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The Dynamics of Metals in the American Oyster, *Crassostrea virginica*. II. Environmental Effects[1]

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ABSTRACT: The dynamics of Mn, Fe, Zn, Cu and Cd in the shell and soft tissues of the American oyster, *Crassostrea virginica*, were observed in oysters exposed *in situ* to a metal-contaiminated environment from September, 1972 until August, 1973. Zn and Cu accumulated in soft tissues of exposed oysters reaching levels of 4100 μg Zn/gm and 450 μg Cu/gm compared to 1700 μg Zn/gm and 60 μg Cu/gm for controls (dry weight basis). The relative enhancement of metals in oyster soft tissues exposed to the contaminated environment over controls reflected the pattern of metal contamination in sediments. Although growth of the oysters, as measured by soft tissue dry weight and shell dimension, was identical, shells of exposed oysters were significantly thinner than controls (16%). Trace metal incorporation into shell was affected with Mn deposition surpressed and Fe, Zn and Cu slightly increased. Uptake of metals by oyster soft tissues was seasonally dependent with rapid uptake occurring in the summer and fall but delayed uptake occurring in the early spring.

Introduction

The uptake of metals by oyster soft tissues when exposed to environmental conditions modified by human activities has been observed in field surveys (Hiltner and Wichmann 1919; Chipman et al. 1958; McFarren et al, 1961; Pringle et al. 1968; Kopfler and Meyer 1969; Huggett et al. 1971; Windom and Smith 1972; Kopfler and Mayer 1973) and studied in the laboratory (Chipman et al. 1958; Pringle et al. 1968; Shuster and Pringle 1969; Preston 1971; Romeril 1971; Kerfoot and Jacobs 1973). These studies have provided a great deal of insight into the problem of metal accumulation, but they also have major drawbacks from a theoretical point of view. Field surveys are useful for identifying metal contamination hot spots of public health concern, but in general only provide a snapshot picture of metal dynamics. Seasonal effects have not been taken into consideration in sampling programs and the sampling of natural populations with their high level of biological variability causes difficulties in interpretation of data. On the other hand laboratory studies have a tendency to use high dose levels which make it hard to extrapolate back to the effects at environmental concentrations, and are always plagued by the question as to whether or not the physical-chemical speciation of a dosing metal is similar to that encountered by the oyster in the real world.

In order to overcome these difficulties several "field experiments" (Seymour 1966; Preston 1967; Ikuta 1968; Roosenburg 1969; Drobeck and Carpenter 1970; Wolfe 1970; Kopfler and Mayer 1973) have been carried out to study the uptake or loss of metals under real world conditions. These studies combine the laboratory approach with field conditions, thus allowing a more realistic setting for the experiment with respect to concentration and physical-chemical speciation. The major drawback to this approach is the lack of control over various conditions. However, if experimental stations are carefully chosen so as to match as many environmental factors as possible—salinity, temperature, light, food supply (both quantity and quality), etc.—these studies complement the understanding of metal cycling provided by laboratory experiments.

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The experiment reported here is a "field experiment," designated Oyster-II, designed to: (1) determine the relationship between the metal concentration in oyster soft tissues and the level of environmental contamination as measured by metal concentrations in sediments, (2) investigate the effects of excessive metal contamination on normal dynamics of metals in soft tissues and shell, and (3) investigate seasonal differences in the rate of metal uptake by oyster soft tissues in a contaminated environment.

Methods

To carry out this experiment a stock of approximately 500 genetically similar, hatchery-reared oysters were set out in plastic trays at the Smithsonian Chesapeake Bay Center for Environmental Studies (CBCES) pier, Rhode River, Maryland (Fig. 1) in June, 1972. Although the Rhode River is a small tributary of the Chesapeake Bay (3 km × 1 km) there are several distinct watersheds present. The watershed in which the CBCES pier is located is subject to little impact of human activity. The oysters at the CBCES pier were allowed to adapt to local environmental conditions for two months, at which time (September, 1972) a sub-population of 150 oysters, designated Cadle-1, was transferred to another watershed in the Rhode River system known as Cadle Creek. The Cadle Creek watershed is a highly impacted region supporting a significant human population and some light industry including a pleasure boat manufacturing company and several marinas. The degree of metal contamination in these two areas was determined by sediment analyses (see below). The two experimental stations were closely matched with respect to salinity and temperature with salinities ranging between 2 and 13 o/oo during the study period and temperatures ranging between 1 and 32 C (Cory et al. 1974). The difference in salinity and temperature between the CBCES pier and the Cadle Creek station never exceeded 5%. These two groups of oysters were sampled monthly until April 1973 when a second subpopulation of 150 oysters, designated Cadle-2, was transferred from the CBCES pier to Cadle Creek. All three groups were sampled monthly thereafter. The experiment was prematurely terminated due to high mortalities in the control group (CBCES pier) during June 1973 and in the exposed groups (Cadle Creek) during August 1973 as a result of fouling by bryozoan growth of the plastic trays in which the oysters were suspended. In spite of the fact that the seasonal dynamics were not observed for a full year cycle, information on the influence of environmental contamination on metal uptake and metabolism was obtained.

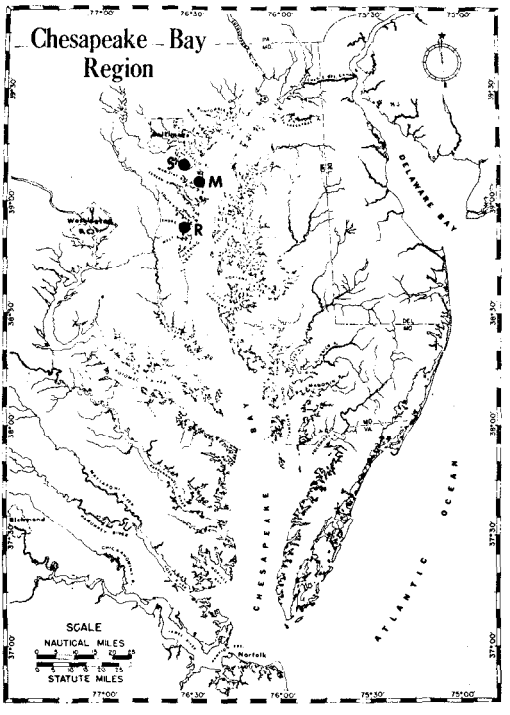


Fig. 1. Location of experimental stations in the Chesapeake Bay; R-Rhode River, M-Magothy River, and S-Stoney Creek.

Monthly samples of 10 oysters each were collected from each station and measurements of soft tissue dry weight, shell dimensions (height and width), shell weight (February–June, 1973 only) and soft tissue concentrations of Mn, Fe, Zn, Cu and Cd were made (Frazier, 1975). Early in the experiment it became obvious that the shells of oysters in Cadle Creek were quite fragile and easily broken while shucking. To obtain a quantitative measure of shell thickness we calculated the weight of shell per unit area (mg/cm^2). This was determined by dividing the shell weight by the shell surface area, SA, estimated by Galtsoff's formula (Galtsoff 1964), $SA = 2.5h^{1.56}$, where h is the shell height. To verify that this formula was valid for our

hatchery reared oysters, 18 shells were randomly selected and the surface area calculated by Galtsoff's formula was compared to the actual surface area measured by tracing the shell outline and computing the area. A linear regression of the computed surface area against the actual surface area gave $SA_{cal} = 0.946\ SA_{actual} + 1.97$ with a correlation coefficient of $r = 0.962$. Galtsoff's method of estimating surface area resulted in an error of less than 6% when compared to the actual projected surface area of the shell. The use of projected area leads to a slight underestimation of the true surface area due to the curvature of the shell. However, this bias is consistent for all samples and hence the ratio of shell weight to estimated shell surface area is a good relative indicator of shell thickness.

Sediment analyses for metals were used to evaluate environmental contamination. Four samples of sediments were collected at the CBCES pier and one station in Cadle Creek (CC2) during 1972 and a survey of 4 stations in Cadle Creek (CC1-CC4) was made on 11 August, 1972. The station at the head of Cadle Creek (CC4) was within 30m of the oyster station. Samples were collected by an Eckman dredge (top 5 cm) and transported to the laboratory in plastic bags on ice. Interstitial water was removed by vacuum filtration and samples were oven dryed to constant weight at 60 C. Since sediments from both stations were of a fine clay texture it was necessary to grind the dry sample in a porcelain mortar and pestle and filter it through a plastic screen. A 1 g aliquot of sediments passing through the filter was leached in hot, concentrated nitric acid for 4 hours and the leachate analyzed by atomic absorption spectrophotometry. Data are reported as μg metal per gm dry sediments (particle size less than 202 μ).

Shells from each of the first two oyster groups (CBCES pier and Cadle-1) were analyzed at two different times (1 February, 1973 and 7 June, 1973). Right valves were crushed in a porcelain mortar and 1 g aliquots were digested and analyzed by the same procedure as previously reported (Frazier 1975).

Results

The degree of environmental metal contamination at the two experimental stations was quantified by sediment analysis. The concentrations of metals in sediments at the CBCES pier (Table 1) are within the range of normal concentrations found in clay sediments from the Chesapeake Bay (Pheiffer 1972). The results of the Cadle Creek survey, 11 August 1972, (see Fig. 2 for Mn, Zn, and Cu data) indicates that the mouth of Cadle Creek (0–0.4 km) is characterized by normal sediments. The error bars at station CC2, mean ± one standard deviation (n = 4), give an indication of the variability in sediment analysis. As you approach the head of Cadle Creek (CC4) the system becomes progressively more contaminated with respect to Zn and Cu while Mn is not significantly affected (large natural variability). The means ± one standard deviation for the concentration of Mn, Zn and Cu at the CBCES pier are indicated on the right hand side of Fig. 2 to emphasize the degree of contamination at the Cadle Creek experimental station (30 m from CC4). The actual data at CC4 are given in Table 1.

The overall growth of the two oyster populations throughout the study period was quite similar, however, the exposed oysters in Cadle Creek showed greater fluctuations in soft tissue dry weight than controls at the CBCES pier (Fig. 3). Net growth during the fall 1972 growing season for both groups was 0.23 g which is equivalent to a 100 percent increase in soft tissue solids. Following the

TABLE 1: The concentration of metals in normalized sediments (< 202 μ particle diameter) from the Rhode River. Mean (± 1 standard error) of four samples at CBCES pier. One sample only at Cadle Creek-CC4

	Concentration (μg/gm dry wt.)				
	Mn	Fe	Zn	Cu	Cd
CBCES Pier	50 ± 29	18,800 ± 2000	48 ± 6	5.6 ± 1.8	0.3 ± 0.1
Cadle Creek-CC4	90	26,300	232	123	0.7

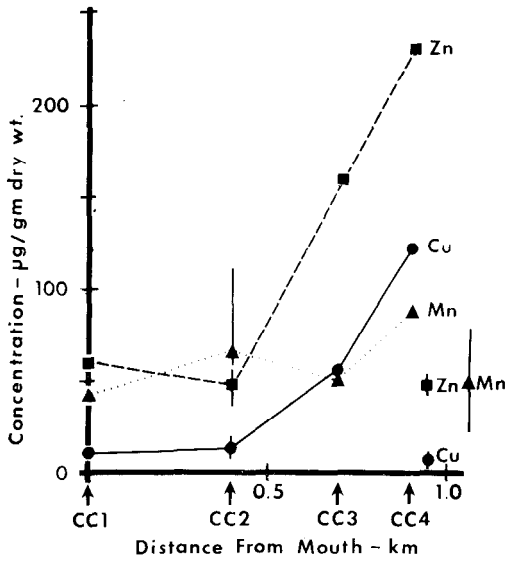


Fig. 2. Concentration of metals in sediments of Cadle Creek collected in survey of 11 August, 1972. Corresponding values for CBCES pier appear in right hand margin. Where given error bars are ± 1 standard deviation.

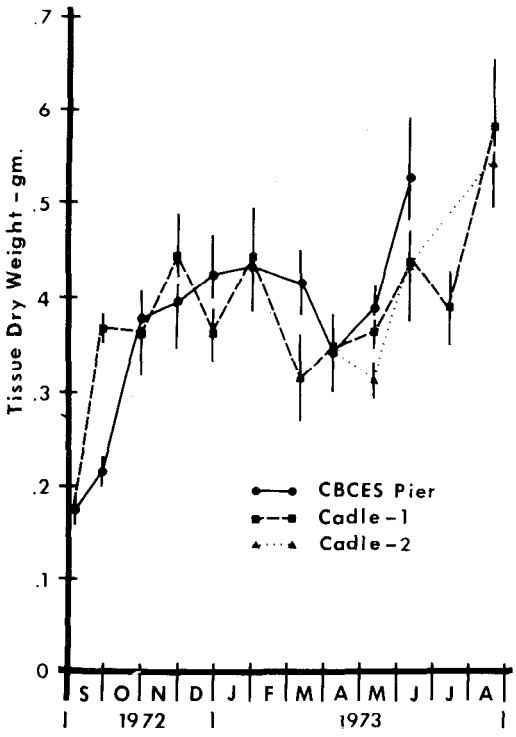


Fig. 3. Soft tissue dry weight for control (CBCES pier) and exposed oysters (Cadle-1 and -2). Mean ± 1 standard error of the mean (n = 10). Key: ●–CBCES pier, ■–Cadle-1, ▲–Cadle-2.

loss of body weight due to glycogen utilization in the winter, spring growth was equally good for all three groups. Shell growth was also similar between groups with average shell height increasing by 11 mm from 35 mm to 46 mm during the fall of 1972. Based on these two indices—soft tissue dry weight and shell dimensions—the growth of the two oyster populations was identical.

On the other hand, shell thickness (mg shell/cm^2) was significantly different between control and exposed oysters for four of the five months for which appropriate data were taken (Table 2). During February, March, April and June, 1973, the exposed oysters' shells averaged 16% thinner than controls. This quantitative measurement supports the subjective observation that the exposed oyster shells were thinner than normal.

The concentration of Mn, Fe, Zn, Cu and Cd in oyster soft tissues (dry weight basis) is given in Fig. 4 A–E. Several points should be noted concerning this data. (1) The high concentration of Mn in control oysters during October, 1972, is a real phenomenon and not the result of sample contamination. Exposed oysters did not exhibit this peaking effect. Throughout the remainder of the study period, Mn levels in control and exposed oysters closely agreed. (2) Iron concentrations for the three groups of oysters follow identical dynamics for the entire study period. (3) Zinc and copper were accumulated to great excess by Cadle-1 reaching levels of 4100 µg Zn/gm and 450 µg Cu/gm soft tissue dry weight compared to 1700 µg Zn/gm and 60 µg Cu/gm for controls during December, 1972, and January, 1973. It appeared that tissue levels of Zn and Cu reached equilibrium with available environmental levels since the Zn and Cu concentrations leveled off after 3 months. This conclusion is supported by the fact that the body burdens remained statistically constant through the winter after the initial period of rapid uptake (Fig. 5 A & B). The initial rate of Zn and Cu uptake, as determined from body burdens, is 350 µg Zn/month and 50 µg Cu/month. Cadle-2 oysters, which were transferred in April 1973, exhibited a delay of two months before rapidly accumulating Zn and Cu to levels approaching Cadle-1 (Fig. 5 A & B). This behavior differs markedly from Cadle-1

TABLE 2: Shell surface area, weight, thickenss and Δ-thickness (exposed-control) for five monthly samples (February–May, 1973). Mean ± standard error of the mean (n = 10).

Sample Date	Stock[1]	Shell Surface Area[2] (cm^2)	Shell Weight (gm)	Shell Thickness (mg/cm^2)	Thickness (% of controls)
2/1/73	C	24.9 ± 1.1	6.6 ± 0.4	267 ± 21	
	E	27.4 ± 1.2	6.1 ± 0.3	223 ± 15	−16%[3]
3/8/73	C	25.7 ± 1.2	6.8 ± 0.4	267 ± 22	
	E	21.8 ± 1.5	5.2 ± 0.4	237 ± 26	−11%[3]
4/6/73	C	22.4 ± 1.1	6.3 ± 0.5	280 ± 27	
	E	25.0 ± 1.6	5.6 ± 0.4	224 ± 22	−20%[3]
5/8/73	C	24.5 ± 0.8	7.0 ± 0.3	287 ± 16	
	E	25.4 ± 1.5	7.1 ± 0.4	277 ± 23	−3%
6/7/73	C	28.0 ± 2.4	8.8 ± 0.8	322 ± 20	
	E	28.9 ± 1.9	7.9 ± 0.5	275 ± 5	−15%[3]

[1] C—Control (CBCES pier), E—exposed (Cadle Creek).
[2] Calculated from Galstoff's formula: $SA = 2.5\ h^{1.56}$
[3] Significant by the Students t-test (1% level).

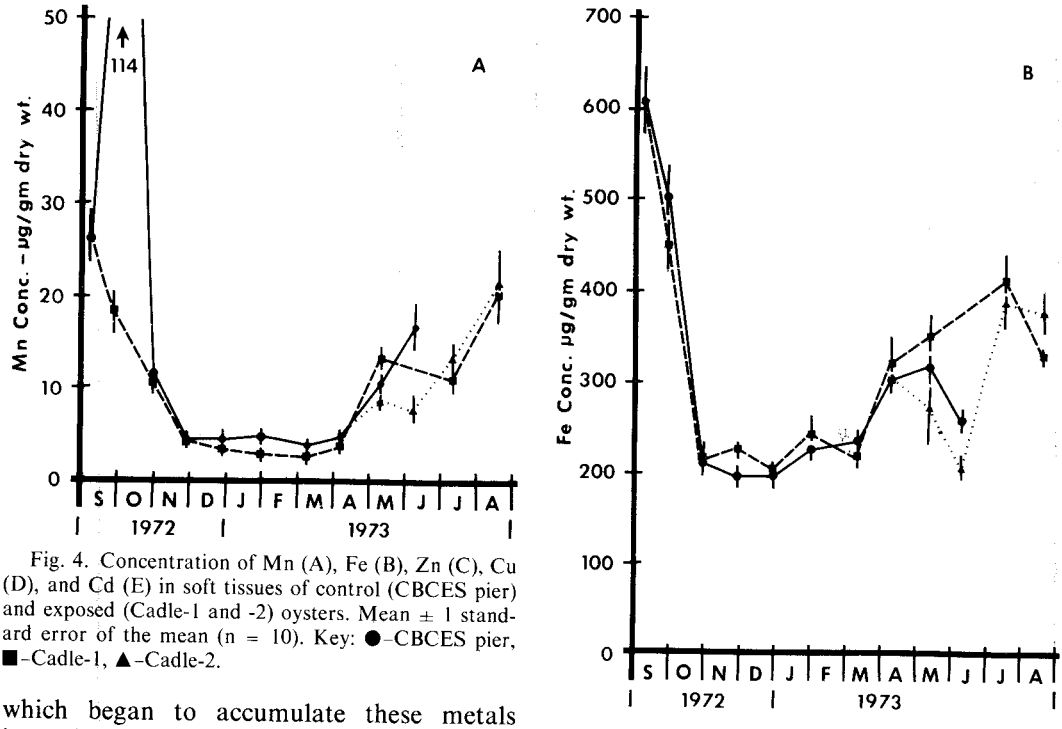
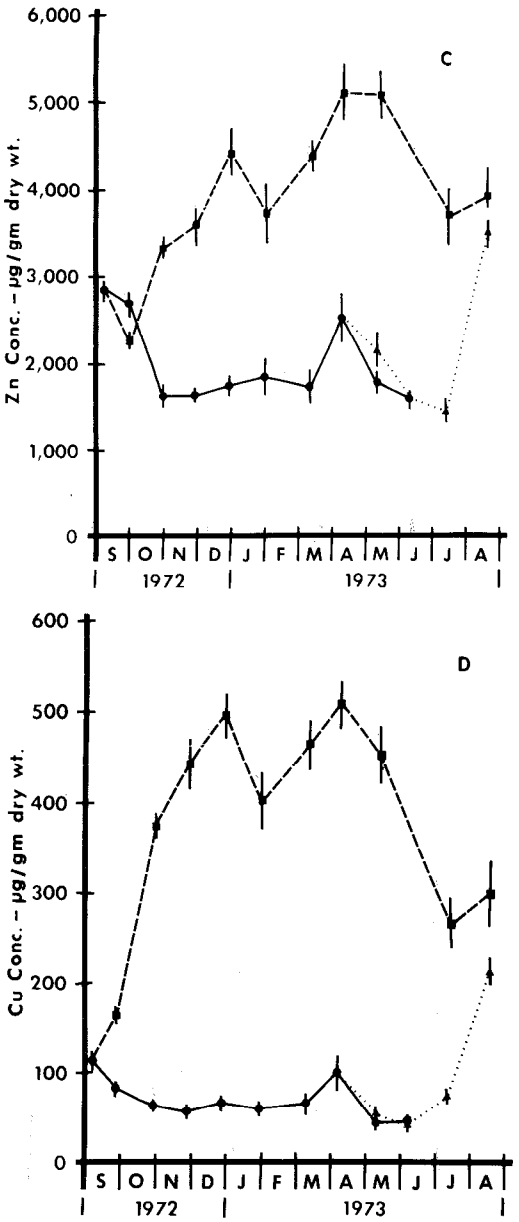
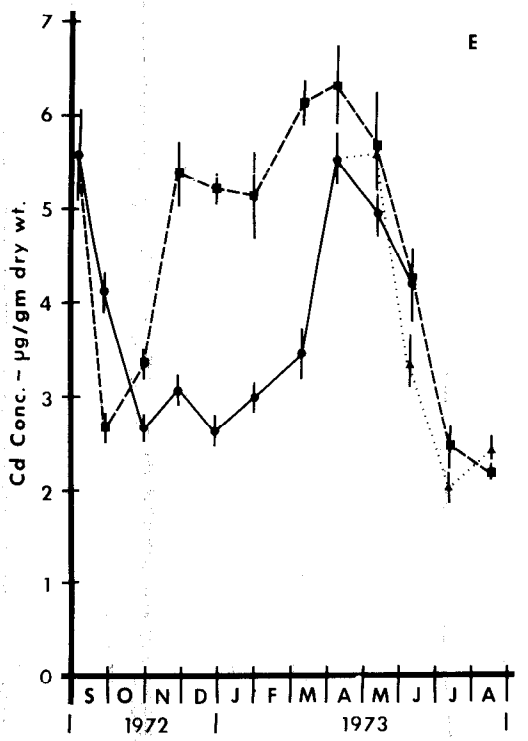


Fig. 4. Concentration of Mn (A), Fe (B), Zn (C), Cu (D), and Cd (E) in soft tissues of control (CBCES pier) and exposed (Cadle-1 and -2) oysters. Mean ± 1 standard error of the mean (n = 10). Key: ●–CBCES pier, ■–Cadle-1, ▲–Cadle-2.

which began to accumulate these metals immediately after being transferred to Cadle Creek. (4) Cadle-1 Cd concentrations increased rapidly in November, 1972, then leveled off, resulting in a significant difference between Cadle-1 and controls during the winter of 1972–73. In April control levels increased, peaking in May, then decreased along with Cadle-1 and -2.

The concentration of metals in control and Cadle-1 oyster shells are given in Table 3 along with previously reported data on oyster shells at the CBCES pier, designated Oyster I (Frazier 1975). Comparing Oyster I and Oyster II at the CBCES pier we note that Mn, Fe and Zn are significantly lower in Oyster II shells, Cu is higher and Cd is undetectable in all shells. It must be remembered that Oyster I and Oyster II are different genetic stocks of oysters and different year classes (Oyster I—1970, Oyster II—1971). When comparing Oyster II at the CBCES pier and Cadle Creek we note





that Cadle-1 shells are deficient in Mn and have elevated levels of Fe, Zn and Cu.

Discussion

The use of direct water analysis to determine estuarine water quality with respect to trace metals has been limited because of analytical and sampling problems. The analytical problems relate to the time and labor involved in accurate metal analysis at the trace levels in natural waters. The sampling problems are related to both the physical problems of collecting and handling water samples without contamination and the statistical problem of obtaining a significant sample of a fluctuating system. The use of sediment analysis as an alternative to water analysis can clearly demonstrate local sources of metal contamination (Bender et al. 1972). However, the relationship between metal concentration in sediments and water quality is complicated by the lack of understanding of the physical-chemical processes which control the partition of metals between the aqueous overlying phase and the solid sediment phase through the intermediary interstitial water. Sediment sampling is also plagued by the problem of spatial variability in the physical properties of the sediments themselves. Since metal absorption is basically a surface effect, variation in surface area per unit weight, which is dependent on the particle size distribution of the sediment, will complicate interpretation of concentration measurements. Even when these difficulties are resolved the problem still remains to relate such an analysis, metal concentrations in sediments, with biological effect either in the form of food chain contamination or as toxicity.

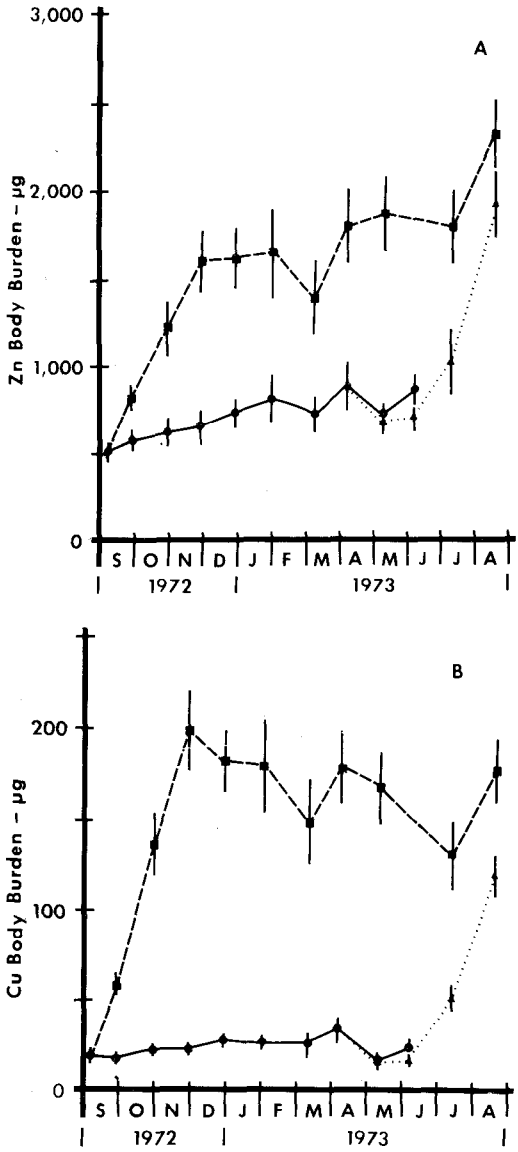


Fig. 5. Body burden of Zn (A) and Cu (B) in control (CBCES pier) and exposed (Cadle-1 and -2) oysters. Mean ± 1 standard error of the mean (n = 10). Key: ●–CBCES pier, ■–Cadle-1, ▲–Cadle-2.

In the experiment reported here, analysis of sediments clearly indicated the presence of metal contamination of the environment. Although particle size distribution in the sediments was not determined, the texture of the sediments from all stations was visually the same, a fine clay. Also, the sieving of the sediments through a 202 μ screen tends to normalize the particle size distribution by removing large particles which distort the surface area per unit weight factor. Thus the metal contamination at the head of Cadle Creek is real and not merely an artifact of sediment characterization. Furthermore, the rapid uptake of these metals by oysters located in this environment demonstrates that these metals were biologically available either directly in the water or indirectly through the food chain. If the biological availability of metals in the ecosystem is directly proportional to the normalized sediment concentration, then at equilibrium, the ratio of metal concentrations in biological tissues between any two stations should be equal to the ratio of normalized sediment concentrations between those same stations. The ratio of metal concentration in oyster soft tissues between Cadle-1 and CBCES pier was calculated from data collected in March, 1973. March was selected since it was felt that Cadle-1 soft tissues had reached equilibrium by this time. Furthermore, during the winter the soft tissue concentrations are not affected by the seasonal dynamics associated with growth, reproduction and shell metabolism (Frazier 1975). The ratios for oyster soft tissues and sediments are given in Table 4. The sediment data indicate that relative to natural levels (CBCES pier) the head of Cadle Creek is more contaminated with Cu (22 times natural levels) than Zn (4.8 times) even though the absolute concentration of Zn

TABLE 3: Concentration of metals in oyster shell.

Experiment	Station	Date	n	Concentration (μg/gm dry shell)				
				Mn	Fe	Zn	Cu	Cd
Oyster I*	CBCES Pier	3/28/73	5	520 ± 40	20 ± 13	2.1 ± 0.4	0.13 ± 0.05	<0.1
		6/7/73	5	490 ± 80	18 ± 8	3.0 ± 1.6	0.19 ± 0.09	<0.1
Oyster II*	CBCES Pier	2/1/73	4	440 ± 40	7 ± 6	2.0 ± 0.2	0.65 ± 0.32	<0.1
		6/7/73	4	380 ± 60	4 ± 3	1.8 ± 0.2	0.59 ± 0.24	<0.1
Oyster II*	Cadle Creek	2/1/73	4	360 ± 30	15 ± 11	5.3 ± 0.7	1.15 ± 0.22	<0.1
		6/7/73	4	330 ± 70	12 ± 6	4.4 ± 1.1	0.92 ± 0.25	<0.1

* Oyster I refers to the experiment reported in Frazier, 1975. Oyster II refers to this experiment.

TABLE 4: Ratio of metal concentrations at Cadle Creek to CBCES Pier for sediments, soft tissues, and shell.

	Mn	Fe	Zn	Cu	Cd
Sediments	1.8	1.4	4.8*	22*	2.2*
Soft tissues	0.6	0.9	2.5*	7.8*	1.9*
Shell	0.8*	2.5*	2.6*	1.7*	—

* Ratio significantly different from 1.

is greater than Cu in sediments. A similar result holds for oyster soft tissues although the magnitudes of the ratios differ, e.g. the ratio of Cu levels in oyster soft tissues is 7.8, not 22 as predicted by the sediment ratio. Although the ratio of metals in biological tissues is not equal to the ratio in sediments, the ordering of relative metal contamination, expressed by the Cadle Creek: CBCES pier ratio, is similar for both sediments and oyster soft tissues; $Cu>>Zn>Cd>Mn\simeq Fe$. The conclusion is that the sediment ratio is not a perfect predictor of biological tissue levels, but does indicate the relative trend in availability of metals to biota, i.e. higher metal concentrations in normalized sediments (restricted particle size distributions) indicate greater bio-availability. This trend does not necessarily hold true for systems in which the sediments are not in some form of quasi-equilibrium with overlying waters. Thus, higher concentration of metals in dredge spoils does not necessarily mean they will be biologically available when dumped in uncontaminated regions. The physical-chemical and diffusion processes which control the exchange of adsorbed metals with overlying waters and benthic biota will ultimately determine the bio-availability.

The effect of environmental pollution on oyster growth and condition has been previously noted (Galtsoff 1964; Quayle 1969). The shell thinning observed here, 16% reduction in exposed shells (Cadle-1), is another manifestation of the effects of environmental pollution. A plausible hypothesis explaining this effect is that metal contamination of oyster soft tissues interfered with the shell calcification processes occurring in mantle epithelia. The basis for this hypothesis is that several metals, including Cu and Cd, have been shown to inhibit zinc metalloenzymes involved in shell metabolism, in particular alkaline phosphatase (Applebury et al. 1970) and carbonic anhydrase (Coleman 1965). Since the concentration of both of these metals in soft tissues was elevated above control levels for a significant portion of the study period metal toxicity may have been involved in shell thinning. Another indication that normal shell metabolism has been disrupted is the fact that the levels of several metals in Cadle-1 shell are significantly different from control levels, Oyster II–CBCES pier (Table 3). If the role of trace metals in shell deposition was totally independent of environmental conditions (perfect homeostatic control), then all the metal concentration ratios in oyster shell between any two stations would be 1. The existence of such a high degree of tissue control over trace metal deposition in shell is indicated by the relationship between metal concentrations in shell and soft tissue under normal conditions; i.e. Mn is very high in shell compared to soft tissues (50 times) while Zn concentrations are low in shell compared to soft tissues (0.001 times). Thus, the mechanisms involved in trace metal incorporation into shell are highly discriminatory. The fact that the metal ratios are significantly different from 1 indicates that the local environment in Cadle Creek does have an effect on trace metal deposition in shell. However the deposition observed is not proportional to metal availability as indicated by sediment and soft tissue analysis. The order of metal concentration ratios in shells is $Zn\simeq Fe>Cu>Mn$ compared to $Cu>>Zn>Cd>Mn\simeq Fe$ for sediments and soft tissues (Table 4). The deviation of metal concentration in Cadle-1 oyster shells from control levels, and its lack of correlation with environmental exposure, may be a result of the specific failure of biological control mechanisms affecting the role of trace metals in shell metabolism as a result of the toxic effects of metal pollution.

The behavior of Mn is a good example. The deficiency in shell Mn in Cadle-1 oysters could be related to the differential behavior of Mn in soft tissues observed in the fall of 1972 (Fig. 4A). This Mn peaking phenomenon is a real effect and has been observed in other oyster populations in the Chesapeake Bay. Oysters located in the Magothy River and Stoney Creek, a tributary of the Patapsco River (Fig. 1) showed a similar peaking effect in September, 1972 (Fig. 6). Additional research is required to definitely establish the

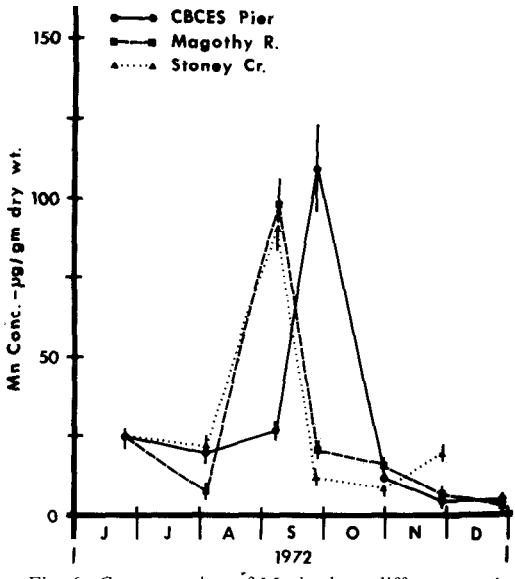


Fig. 6. Concentration of Mn in three different stocks of oysters in the Chesapeake Bay. Mean ± 1 standard error of the mean (n = 10). Key: ●–CBCES pier, ■–Magothy River, ▲–Stoney Creek.

role of Mn in relation to shell metabolism. However, the high level of Mn in shell indicates its probable importance in shell deposition.

From this discussion, it is clear that local environmental conditions in Cadle Creek affected shell thickness and trace metal deposition in shell, and these effects have been attributed to sublethal responses of the oyster to metal pollution. However, the data presented are not sufficient to rule out other possible mechanisms which could account for the observed effects. Shell thinning could result from poor environmental conditions which force the oyster to remain closed for extended periods of time during which metabolic acids accumulate in the shell cavity. Under these conditions shell dissolves, releasing CO_3^{-2} which acts as a buffer to maintain an acceptable internal pH, thus reducing shell thickness (Korringa 1952). Other agents present in polluted waters which could inhibit shell metabolism are CN^{-1} and SO_4^{-2}. The presence of these agents was not verified.

The reduction in shell thickness, by whatever mechanism, is an example of a sublethal response of an organism to environmental stress. This response can have a significant impact on the ecological distribution of the organism, not as a result of direct mortality but by reducing the resistance of the organism to predation. In this case, shell thinning will render oysters more susceptible to predation by oyster drills, boring sponges and blue crabs.

The uptake of Zn and Cu in Cadle Creek oysters appeared to be seasonally dependent. Cadle-1 oysters, which were transferred in the fall, immediately took up these metals at a rapid rate while Cadle-2, which was transferred during the early spring, experienced a 2 month delay before accumulating metals at a rate similar to Cadle-1 (Fig. 5A and B). The reason for this delay is not clear since the water temperature had already risen to the point where normal water pumping should occur (10 C). Since the high concentration of metals in oyster soft tissues is due to protein binding it may be that metal uptake was limited by protein synthesis which was still surpressed in the early spring due to low water temperatures (< 20 C). Such inducible metal binding proteins are well known in other biological systems (Shaikh and Lucas 1970; Consins 1974) and appear to act as detoxifying agents. Since the oyster is a sessile organism it would not be surprising that it has evolved a similar mechanism to protect itself from the toxic effects of metals.

Conclusions

The relationship between normalized sediment concentrations of metals and oyster soft tissue concentration is not linear. However, the ratio of normalized metal concentration in sediments between contaminated environments and natural areas can be used to predict the relative enhancement of metal concentration in oyster soft tissues.

The shells of oysters in areas contaminated with metals are significantly thinner than controls (16%). Although this effect may not be a direct consequence of metal poisoning, this sublethal response can affect the ecological distribution of the oyster by permitting increased predation. The incorporation of trace metals into shell was also affected by environmental conditions.

The uptake of metals by oyster soft tissues was dependent on the season of the year when the oysters were transferred to the contaminated environment. Uptake took place rap-

idly in the summer and fall but was delayed in the early spring.

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