replicate plates in each rack. The racks are kept in 2 l glass boxes and aerated. Air is passed first through activated charcoal, followed by a fine filter to remove particles. Colonies are fed daily to satiation with newly hatched Artemia salina nauplii. For 3 days after subculturing the colonies are allowed to become established before then exposing them to the water samples or experimental solutions for a further 11 days.

Typically the control group of organisms in each experiment is grown in water collected near the Eddystone Rock (ca. 13 Km offshore) at approximately the same time as the samples for bioassay. This water is considered to be of almost oceanic quality and in our experiments hydroids have not grown significantly better in water from anywhere else. However, the quality of this water is not invariably and in the April 1980 experiment, the Eddystone water was for some reason unknown to us of inferior quality. Variations in water quality offshore may be due to blooms of some species of microalgae which secrete toxic metabolites. It may therefore be relevant that there was a bloom of Phaeocystis at the Eddystone when water for this experiment was collected (D. S. Harbour, personal communication). On this occasion Station 3 was retrospectively designated as the control, but for all other experiments control hydroids have been grown in water from the Eddystone.

Three responses of the colonies to stress induced by toxic agents are used to provide indices of water quality; Fig. 2 shows the results of experiments selected to illustrate the form of the responses. The responses are non-specific and have been developed in experiments with varied levels of salinity, metals and other toxic agents (Stebbing, 1976, 1979, 1981a). Metal levels given are nominal concentrations above ambient levels in Eddystone sea water and are replaced daily with newly made up metal solutions.

The first response is the specific growth rate of experimental colonies expressed as a percentage of that of control colonies ($R\%\) . It has been shown that $R\%$ fluctuates with time due to the action of a control mechanism which regulates specific growth rate (Stebbing, 1981b). Although marked
fluctuations are observed when colonies are exposed to concentrations of copper as low as 1 µg/l, it has proved impracticable to use the amplitude or duration of the fluctuations themselves as an index of water quality. The growth index used is the mean $R^\%$ for the 11 days that the colonies are exposed to the water samples (Fig. 2). Although this is a precise index (S.E. < ±5%), it is not as sensitive as the other two indices. The mean threshold concentration at which inhibition of colonial growth rate by copper becomes significant is 14.5 µg/l, while other indices are responsive to much lower levels.

The tendency of stolons to curve, when colonies are exposed to unfavourable conditions, instead of growing radially from the explant, is a slightly more sensitive index (Fig. 2), although it tends to be more variable than the other indices. Nevertheless, the signal to noise ratio has proved good enough to exhibit significant responses to variations in water quality in Swansea Bay (Stebbings, 1979). Curving is assessed visually in terms of numbers of stolons longer than 2 cm that curve by 90° or more, expressed as a proportion of the total number of stolons of at least 2 cms. Frequency increases at copper concentrations of 10 µg/l or more, but decreases above those concentrations that inhibit colony growth.

The remaining index is the stimulation of gonozooid frequency, which, like the other indices is a generalized response to stress (Stebbing, 1980, 1981a). Marked increases in gonozooid frequency occur at concentrations of added copper of as little as 0.1 µg/l (Fig. 2), so this is the most sensitive of the three indices. Gonozooid frequency is expressed as the number of gonozooids as a proportion of the total number of colony members.

One complication in the use of gonozooid frequency as an index of stress to which the hydroids are exposed is that the relationship is biphasic. A consequence is that within certain ranges of stress levels, given in terms of copper concentration in Fig. 2, a gonozooid frequency may indicate the effect of two possible copper concentrations. Furthermore, above 3-4 µg/l of copper, gonozooid frequency becomes less than that in the control colonies, but at these higher levels the other responses usually indicate which part of the gonozooid frequency curve is appropriate.

All the indices are non-specific effects of, or responses to, stress (Stebbing, 1981c) and are observed in experiments with approximately the same relative sensitivity whatever agent is used.

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**Fig. 2.** The responses of *Campanularia flexuosa* that are used as indices of water quality. The form of the responses and their relative sensitivities are shown by the results of experiments with copper. The responses are (A) specific colonial growth rate ($R^\%$ for 0-11 days), (B) gonozooid frequency, (C) stolon curving frequency. Data points represent the mean of seven colonies and bars indicate standard errors of the means. Open and solid squares differentiate between the results of two experiments.
Trace metal analysis

Trace metal levels in sea water are typically too low for direct determination by flame atomic absorption; it is necessary therefore to use preparatory methods to separate and concentrate metals from sea water samples. We have used two methods, the first of which is based on procedures described initially by Riley & Taylor (1968) using a chelating ion exchange resin. More recently this technique has been used by Kingston et al. (1978). The second is a liquid–liquid extraction method using a mixture of chelating agents and is described by Danielsson, Magnusson & Westerlund (1978). Both procedures provide a means of extracting and concentrating chelatable metal (which includes the ionic fraction), rather than the total metal in sea water. (Kingston et al. 1978; Danielsson et al. 1978; Florence & Batley, 1976, 1977; Abdullah, El-Rayis & Riley, 1976).

Water samples for metal analysis were collected and filtered at the same time as those for bioassay, using the same equipment. Samples of 10 l were taken and stored in polyethylene containers that had been used exclusively for sea water for some years. After filtration 9 l were set aside for resin extraction and the remaining 1 l for solvent extraction. 10 ml Chelex 100 resin (100–200 mesh), in the ammonium form, was packed into a 12 mm by 10 cm glass column and supported by a cleaned glass wool plug. The seawater sample, at its natural pH, was then passed through the column at a flow rate of 3 ml/min. Column blanks were determined by passing the effluent from the sampling column through a second column in series. Alkali and alkaline earth metals were eluted from the column with 1 M ammonium acetate, prior to the elution of metals with 2 M nitric acid and their determination by flame atomic absorption.

The solvent extraction procedure used here is that given by Danielsson et al. (1978). Metal carbamate complexes were extracted from sea water (buffered to pH 5) with ammonium pyrrolidine dithiocarbamate (APDC) and diethylammonium diethylidithiocarbamate (DDTC) into 1,1,2-trichloro-1,2,2 trifluoroethane (Freon TF) and back extracted into 0.3 M nitric acid, prior to analysis by flame atomic absorption. Sea water samples stripped of these trace metals were used to provide sample blanks.

As the use of Chelex ion exchange resin for extracting and concentrating metals from sea water is slow and requires samples of 9 l or more, a comparison was made with the more rapid solvent extraction method for which a sample of only 500 ml is needed. Similar concentrations of copper and zinc are extracted using the two techniques (Fig. 3), but the solvent extraction method is of limited use in this work since its concentration factor of 50 (compared with a value of 400 for the Chelex resin technique) is too low to result in extracts containing detectable metals other than copper and zinc.

At the end of the experiments, hydroid tissues were analysed to determine the concentrations of metals to see whether the responses were related to the levels that had accumulated in their tissues. Colonies attached to glass plates were rinsed with deionized water before removing them. The tissues were then freeze dried in 10 ml Perspex pots. The dry samples were weighed and placed in the Teflon vessel of a Uni-seal decomposition bomb, digested with 2 ml of concentrated nitric acid (Aristar) and heated for 1 h at 120 °C. The cooled acid digest was transferred to a 10 ml volumetric flask and the Teflon vessel rinsed with 3 ml of deionized water. A further 2 ml of deionized water was then heated in the decomposition bomb for 30 min at 120 °C. All washings were added to the original sample, which was diluted to 10 ml with deionized water. Blanks were determined by repeating the same procedure with sample absent.

Recoveries of cadmium, copper, lead, nickel, zinc, iron and manganese in the ionic form are given in Table 1. Mixed standards, prepared from BDH Standard Solution of nitrate salts were used. For solvent extraction, 10 µg of metal was added to 500 ml of sea water, and for Chelex column extraction 10 µg of metal was added to 1 l of sea water. Clearly the solvent extraction procedure is unsuitable for manganese.
Recoveries were not carried out for hydroid tissue digestion because of the small quantities of sample available. Previous recovery data for Mytilus tissue using this digestion technique have proved satisfactory (> 90% recovery) (Cleary et al., unpublished).

![Graph showing analysis of water samples from stations 1 to 16 for copper and zinc in April 1981. Solid histograms indicate analyses using the solvent extraction technique and cross-hatched histograms indicate those using Chelex ion exchange resin to extract and concentrate the metals.]

Table 1. Recoveries of 10 μg of metals from sea water using two extraction techniques

<table>
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<tr>
<th>Metal</th>
<th>Recovery by Chelex extraction (μg)</th>
<th>Recovery by solvent extraction (μg)</th>
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</thead>
<tbody>
<tr>
<td>Cd</td>
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<td>9.7</td>
</tr>
<tr>
<td>Cu</td>
<td>9.3</td>
<td>9.7</td>
</tr>
<tr>
<td>Fe</td>
<td>9.6</td>
<td>9.6</td>
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<tr>
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<td>10.1</td>
<td>10.1</td>
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<td>10.6</td>
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<tr>
<td>Zn</td>
<td>9.5</td>
<td>9.8</td>
</tr>
</tbody>
</table>

RESULTS

Responses of the hydroid

The responses of Campanularia to water samples from the River Tamar, Plymouth Sound and Cattewater in three experiments (April 1980, July 1980, May 1981) are given in Fig. 4. In none of these experiments was mean specific
growth rate ($R^\circ_0$) significantly inhibited with respect to the controls, suggesting that such variations in water quality as might exist do not have a deleterious effect on the metabolism of the hydroid colonies. Growth is a good index of health, in the sense that biosynthesis is an integration of many different physiological processes, and their impairment is likely to be reflected in the growth of an organism.

There were significant responses of the two more sensitive indices in all three experiments. In April 1980 stolon curving frequency was high at Stations 11 and 14, and this response was matched by increases in gonozoid frequency at these
two stations. However, these increases in gonozooid production were part of a clearly defined increase in this response from Station 13 upstream which reaches a peak at Station 15, with a frequency more than 25 times that of the control colonies. At Station 16 – the most upstream station in this experiment – gonozooid frequency was less, although still significantly greater than the control colonies.

The experiment in July 1980 did not show the same pattern, although there were significant increases in gonozooid frequency at Stations 7, 10 and 18. The most marked increase in frequency, over 20 times that in the control colonies, occurred at Station 18. In a further experiment during May 1981 there were significant increases in stolon curving in colonies cultured in samples from

Fig. 5. Analysis of five metals in water samples taken at the stations indicated (see Fig. 1) in April and July 1980. Metals were extracted from the 91 samples using Chelex ion exchange resin. The horizontal lines indicate the detection limits for each metal.
Stations 13, 15 and 16, and increases in gonozooid frequency in samples from Stations 13 and 14.

It is clear that water samples from the more upstream stations in the River Tamar caused the greatest responses of the hydroids, particularly between Stations 13 and 16 in April 1980 and May 1981. However, it is possible that water quality may not continue to deteriorate upstream because, in these two experiments at least, the responses appear to be somewhat less marked at the upstream station.

![Graphs showing salinity, temperature, and turbidity measurements](image)

Fig. 6. Salinity, temperature and turbidity measurements made at the time of collection of water samples for bioassay experiments (Fig. 4) and metal analysis (Fig. 5). Station positions are given in Fig. 1.

**Metals in River Tamar**

Water samples for metal analysis were taken at the same time as those for bioassay in April and July 1980. The metals analysed in April were iron, copper, zinc, manganese, cadmium, lead and nickel, but of these lead and nickel were below detection limits (Fig. 5). The metal concentrations were generally lower in July than in April because of their dilution by more fresh water than is seasonally typical, particularly at the more upstream stations (Fig. 6). The iron concentrations tend to be more variable than other metals, due to the tendency for iron to form colloidal species which are not extractable by the Chelex resin technique.

Levels of copper in the Sound (Stations 3–6) were lower than those in the River
Tamar in April 1980, and a similar trend was found in July. Both experiments reveal a tendency for an increase in levels up the Tamar, but superimposed on these are localised peaks, for example, at Station 13 in July and at Station 12 and 15 in April. The highest copper concentration in either experiment was found at Station 15, which coincides with the highest gonozooid frequency of all the samples tested in this experiment.

The highest level of zinc was found in the Cattewater in April 1980. As the responses to water samples were only marginally significant, if at all, no samples were taken there for analysis or bioassay in later experiments. The next highest zinc level was found at Station 15, coinciding with the highest concentration of copper and the highest gonozooid frequency recorded in this experiment (Fig. 4).

The highest levels of manganese occurred in the Cattewater and at Station 12 in April, but the most marked feature of these data is the trend of increasing concentration with distance upstream. In a recent study of manganese in the Tamar (Morris, Bale & Howland, 1982) it has been shown that this distribution is quite typical and is due to a number of factors. The concentrations of dissolved manganese in surface waters are due largely to the behaviour of the particulates with which manganese associates. There is resuspension of dissolved manganese from reduced sediments and subsequent deposition after association with suspended particulates with each tidal cycle. The effect of variations of factors that impinge on this process, primarily turbidity and salinity, accounts for the observed distribution of dissolved manganese in their data and a similar explanation may apply further downstream (Figs. 5, 6).

The distribution of cadmium is unlike any of the other metals. Levels in the Cattewater and Sound were low, but in the River Tamar in April 1980 there were higher concentrations from Stations 11 to 15 with a maximum at Station 14 of nearly 1.5 μg/l. The pattern of distribution suggests a localized input, and the absence of levels greater than 0.5 μg/l in July 1980 indicates that it was also a transient input.

Correlations between responses and metals

We have attempted to correlate the variations in hydroid responses to water samples with the concentrations of metals analysed. It seems that any response of an organism to variations in water quality is likely to be a response to some integration of the biologically active contaminants present. Strong correlations with individual contaminants are unlikely unless their effect is overwhelming. Furthermore, because of the numerous constituents of domestic and industrial effluents that may elicit the hydroids' stress responses other than the seven metals analysed here, we did not necessarily expect the metal data alone to account entirely for the bioassay results.

If the concentration of a metal in a sample does not exceed that found to elicit hydroid responses in laboratory experiments with individual metals, then it can be assumed that the metal would be insignificant in the bioassay experiments. The concentrations of iron, manganese and zinc fall into this category and may be
ignored because of their low toxicity, but the concentrations of copper and cadmium in some water samples from the River Tamar exceeded those known from experiments to be capable of stimulating gonozooid production. In experiments with these metals, sea water to which as little as 0.1 µg/l of copper or 0.5 µg/l of cadmium has been added caused significant stimulation of gonozooid production (Stebbing, 1980).

The distributions of copper and cadmium in the samples and the responses of the hydroids to those samples sometimes suggests that both metals might on some occasions be important. For example, in April 1980 the highest levels of copper occurred in the sample from Station 15, which produced the highest gonozooid frequency. The highest level of cadmium in this experiment occurred in a water sample from the adjacent Station 14, which also produced a significantly elevated gonozooid frequency. Plots of gonozooid frequency against metal concentration for copper and cadmium gave correlation coefficients of 0.5512 and 0.5259 (both significant at the 5% level) but when the concentrations of copper and cadmium are added together the correlation coefficient increases to 0.6112 (significant at the 2% level) suggesting that the observed response may be elicited in part by the combined effect of copper and cadmium, although other unidentified toxicants may also be important.

Chemical stability of stored water samples

As soon as water is sampled it may begin to change chemically in a way that makes it less representative of the water from which it was taken. Such changes are likely to affect the responses of bioassay organisms to a water sample, making it difficult to extrapolate back to the environment (Hennig, Hennig & Greenwood, 1981). Chemical changes related to microbial activity have been minimized by membrane filtration (0.45 µm), as this has been found to reduce changes in metal levels in solution during storage (Dr A. W. Morris, personal communication). However, on learning that membrane filters may remove some toxicants (PCBs and DDT) from the solution (Kurtz, 1977) we began to use glass fibre filters instead (Whatman GF/F) which have been used throughout these experiments.

In order to determine the stability of metal concentrations during bioassay experiments subsamples from the 10 l samples (polythene containers) in April 1980 and from the 20 l samples (glass containers) in July 1980 were analysed by the solvent extraction technique (Fig. 7). In both cases there is a suggestion of some loss of zinc from solution over the period of the experiment, while the data for copper are more variable. The type of container does not appear to have any effect on the loss or retention of zinc and copper in sea water.

While it might be preferable to arrest all such chemical changes, this is not practicable without changing the biological properties of the water samples. The most obvious course to reduce chemical changes that might affect the results of a bioassay experiment is to shorten the experiments, but for the hydroid bioassay sensitivity of the colonies is a function of the duration of exposure over the first 11 days. While the levels of metals and other contaminants in solution may change
in that time, it does appear that the relative differences for any single contaminant between water samples would remain because the samples are treated and stored identically. Such minor changes as do occur (Fig. 7) do not therefore invalidate the results of bioassay experiments, but suggest that they are most valid when considered in relation to one another rather than in isolation.

**Fig. 7.** Analyses of copper and zinc in water samples used in experiments in April 1980 and July 1980. Samples were analysed at the beginning (solid histograms) and end (cross-hatched histograms) of the bioassay experiments. Analyses were carried out using the solvent extraction technique.

**Acidification of water samples and DOC**

Levels of copper, zinc and iron in samples from the Tamar extracted using the solvent extraction technique (Fig. 8) were compared with levels in the same samples acidified to pH 2.0 before solvent extraction. Levels of metals extracted at pH 2 do not correlate with those extracted at pH 5, but significantly greater amounts are extracted at the lower pH (particularly for iron). More interesting is the fact that the levels of metals in samples acidified to pH 2 show similar spatial patterns, having peaks in concentration at Stations 8 and 18. Dissolved organic carbon (DOC) analyses of sea water are also carried out at pH 2, and DOC values show good correlations with copper, iron and zinc extracted at this pH. Florence & Batley (1977) in their review of trace metal speciation in natural waters show that up to 30% of the dissolved zinc and 10–60% of dissolved copper exist in forms associated with organic matter and that acidification to pH 2 releases metals from complexed forms. However, more stringent acidification is necessary to yield total metal. Organically bound metals are biologically less available and therefore less toxic to aquatic organisms than ionic metal species. The chelating resin Chelex-100 does not remove metals associated with organic or inorganic colloidal particles, or those forming strongly bound organic complexes, and so provides a better estimate of biologically available metal when used at the natural pH of sea water (Fig. 5) than extraction methods under more acidic conditions (Fig. 8).
Metal analysis of hydroid tissues

After the experiments metal levels in hydroid tissues were determined (Table 2). Unfortunately only zinc and copper were detectable in the acid digests, probably because of the small quantities of tissue available for analysis (2–10 mg). However, neither the zinc nor the copper data for April or July reflect the increases in gonozooid frequency found in hydroids cultured in water from the upstream stations. Nor did the tissue data reflect the lower levels of zinc present in the water sampled in July.

Fig. 8. Analyses of copper, zinc, iron and dissolved organic carbon (DOC) in water samples from stations given in Fig. 1. Metals were concentrated using the solvent extraction technique. Solid histograms indicate levels after extraction at pH 5 and cross-hatched histograms indicate levels after extraction at pH 2 (the same pH used for determination of DOC).

DISCUSSION

Interpretation of hydroid responses

The most common kinds of indices of the effects of toxic levels of marine contaminants are the deleterious effects; for example, the inhibition of reproduction, the occurrence of morphological abnormalities or increased susceptibility to disease. Here we demonstrate that while colonial growth is not significantly affected by the variations in water quality in the River Tamar, nevertheless there are marked variations in the responses of hydroids to the water samples. In the case of gonozooid frequency it seems that we are observing an adaptive response to stress and it is natural that such a response should be more sensitive than those
indices that measure damage to organisms (Fig. 4). It seems that in waters that become unfavourable for *Campanularia*, the colonies respond by redirecting nutrients to produce gonozooids instead of hydranths. The increased number of gonozooids may be expected to produce more planktonic larvae, resulting in greater dispersal of the genotype and increasing the likelihood that the genes would survive localised toxic effects that might prove lethal for the parent colony (Stebbing, 1980, 1981a).

**Table 2. Metal analyses in hydroid tissue after bioassay experiments**

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<tr>
<td></td>
<td>mg dry weight</td>
<td>µg/mg dry weight</td>
<td>µg/mg dry weight</td>
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<td>April</td>
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**Hydroid responses in relation to metal concentrations**

It has been shown that sometimes the levels of copper and cadmium found in the River Tamar exceed those known from our laboratory experiments to be capable of stimulating gonozooid production. Furthermore, on more than one occasion at certain stations significant responses (Fig. 4) seem to be related to the concentrations of these two metals (Fig. 5).

The toxicity of copper and cadmium is dependent upon the concentration of free metal ions in sea water, which may be significantly less than the total dissolved metal concentration. Cupric ion activity has been shown to be the best index of copper toxicity in bacteria (Sunda & Gillespie, 1979), algae (Sunda & Guillard, 1976; Anderson & Morel, 1978), fish eggs (Engel & Sunda, 1979) and fish (Brown, Shaw & Shurben, 1974). Likewise cadmium toxicity has been shown to be a function of the concentration of free metal ions in experiments with *Palaemonetes pugio* (Sunda, Engel & Thuotte, 1978) and other organisms (Engel & Fowler, 1979).
Much of the copper in sea water, particularly in coastal and estuarine waters, is bound in organic complexes which determine its toxicity. Cadmium availability, on the other hand, is determined by inorganic complexation with the chloride ion (Sunda et al. 1978). Salinity therefore is a key factor in cadmium toxicity, perhaps explaining the general observation that LC50 values for fresh water organisms tend to be about an order of magnitude lower than those for marine organisms.

It has been shown (Fig. 3) that Chelex extraction at pH 8 and solvent extraction at pH 5 yield roughly similar levels of copper and zinc, and that these levels are not total levels (Fig. 8). The chelatable fraction must include the biologically available ionic fraction, and the metal analyses given (Fig. 5) may be expected to reflect the fraction that is available. Acidification to pH 1-3 has in some samples from the River Tamar given total copper concentrations of 12–13 µg/l, where Chelex extraction gave concentrations of only 1–2 µg/l (Cleary, unpublished). If all of the copper in these samples was present as ionic copper, it would have deleterious effects on sensitive organisms, so the amount of dissolved organic matter capable of complexing ionic copper becomes an important factor in determining biological water quality (Fig. 8).

The unavoidable limitation of the bioassay to waters of relatively high salinity restricted the survey to the waters of the Sound and the industrialized and urbanized lower reach of the Tamar, where on several occasions our experiments have shown waters capable of eliciting the stress responses of Campanularia (Fig. 4). While these data are not conclusive in demonstrating whether this is the effect of local inputs or of metals brought down from the metalliferous area to the north, two of the three experiments suggest the possibility that there may not be any further deterioration of water quality in near-surface water upstream, with the implication that urban inputs may be involved.

**Hydrographic considerations**

The bioassay data (Fig. 4) suggest that contaminants that cause the hydroid responses are not rapidly mixed, in that there are significant changes in responses from one station to the next. This is confirmed by seeing clear and stable boundaries between waters of different turbidity (Fig. 6), and marked variations in the levels of metals in water samples from adjacent stations (Fig. 5).

All the samples for bioassay and chemical analysis were taken near the surface (0.5–1 m). While some effluents discharge near the surface most discharge into deeper more saline water. It is unlikely therefore that our surface samples are representative of the water column and there are likely to be variations in water quality with depth which we are now beginning to investigate.

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REFERENCES


HYDROID RESPONSES TO METALS IN RIVER TAMAR


