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## BENTHIC ECOLOGY

Convenor: *Dr. Linda Schaffner*

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*Toward a Sustainable Coastal Watershed:  
The Chesapeake Experiment. Proceedings of a Conference  
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A RECONNAISSANCE SLED IN DEVELOPMENT FOR MONITORING THE MARINE AND ESTUARINE  
BENTHIC LANDSCAPE

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*Abstract:* The Jefferson Acousto-Optic Benthic (JAB) sled is a new benthic reconnaissance device under development at the Virginia Institute of Marine Science (VIMS). The JAB sled is designed to be towed on the seafloor. It is equipped with a still film camera with a laser array for scaling distances, two live video cameras monitored from the boat (one video camera in an angled forward orientation to view the oncoming path of the sled, one video camera in a sediment profiling prism altered to plow through the surface sediments), physicochemical sensors for the near-bottom water column conditions, and optical backscatter sensors to detect suspended material concentrations. The goals for the JAB sled are to provide detailed microscale data over large distances of the seafloor or bay bottom for characterizing the benthic landscape, its heterogeneity, and ecotones. The JAB sled will also increase the accuracy of ground truthing by providing visual and physical data from the bottom transmitted concurrently with side-scan sonar images of the bottom. The sled will create a plowed furrow through the sediments that will appear on the side-scan images. Precise correlation of side-scan images with ground-truthing images will be achieved by the appearance of this furrow line in the side-scan image. Live video of bottom surface features and bottom profile features will be viewed from the point at which the furrow image originates in the record, as will measures of dissolved oxygen, pH, temperature, salinity, suspended solids, and depth. All this will be coordinated with a ship-based global positioning system (GPS) to provide completely verifiable data on the substrate and near-bottom conditions over large tracts of the benthic landscape. The image data of the bottom will be cross-referenced with physical bottom samples that will be taken from the flat edge of the plowed furrow by divers or by coring devices equipped with a camera to view the bottom contact point.

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EUTROPHICATION AND MACROBENTHIC COMMUNITIES OF THE LOWER  
CHESAPEAKE BAY:

I. ACUTE EFFECTS OF LOW DISSOLVED OXYGEN IN THE RAPPAHANNOCK RIVER

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*Abstract:* Eutrophication and vertical stratification of the estuarine water column can result in hypoxic bottom waters (< 2.0 mg/l) and resultant reductions in benthic community parameters. Stratification is often driven by spring water flow, and we studied the effects of hypoxia in the summer of 1993 following the highest recorded river flow in the Rappahannock River. Benthic community structure was characterized at five depths (ranging from 5 to 23 m) within the mesohaline region of the river. Hydrographic parameters were monitored from March to September to examine any temporal-lag responses of the benthos to hypoxia. A threshold dissolved oxygen value was determined below which benthic community parameters were expected to be degraded due to stress.

INTRODUCTION

The occurrence of hypoxic and anoxic bottom waters in marine and estuarine systems has been reported throughout the United States (Falkowski et al. 1980, Justic et al. 1993, Officer et al. 1984) and Europe (Friligos and Zenetos 1988, Justic 1987, Rosenberg and Loo 1988) with increasing frequency in recent years. Low dissolved oxygen conditions have the potential to threaten the natural structure and functioning of benthic communities through mortality, sublethal stresses, and indirect restrictions upon habitat availability. The effects of anoxia and hypoxia can also be transferred up the food chain through trophic interactions (Pihl et al. 1992) and can interfere with the processing of organic material and nutrients in estuarine systems (Rosenberg et al., 1991). Hypoxia generally refers to dissolved oxygen concentrations below 2.0 mg/l, a level where abrupt benthic macrofaunal community changes are often observed (Harding et al. 1992, Rosenberg 1980). Anoxia is defined as the absence of oxygen in the water column.

In Chesapeake Bay, stratigraphic evidence demonstrates an inherent long-term pattern of periodic anoxia in the deeper waters of the mainstem as benthic oxygen demands exceed bottom-water reaeration

processes during the summer months owing to the presence of a natural pycnocline (Breitburg 1990, Cooper and Brush 1992, Harding et al. 1992, Schaffner et al. 1992). The greatly increased areal extent and volume of the anoxic water mass in the Bay since the middle of this century has been attributed to the anthropogenic impacts of eutrophication (Cooper and Brush 1993, Taft et al. 1980). Excessive anthropogenic nutrient loadings into the Bay can greatly enhance bottom-water and benthic microheterotrophic oxygen demands, which can in turn stimulate earlier and more extensive occurrences of the oxygen-depleted water mass. Hypoxic and anoxic conditions presently occur in the deeper portions of the Bay and its tributaries on an annual basis (Officer et al. 1984).

In the Rappahannock River, a tributary of Chesapeake Bay, annual summertime anoxia occurs in the deep basin located just inside the river's mouth. Although intrusion of anoxic bottom waters from the Bay's mainstem can increase the severity and duration of low dissolved oxygen conditions in the lower Rappahannock River, the seasonal occurrences of hypoxia in the deep basin are independent of mainstem patterns (Kuo et al. 1991).

Because of a weak longitudinal salinity gradient, and thus gravitational circulation, the Rappahannock River experiences annual low dissolved oxygen conditions that are absent from other tributaries of the Lower Bay (e.g., the James River), which receive higher nutrient loadings but maintain stronger salinity-based circulation patterns (Kuo and Neilson, 1987).

Current benthic monitoring of macroinfaunal responses to seasonal low dissolved oxygen in the mesohaline regions of the Rappahannock River is carried out in the deepest portion of the anoxic basin at Virginia Benthic Biological Monitoring Stations LE3.2 and LE3.4 (see Dauer, 1993b for further information). Because the deeper areas of the Bay and its tributaries have a natural propensity to become anoxic owing to physical factors, the deepest areas are likely to experience oxygen stress regardless of anthropogenic influences. In light of this, the present benthic monitoring station in the lower Rappahannock River may be too deep to provide independent information on the expected water quality and biotic changes associated with Chesapeake Bay restoration efforts. Regardless of changing environmental quality, the benthos of this area may always experience low dissolved oxygen concentrations during the summer months.

The purpose of this study is to document

intermediate levels of hypoxia and the subsequent benthic invertebrate responses along the depth gradient extending upward from the deep water monitoring station. Shallower depths within the lower Rappahannock River basin that are presently under low dissolved oxygen stress may be more indicative of long-term changes in water quality as they are located in that area of the bottom where hypoxia is primarily dependent upon eutrophication influences. The shallowest areas along this depth gradient can serve as within-habitat reference sites to compare with the patterns observed in benthic community responses to naturally occurring hypoxic and anoxic conditions within the deeper depths of the basin.

#### METHODS

Macrobenthic communities were randomly sampled during March, June, August, and September of 1993 along five equidistant water depths extending upward from the Virginia Benthic Biological Monitoring Station LE3.4 (mean water depth of 21 m) located in the deep basin just inside the mouth of the Rappahannock River (figure 1). These dates were chosen to bracket the low dissolved oxygen events that occur in the Rappahannock River and to

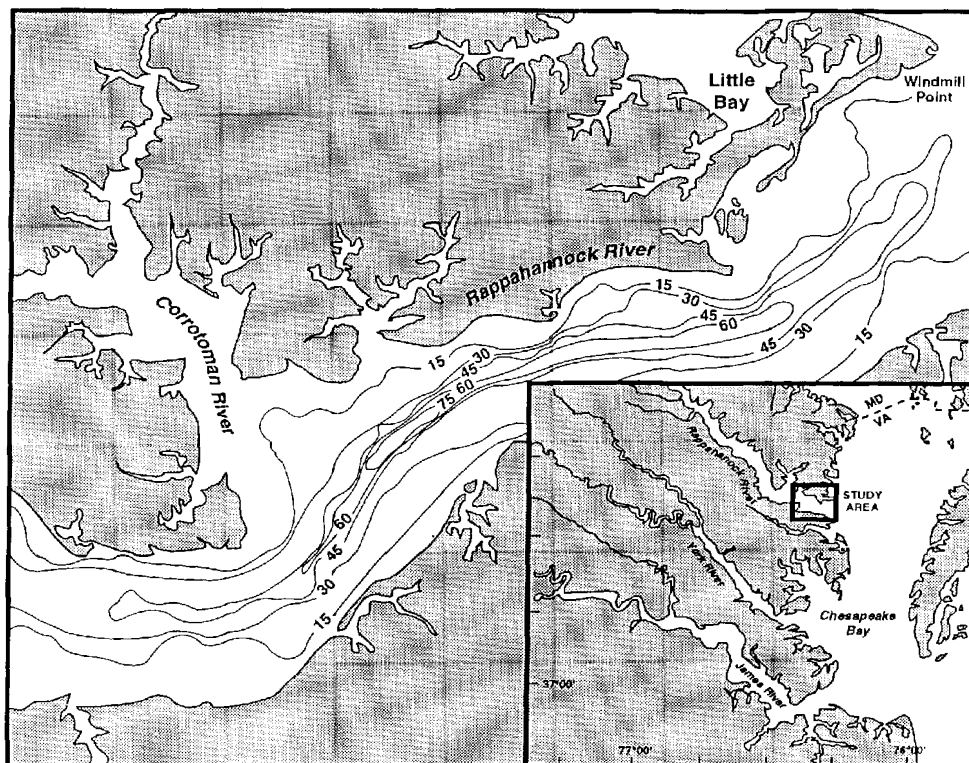


Figure 1. Sampling location on the lower Rappahannock River. Depth contours, in feet, correspond roughly to 5, 10, 15, 20, and 25 m

sample during their greatest areal extent in early August when water temperatures are greatest (Kuo and Neilson 1987).

Five replicate sediment cores were collected during June, August, and September at each depth (5, 10, 15, 20, and 25 m) with a box coring device (surface area of 184 cm<sup>2</sup>) that samples to a depth of 25 cm. March samples consisted of three replicate cores taken at each depth excluding the 5 m sampling station. Each core from the June, August, and September sampling dates was separated into 0-5 cm and 5-25 cm depth partitions to quantify macrofaunal depth distributions. After removing a standardized subsample for grain size and total organic carbon analyses, sediment cores were sieved in the field through a 0.5 mm screen to retain macroinvertebrate fauna, then relaxed in diluted 70.0% isopropanol, and preserved in a buffered, 5.0% formalin to seawater, rose bengal solution. Organisms retained were identified to the lowest possible taxon and enumerated. Biomass measures were expressed as ash-free dry weight (AFDW).

Hydrographic data, including temperature, salinity, and dissolved oxygen, were monitored at 5 m depth increments throughout the basin on a biweekly basis from June through September using daily calibrated YSI Model 33 salinity-conductivity-temperature and YSI Model 58 dissolved oxygen meters.

A two-way multivariate analysis of variance (MANOVA) was performed on ranked data to test for overall temporal and depth-related effects on benthic community parameters. A Tukey's Studentized Range multiple comparison procedure was used in conjunction with the MANOVA to locate significant differences between biotic variables at each sampling depth and time. A one-way analysis of variance (ANOVA) was combined with a Tukey's Studentized multiple comparison test and performed on squared-transformed data to test for significant differences in sediment grain size between sampling depths.

## RESULTS

### Dissolved Oxygen

Hypoxic conditions were first observed in the deep basin at depths greater than 15 m in mid-May of 1993. This early event of low dissolved oxygen concentration (0.8 mg/l) coincided with large temperature and salinity discontinuities observed between 10 m and 15 m on May 20. This intense

stratification was possibly related to the heavy river flow observed in the Rappahannock River in the spring of 1993. The U. S. Geological Survey (USGS) reported a spring freshet of 15,973 ft<sup>3</sup>/sec between February and March of 1993. The long-term average (1985-1991) for this time period is 5,069 ft<sup>3</sup>/sec. The relationship between freshwater flow and dissolved oxygen concentration is poorly understood in general, however. At the shallowest, 5 m sampling depth, hypoxic conditions (0.2 mg/l) were observed on only one date, July 12. All other oxygen concentrations at this depth remained above 3.0 mg/l throughout the summer. Hypoxic conditions below 1.3 mg/l occurred continuously at the 10 m depth from June 24 through July 20. Dissolved oxygen remained low (< 0.9 mg/l) from June 24 until August 19 at all depths greater than 15 m. Surface values of dissolved oxygen were never recorded below 6.2 mg/l.

### Sediments

Sediments at the shallowest, (5 m) station were significantly different from all other depths for both grain size (ANOVA,  $F=49.48$ ,  $p < 0.0001$ ) and total volatiles, a measure of organic carbon (ANOVA,  $F = 17.34$ ,  $p < 0.0001$ ). Silt-clay percentages at the 5 m depth varied widely around a mean of 36.50% (+/- 27.24), while sediments among all deeper depths ranged between 96.18% and 99.00% silt-clay. Total organic carbon averaged 3.62% (+/- 2.41) at 5 m while volatiles for all deeper sampling depths combined ranged from 8.80% to 10.66%.

### Benthic Macrofaunal Responses

Over the course of the study, 45 species of benthic macroinvertebrates were collected and identified. Of these, only three rare species, *Molgula luteolenta*, *Neonysis americana*, and *Sabella microphthalma*, were excluded from analysis owing to their epifaunal nature. Of the 42 infaunal species retained for further analysis, 38.10% (16 species) could be classified as either opportunistic or equilibrium taxa following Dauer et al., 1992 and Ranasinghe et al. (1993). These 16 species accounted for 81.50% of the biomass and 86.90% of the abundance of all fauna collected.

Four measures of macrobenthic community structure—species richness, biomass, abundance, and Shannon Weaver's diversity measure ( $H'$  calculated to the base 2), were all significantly different (MANOVA,  $p < 0.0001$ ) for overall time,

depth, and interaction effects for all multivariate test criteria (Wilks' Lambda, Pillai's Trace, Hotelling-Lawley Trace, and Roy's Greatest Root). A summary of the Tukey's Studentized Range Test used to follow up the MANOVA results of depth effects on ranked community data is shown in table 1. All of the benthic community measures differed significantly among months of the study. Regardless of the biotic variable measured, the 5 m sampling depth was always significantly different from the deeper stations.

Benthic communities displayed a systematic response to the low dissolved oxygen conditions observed throughout the summer of 1993. The four biotic community variables measured (i.e., species abundance, biomass, richness, and diversity) all responded similarly to hypoxia throughout the summer. The degradation of infaunal abundance and biomass at each sampling depth shown in table 2 is representative of all the variables measured. Both macroinfaunal densities and biomass declined at all depths deeper than 15 m in June in response to the early deep water hypoxia event of May 20; the communities at 5 m and 10 m were unaffected. By August, communities at all depths sampled displayed degraded community structure in response to the hypoxia observed throughout the water column during the preceding month.

Table 1. Results of Tukey's Studentized Range Test (performed on ranked data) reproduced with raw data for overall depth effects. Values are means ( $\pm 1$  standard deviation) among months of the sampling period. Depths with different letters are significantly different from each other at  $p = 0.05$  for each biotic variable.

DEPTH (m)	ABUNDANCE (ind/m <sup>2</sup> )	BIOMASS (g/m <sup>2</sup> )	DIVERSITY (H')	RICHNESS (spp/rep)
5	6807.67 (10324.95) A	5.920 (8.42) A	1.44 (0.38) A	9.87 (6.36) A
10	947.50 (818.82) B	0.387 (0.32) B	0.76 (0.55) B	4.08 (3.03) B
15	551.50 (762.76) C	0.892 (1.38) BC	0.74 (0.76) B	3.52 (3.74) BC
25	1026.50 (1699.07) C	0.360 (0.50) BC	0.65 (0.65) BC	3.97 (4.57) BC
20	670.75 (1144.15) C	0.262 (0.39) C	0.22 (0.27) C	2.05 (2.02) C

Table 2. Summary of dissolved oxygen and biotic measures at each sampling depth before and after the occurrence of summer low dissolved oxygen. Values represent means ( $\pm 1$  standard deviation). Pre-hypoxia abundance, biomass, and percent of opportunistic biomass at the 5 m sampling depth reflect June measures. Density less than 5 cm refers to the frequency of individuals occurring at less than 5 cm vertical depth in the sediment. As March values are not available for this measure, the June data are presented. This represents pre-hypoxia values for the 5 m and 10 m sampling depths and during-hypoxia measures for the deeper depths. Post-hypoxia values reflect refaunation by small, shallow-dwelling opportunistic species.

DEPTH	MIN. DO RECORDED (mg/l)	FREQ. OF RECORDED HYPOXIA	ABUNDANCE (ind/m <sup>2</sup> )		BIOMASS (g/m <sup>2</sup> )		PERCENT OPPORTUNISTIC BIOMASS		PERCENT DENSITY <5cm	
			Pre-hypoxia (March)	Post-hypoxia (September)	Pre-hypoxia (March)	Post-hypoxia (September)	Pre-hypoxia (March)	Post-hypoxia (September)	Pre-hypoxia (June)	Post-hypoxia (September)
5 m	0.2	5.56%	18729(2932)	721(381)	15.615(3.201)	0.504(0.416)	15.06(5.25)	74.88(14.6)	91.34(2.21)	94.44(7.86)
10 m	0.2	23.53%	1775(749)	321(132)	0.687(0.262)	0.138(0.087)	71.81(17.5)	82.00(24.9)	71.01(17.04)	90.00(22.36)
15 m	0.1	50.00%	1679(1426)	218(159)	2.938(3.428)	0.115(0.070)	41.08(34.71)	56.67(36.51)	94.00(13.42)	100.00(0)
20 m	0.1	60.00%	2385(345)	126(130)	0.840(0.231)	0.092(0.087)	77.05(25.72)	30.00(29.82)	100.00(0)	100.00(0)
25 m	0.1	60.00%	3568(725)	161(74)	1.107(0.630)	0.149(0.065)	61.03(4.93)	78.33(21.73)	100.00(0)	100.00(0)

Abundance dominants shifted at the 5 m station after the hypoxic event of July 12. In June, the community at 5 m was dominated by the large (5-15 mm), deep-dwelling *Macoma balthica* bivalve, the bivalve *Mulinia lateralis*, and by a dense population of the polychaete *Streblospio benedicti*. By August, all of these species declined in abundance. August communities at 5 m contained a more even distribution of opportunistic polychaetes than in June, including *Glycinde solitaria*, *Nereis succinea*, and *Paraprionospio pinnata*. Although *Macoma balthica* dominated biomass in June samples collected at the 10 m and 15 m depths, it did not occur in dense populations. As with the 5 m depth, no *Macoma balthica* were found at these intermediate depths after the June sampling date, while the communities at 20 m and 25 m were dominated by opportunistic fauna both before and after the hypoxic conditions of the summer months (table 2). Species that were key in the September recovery seen at all depths greater than 5 m include *Paraprionospio pinnata*, *Streblospio benedicti*, and the roving gastropod *Rictaxis punctostriatus*.

Vertical depth distributions of benthic infauna also changed at the shallow and intermediate depths in response to summer low dissolved oxygen conditions (table 2). From June to September, the percentage of individuals found in the upper 5 cm of sediment increased at the 5, 10, and 15 m sampling depths. Neither the 20 m or 25 m sampling depths contained fauna inhabiting sediments deeper than 5 cm at any time during the study.

## DISCUSSION

Direct comparison of the 5 m station with all deeper stations was not possible. The sedimentary differences seen between the shallowest station and the deeper areas (36.5% versus >96.0% silt-clay, respectively) precluded using the 5 m data as a within-habitat reference. The impact of sedimentary differences among stations upon the benthic communities could not be accounted for statistically by an analysis of covariance (ANCOVA) owing to extreme violations of the assumptions underlying the test. The influence of sediment grain size on benthic macrofaunal community composition and structure is well documented (Bloom et al. 1972, Fresi et al. 1983, Gaston and Nasci) 1988). The appearance of a more diverse community with denser, larger fauna at 5 m can

be explained to some extent, wholly on the basis of sediment differences. Sandier sediments often support more diverse communities than do finer-grained habitats owing to an increased availability of niche space for epifaunal and deposit-feeding infaunal species (Boesch 1973, Hyland et al. 1991). Additionally, because of increased permeability and porosity, sandier sediments typically display deeper RPD (redox potential discontinuity) layers than do finer-grained habitats owing to a larger area open to the downward diffusion of oxygen. However, anoxic and hypoxic conditions can overwhelm this benefit of high permeability found in sandy sediments (Levin et al. 1991).

Although the occurrence of hypoxic conditions during the summer of 1993 caused obvious degradation of deepwater benthic communities within the lower Rappahannock River, it is difficult to separate the possible effects of predation pressure. Benthic feeding vertebrate and invertebrate predators are common in the mesohaline sections of Chesapeake Bay and its tributaries. These species include spot (*Leiostomus xanthurus*), Atlantic croaker (*Micropogon undulatus*), winter flounder (*Paralichthys dentatus*), hogchoker (*Trinectes maculatus*), blue crab (*Callinectes sapidus*), and the mud crab (*Rhithropanopeus harrisi*) (Holland et al. 1980). Typical predation pressure by these species upon the benthic macrofauna that were dominant during June on the shallow shoals of the Rappahannock River's deep basin is greatest during the summer months, and high macroinvertebrate mortalities are common (Baird and Ulwanowicz 1989, Dauer et al. 1982, Vimstein 1977).

Predation was more likely responsible for benthic mortalities at the 5 m depth than at deeper stations, however. The predaceous species listed above are all highly mobile, and could have been driven upward from the deeper depths of the basin by the earlier and more severe hypoxic conditions occurring there. Crabs and fish routinely migrate out of areas experiencing low dissolved oxygen conditions and onto the shoals where conditions are more favorable (Breitburg 1992, Diaz et al. 1992, Officer et al. 1984, Pihl et al. 1992). The progressive shoreward movement of commercial crab pots placed in the lower Rappahannock River during the summer of 1993 may reflect such a migration by blue crabs (Smith 1994). Thus, shallow-water benthic communities may experience uncommonly higher predation pressure when deeper waters are under low dissolved oxygen stress.

Many infaunal species, including the June dominant at the 5 m depth, *Streblospio benedicti*, are known to cease activity and lie prone on the sediment surface during times of moderate oxygen stress (Diaz et al. 1992, Jorgensen 1980, Llanso 1991, Rosenberg et al. 1991). Short-term hypoxia at the 5 m depth may have imposed a sublethal stress upon the benthic infauna, which may have driven deep-dwelling forms to the surface and placed them under even greater risk of predation (Pihl et al. 1992). Therefore, the depletion of macrobenthic infauna at the 5 m depth was likely a combination of intermediate physiological stress from low dissolved oxygen conditions and the indirect effect of greatly increased predation pressure as infauna were forced to abandon their depth refuge during a time of high predator density.

Predation was probably not responsible for the mortalities seen throughout the summer at the deeper depths sampled in this study, however. Predation pressure alone is not sufficient to cause the total habitat defaunation (Marsh and Tenore 1990) observed at deeper stations in the Rappahannock River, and as mentioned above, mobile predators tend to abandon oxygen-stressed bottom waters. Dissolved oxygen values at the 10 m depth remained hypoxic for 27 consecutive days, while oxygen concentrations at deeper depths remained below 2.0 mg/l for at least 57 consecutive days, a time period well beyond the tolerance of benthic invertebrates (Rosenberg et al. 1991). In addition, at all stations located deeper than 10 m intact gaping bivalve shells (*Mulinia lateralis*) were found during and immediately following low dissolved oxygen events, indicating that these animals did not die because of ingestion by predators, but through physiological stress directly relating to the hypoxic conditions (Jorgensen 1980, Rosenberg 1980). Few bivalve shells in this condition were observed at the shallowest station. Finally, macrobenthic community density declined with the increasing severity and duration of hypoxia in each depth stratum, indicating that low dissolved oxygen affected the deeper communities directly.

The dominance at the two deepest sampling depths (20 m and 25 m) by opportunistic fauna and the lack of any species inhabiting those sediments below 5 cm (table 2) suggest that these communities are degraded not only each summer following hypoxic conditions, but all year long. Larger, more deeply dwelling equilibrium species cannot become established in these communities before the area is again subjected to summertime hypoxia. Depauperate sediments dominated by

shallow-dwelling opportunistic species are common in degraded macrobenthic communities in Chesapeake Bay and elsewhere (Dauer 1993a, Dauer et al. 1992, Gray 1979, Pearson and Rosenberg 1978, Weston 1990).

Yet, the opportunistic fauna found in the deep basin of the lower Rappahannock River are by no means immune to the effects of low oxygen stress. Although their tolerances may be higher than those of typical equilibrium species, their adaptation to environments subject to hypoxia and anoxia is often related not to their anaerobic resistance but to their life history characteristics (Gray 1979). These species are able to colonize azoic or underpopulated areas quickly owing to their rapid reproductive rates, their ubiquitous nature (high dispersal capabilities), and large brood size (Grassle and Grassle 1974).

Two resilient, if not resistant, species, *S. benedicti* and *P. pinnata*, were primarily responsible for the refaunation of the benthic habitat in the Rappahannock River following summer hypoxia (Smith 1994). Influxes of opportunistic spionid polychaetes directly following anoxia events is common in Chesapeake Bay and other systems (Friligos and Zenetos 1988, Holland et al. 1977, Holland et al. 1980). These species are both highly adapted to continual disturbance and fall into Gray's (1979) r-strategist classification, where ecological success in polluted areas is determined not by tolerance but by an inherent ability to quickly colonize and dominate new areas of the benthos.

In addition to larval recruitment, some of the fauna replenishing the deeper depths following hypoxia events may have emigrated from the shallow shoals along the basin. Adult migration to newly opened habitat has been described before for both benthic predators (Pihl et al. 1992) and invertebrate macrofauna (Holland et al. 1977, Llanso 1991). Dauer and Simon (1976) found that the initial adult colonizers following a natural defaunation of benthic habitat included several opportunistic polychaetes also important in this study, including *Eteone heteropoda*, *Nereis succinea*, *Paraprionospio pinnata*, and *Polydora ligni*. Two bivalve species common in the lower Rappahannock River, *Macoma balthica* and *Mulinia lateralis*, recruit in late fall to early winter; any replenishment of these species in the shallows and mid-water sediments would not have been observed unless sampling had continued past September (Boesch 1973, Holland et al. 1987).

## CONCLUSION

Several lines of evidence suggest that benthic communities found deeper than 20 m in the deep basin are depressed throughout the year in response to annual low dissolved oxygen stress and can therefore provide little or no information relating to overall water quality and environmental restoration goals. Support for this theory includes the observed patterns in basic community measures including infaunal density and diversity, the dominance at these depths by opportunistic fauna, and the lack of any species inhabiting vertical depths greater than 5 cm. These communities likely reside in the area of the benthos that is under anoxia stress owing to natural, not anthropogenic, forces.

Communities inhabiting the intermediate depths of this study (10 m and 15 m) should be examined more

intensively and monitored on a long-term basis as they are exposed to low dissolved oxygen stress that is more likely linked to anthropogenic eutrophication. Benthic communities at both of these depths displayed characteristic changes in the depth distribution and dominance of opportunistic taxa relating to the hypoxic conditions of 1993. Long-term measures of community structure at these depths may show interannual variability that can be directly related to within-year nutrient loadings or river flow. Comparisons of benthic community structure between intermediate depths (10-15 m) and the deeper water monitoring station LE3.4 may allow assessment of the relative contributions and effects of natural physical factors and anthropogenic eutrophication to the annual summer hypoxic events of the lower Rappahannock River.

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EUTROPHICATION AND MACROBENTHIC COMMUNITIES OF THE LOWER CHESAPEAKE BAY:  
II. CHRONIC EFFECTS OF ORGANIC ENRICHMENT IN POCOMOKE SOUND

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**Abstract:** The detection of the effects of organic enrichment on benthic communities is problematic, particularly at intermediate levels of enrichment. The species, abundance, and biomass (SAB) curves of Pearson and Rosenberg (1978) empirically model patterns of organic enrichment but do not present testable hypotheses. To evaluate the effects of chronic organic enrichment on benthic community structure, a field study was conducted in Pocomoke Sound in the vicinity of a seafood processing plant. Macrobenthic communities affected by chronic organic enrichment are hypothesized to have heterogeneous and discontinuous distributions with distance from the source of enrichment. The Two-Termed Local Quadrat Variance Method and the Split Window Distance Method were used to test this hypothesis. Results were compared to the SAB curves.

INTRODUCTION

The environmental condition of Chesapeake Bay and its tributaries has deteriorated in the last 50 years, and this is attributable primarily to increases in eutrophication and toxic substances. Eutrophication from nutrient enrichment from agricultural runoff and other nonpoint sources has produced organic enrichment of the Bay. Enrichment is defined as a level of organic enrichment that changes the structure and function of the ecosystem (Pearson and Rosenberg 1978). Such an enrichment has occurred in Chesapeake Bay as a result of increased inputs of inorganic and organic materials associated with the development of the Bay's watershed. Research efforts in the Bay have documented the well-known effects of organic enrichment, including low dissolved oxygen events (Taft et al. 1980, Officer et al. 1984, Kuo and Neilson 1987) and changes in the benthic community structure (Holland et al. 1977, Dauer et al. 1992).

Changes in the macrobenthic community structure are used to monitor biological effects of organic pollutants on subtidal ecosystems (Pearson and Rosenberg 1978, Warwick 1988, Gray et al. 1988).

Macrobenthic community structure is often used to indicate environmental health because benthic animals have relatively long life spans (integrate water and sediment quality conditions over time) and sedentary life styles (cannot avoid deteriorating water or sediment quality conditions) (Bilyard 1987). The basic parameters in all benthic survey studies are the number of species, community abundance and community biomass. There are two popular descriptive measures available to assess disturbance or stress using benthic community structure - the species, abundance and biomass curves or SAB Curves of Pearson and Rosenberg 1978 and the abundance biomass comparison method or ABC Method of Warwick 1986.

The Pearson and Rosenberg (1978) model (PRM) (figure 1) is empirically derived and can be applied specifically to eutrophication effects. The PRM concludes that organic enrichment of a certain magnitude will superimpose its own gradient on the environment and induce modifications of the distributions of organisms initially controlled by salinity and temperature (Pearson and Rosenberg 1978). The X axis of the model is

defined spatially as distance from the source of organic enrichment or temporally as the time since the enrichment event or cessation of the source of enrichment. The model defines three successional stages: (1) the peak of opportunists where a few characteristic species are found in high abundance, (2) the ecotone point where species abundance declines steeply and the number of species increases steeply, and (3) the transition zone where populations flux and progress toward a more stable "normal" community. Ecotone is used in the model as a descriptive parallel to mean change within a community along a gradient of organic enrichment instead of the traditional definition of a change between two or more diverse communities.

The purpose of this study was to test the empirical PRM at a site on the eastern shore of Virginia's Chesapeake Bay in the low mesohaline region of Pocomoke Sound. A seafood processing plant located at Pig Point on Pocomoke Sound was identified as the potential organic enrichment source, and benthic community structure in the vicinity of the plant was used to spatially test the PRM. Two additional sites in Pocomoke Sound were also sampled as reference sites for comparison with the Pig Point site. Two additional spatial tests- including the Two Tailed Local Quadrat Variance (TTLQV) Test and the Split Moving

comparing the results of the SAB curves applied to data at the Pig Point location to results obtained from the two reference locations. The alternative hypothesis is that the Pig Point data will resemble the PRM distribution.

Pocomoke Sound is a large shallow embayment located on the Eastern Shore of Chesapeake Bay (figure 2). The Pocomoke River and a number of small tidal creeks feed into Pocomoke Sound. The seafood processing plant is located near Saxis, Virginia, and is the only recorded point source of organic material input in Pocomoke Sound (Virginia Department of Environmental Quality). The plant, located on a small cape called Pig Point, mechanically processes ocean quahogs and oysters and has a direct discharge into a narrow and shallow channel that drains directly into Pocomoke Sound. A secondary discharge from the boiler room was identified on the dock side of the property just beyond the end of the channel. The seafood processing plant has a reported flow rate of 0.05 million gallons per day, and an average biological oxygen demand (BOD) of 1417.8 lb/day. BOD values are in repeated violation of the Virginia Department of Environmental Quality permit requirements (90ng/l) and consequently the plant was forced to pay civil penalties and submit a waste water treatment plan.

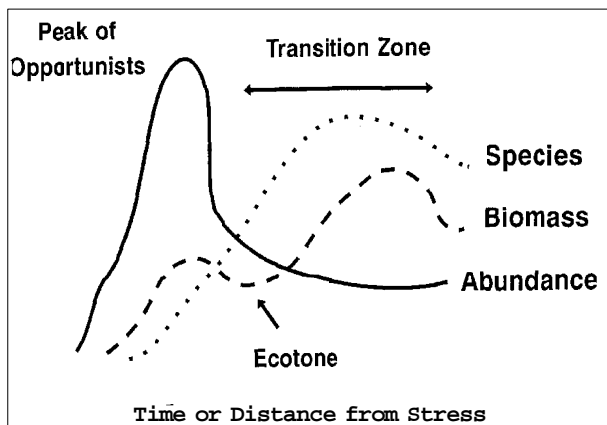


Figure 1. SAB curves along a gradient of organic enrichment. Modified from Pearson and Rosenberg (1978).

Window (SMW) (with Monte Carlo randomization tests- will be applied in future analytical procedures as additional methods to evaluate the spatial community pattern from this unreplicated sample design. The null hypothesis of this study was that macrobenthic community structure in the vicinity of the seafood processing plant was not altered by organic enrichment. The hypothesis was tested by

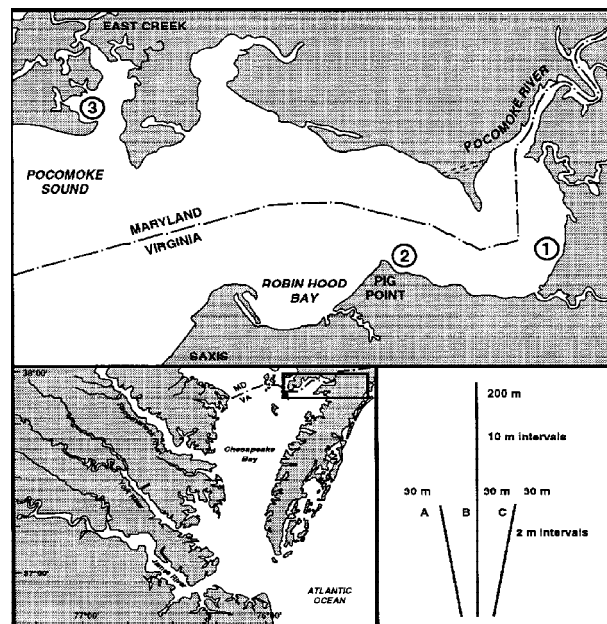


Figure 2. Location of study sites and the sample design. Top: Pocomoke Sound showing the three sites. Site 2 is the seafood processing plant location and site 1 and 3 are the reference site. Lower left: lower Chesapeake Bay. Lower right: orientation, length of transects, and sampling intervals.

In addition, two reference locations were chosen in Pocomoke Sound based on sediment similarity and presumed lack of direct effect from the seafood processing plant. One location (site 3) was located near a wildlife preserve and the other near the Pocomoke River (site 1). Both reference areas have small tidal inlets physically similar to the canal at the Pig Point site.

## METHODS

### Sampling Procedure

The sampling design consisted of three transects oriented from the shoreline to offshore (figure 2). Each group of three transects (A, B, C) was composed of 15 unreplicated samples spaced 2 m apart for a total transect length of 30 m. In addition, one of the transects (B) was extended to include an additional 17 stations spaced 10 m apart for a total transect length of 200 m. Transects started approximately 10 m from the end of the canal. A total of 195 benthic samples were collected from all three locations with the use of a hand-held coring device with a surface area of 81 cm<sup>2</sup> sampled to a depth of 25 cm. Transects were laid out with the use of wooden poles and line marked off in 2 m and 10 m intervals. In this study, only the results for the B transect (200 m length) at sites 2 (Pig Point) and 3 are presented.

### Benthic Sampling

Biological samples were sieved through a 0.5 mm sieve screen and the material retained on the screen was washed into cloth bags and stored in a 10% rose bengal and formalin solution. Samples were sorted in the laboratory and identified to the lowest possible taxonomic level. Organisms were counted for abundance estimates and biomass was calculated as ash-free dry weight (AFDW). AFDW measurements were obtained by taking the difference of the initial oven dry weight (24 hr at 60°C) and the ash weight (4 hr at 550°C). For the SAB curves of the PRM, the species curve was plotted as number of species per sample, the abundance curve as number of individuals per m<sup>2</sup>, and the biomass curve as AFDW per m<sup>2</sup> against distance along the transect.

Sediment samples of 50 g were also taken at each station location for sediment grain size and frequency distribution and for organic content. Silt-clay and sand fractions were separated by wet sieving the sediment through a 63 µm screen. The sand and silt-clay fractions were analyzed sepa-

ately using the techniques of Folk (1974). The percentage of sand, silt, and clay, phi size frequency, and sorting coefficients were calculated. Total organic carbon (TOC) of the sediment was estimated from a 5 g sample and was calculated as the AFDW of the sediment expressed as a percentage of the dry weight of the sediment.

## RESULTS

### SAB Curves

The predicted SAB curves of the PRM (Pearson and Rosenberg 1978) were compared with SAB curves calculated from the B transects (200 m length) from sites 2 and 3 (figure 3). Overall, 23 infaunal species were identified from both locations, 16 from the Pig Point site and 19 from the reference site. The dominant species of both sites were similar, with six of the top ten dominant species (for both biomass and abundance) being the same.

The species curve at the Pig Point site showed (1) an initial zone of low values compared to the reference site at 0-30 m (2) followed by a gradual increase to values higher than the reference site at a distance of 50-70 m, and finally (3) values comparable to the reference site for the remainder of the transect (figure 3A). No azoic zone was found as predicted by the PRM (figure 1). The number of species appeared to be depressed within the first 30 m at the Pig Point site.

The abundance curve at the Pig Point site showed (1) an initial zone of low values compared to the reference site at 0-30 m, and (2) values comparable to the reference site for the remainder of the transect (figure 3B). There was no distinctive peak of opportunistic species as predicted by the PRM (figure 1). Values for the first 30 m at the Pig Point site were approximately an order of magnitude lower than for the reference site.

The biomass curve at the Pig Point site showed (1) an initial zone (0-20 m) of highly variable values ranging from an order of magnitude lower to an order of magnitude higher than values at the reference site, and (2) values for the remainder of the transect that were generally an order of magnitude higher than the reference site (figure 3C). The predicted first peak in biomass of the PRM (figure 1) was not present at the Pig Point site, corresponding with the absence of a peak of opportunists. The high biomass values at the Pig Point site were almost entirely attributable to the bivalve species *Macoma balthica*, which accounted for

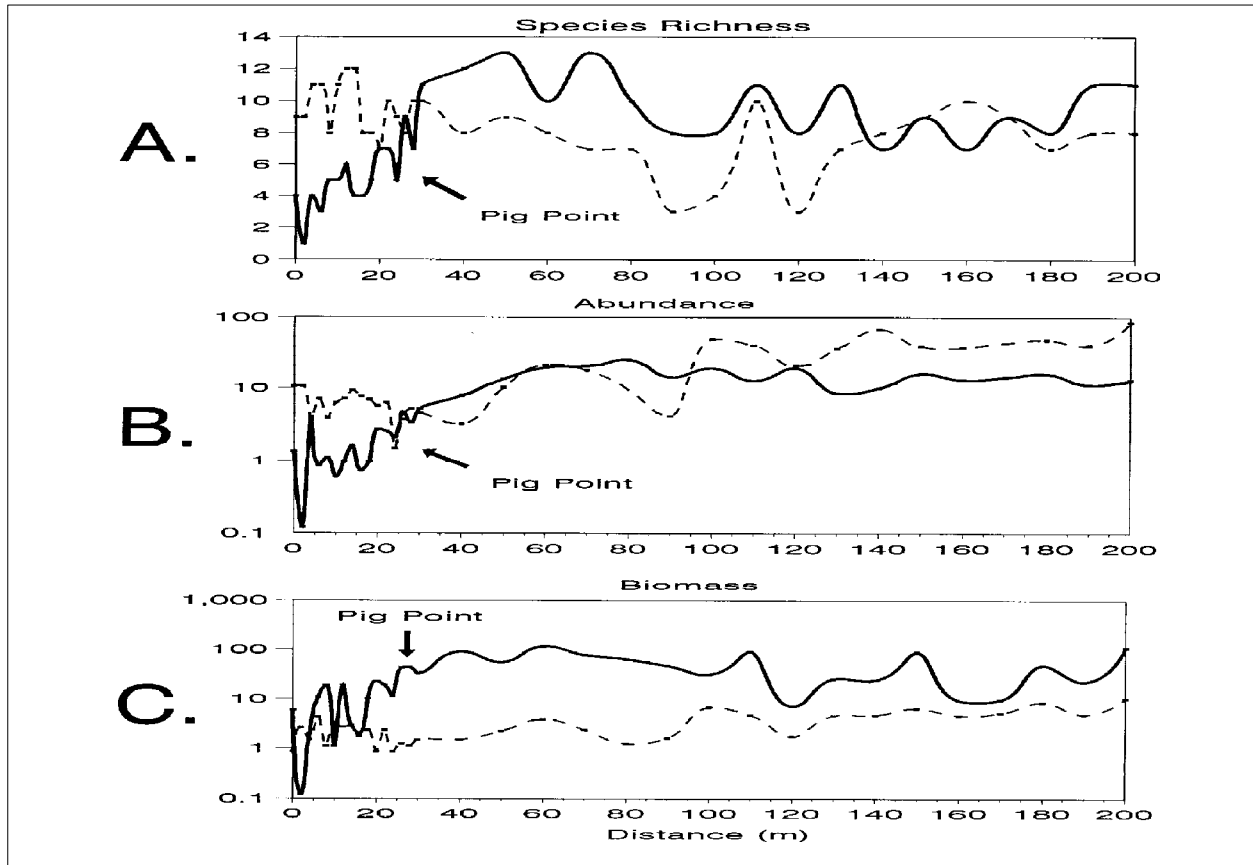


Figure 3. Mean species, abundance, and biomass (SAB) curves for the Pig Point study site 2 and the reference site 3. A: Species richness (species per sample). B: Abundance (individuals per m<sup>2</sup> x 100). C: Biomass (AFDW g per m<sup>2</sup>).

78.7% of the biomass at the Pig Point site. *m. balthica* accounted for 25.2% of the biomass at site 3, second only to *Leptocherius plumulosus*, which accounted for 33% of the biomass at site 3.

Sediment Analysis and TOC

The sediments at both sites were generally characterized as containing less than 40% silt-clay. TOC analysis did not show a pattern of elevated organic content near the source.

DISCUSSION

The SAB curves calculated from data collected in the vicinity of the seafood processing plant in Pocomoke Sound, Virginia generally conformed to the PRM predictions with (1) an initial zone of low values, compared to the reference site, in the 0-30 m region, followed by (2) an increase in values to levels comparable to the reference site for number of species and abundance of individuals and

higher than the reference site for biomass. For the species curve, the PRM predicts initially very low values, followed by a rapid increase to a peak value in the transition zone, and finally a slight reduction in reference communities. The species curve at the Pig Point site conformed to the PRM predictions. Contrary to the PRM predictions, all three SAB curves at the Pig Point site showed the same pattern as described above. The major difference between the results of this study and the PRM predictions were (1) the lack of an azoic zone and (2) the lack of a peak of opportunists which also produced the initial peak in biomass of the PRM. Similar differences with PRM were reported by Ferraro et al. (1991) and Maurer et al. (1993). Maurer et al. (1993) studying the San Pedro Shelf found no sharp decline in SAB curves to azoic conditions and no exclusion of rare species by opportunistic species. Similar results were also obtained by Ferraro et al. (1991), who did not find an enhancement of an opportunistic species with coincidental disruption of community structure.

The absence of an azoic zone was most likely an artifact of where the first sample on the transect at Pig Point was collected. The first sample on transect B was approximately 10 m from the discharge pipe in the drainage canal. It is quite possible that if an azoic zone existed, it was limited to a few meters from the discharge pipe. The lack of a peak of opportunists may be attributable to the highly stressful nature of an estuary and therefore to the natural dominance of species with opportunistic life histories. Defining opportunistic and equilibrium species in Chesapeake Bay has made possible the development of sensitive benthic monitoring procedures (Dauer 1993, Ranasinghe et al. 1993); however, an examination of the species defined as opportunistic versus equilibrium reveals that opportunistic species are generally euryhaline in distribution while equilibrium species are generally most common in higher salinities (high mesohaline to polyhaline). Future studies of stress in estuarine communities, particularly in the lower salinity reaches, should consider the need to develop habitat-specific definitions for opportunistic versus equilibrium groups. Species considered as opportunists in one salinity region might be considered equilibrium species in another salinity region.

The most striking difference between the Pig Point site and the reference site was the very high biomass values at Pig Point after the initial variable zone. This pattern was primarily attributable to the bivalve *m. balthica*. The PRM also predicts an increase in the mean size of organisms with increasing distance from an organic enrichment source. This trend was not supported by the present study as *m. balthica* dominated in biomass and abundance along the entire 200 m transect at Pig Point. The abundance of *m. balthica* at the Pig Point site was higher than at the reference site. The reference site had an average 1.39 *m. balthica* individuals per core while the Pig Point site had 13.87 *m. balthica* individuals per core. *m. balthica* collected in organically enriched areas often show an increase in abundance, biomass, shell growth, and secondary production (Pearson and Rosenberg 1978, Madsen and Jensen 1987, Josefson 1990). Previous usage of *m. balthica* as an equilibrium species (Dauer et al. 1992, Dauer 1993, Ranasinghe et al. 1993) should be reconsidered in light of the results of this study.

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ASSESSMENT OF CHESAPEAKE BAY BENTHIC MACROINVERTEBRATE RESOURCE  
CONDITION IN RELATION TO WATER AND SEDIMENT QUALITY STRESSORS

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*Versar*

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*Abstract:* Benthic macroinvertebrates are an integral part of the Chesapeake Bay monitoring program but their use in Bay-wide assessments of ecological status have been limited by the fact that benthic assemblages are strongly influenced by naturally varying habitat elements, such as salinity, sediment type, and depth. Recent development of benthic community restoration goals for Chesapeake Bay provides a framework for conducting assessments of benthic condition that are habitat-independent. This paper capitalizes on that advance by applying the restoration goals to the Virginia monitoring, Maryland monitoring, and EMAP data sets to estimate the extent of degraded benthic resources and identify regions of the Bay most in need of corrective action. Correlation and chi-square contingency analyses to examine associations between sites with poor benthic condition and measures of pollution exposure in the water column and sediment are also presented.

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COMPARISON OF BEHAVIORAL AND LARVAL TOXICITY MOLLUSC BIOMASSAYS FOR ATRAZINE IN  
FRESHWATER SEDIMENT

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**Abstract:** Atrazine was added to fresh mud sediment (13.7% organic content) as the commercial water-soluble 40W Aatrex® formulation. Atrazine was toxic to recently released larvae of the freshwater mollusc *Corbicula fluminea* (Asiatic clam), with an LD50 of 2,655 ug atrazine/g dry sediment. Mud sediment that was rinsed retained 80% of toxicity to larvae.

Aatrex® atrazine added to mud sediment caused significant burrowing delay in adult *Corbicula*, with an EC50 of 290-1440 ug/g. Aatrex® added to sand sediment (1.4% organic content) caused significant delay in burrowing speed delay starting at 72.4 ug atrazine/g.

This behavioral bioassay effect of Aatrex® added to sediment was inversely proportional to organic content. This could be attributable to a higher dissociation constant for Aatrex® atrazine than for atrazine sorbed to freshwater sediment organic matter. This would result in a higher pore-water atrazine concentration in the absence of organic material.

Sand sediment with added Aatrex® (163 ug/g) that was rinsed caused no burrowing delay in adult *Corbicula*. Aatrex® atrazine apparently was not sorbed to sand sediments. Behavioral bioassays with molluscs may be the most sensitive for detecting bioeffectiveness of pesticides in sediments.

## INTRODUCTION

Atrazine is a pesticide that is considered a Toxic of Concern in the Chesapeake Bay watershed owing to an average of 2,300,000 lb/yr applied primarily to corn (U.S. Environmental Protection Agency 1991). Atrazine is considered one of the most mobile and persistent pesticides in soil (Keim et al. 1989). Pure atrazine has limited water solubility but can be sorbed by suspended sediment. Sediment sorption slows the degradation rate so there is potential for accumulation in sediments. Aatrex® 40W is a commercial water-soluble emulsion formulation of atrazine designed to overcome handling and solubility problems, and is 40.8% atrazine.

Molluscs in general are highly exposed to sediment contaminants owing to lack of mobility and several species of freshwater molluscs in the United States are considered endangered. The larva of the freshwater Asiatic clam (*Corbicula fluminea*), an invasive species, was selected as the

toxicity bioassay mollusc. *Corbicula* larvae are nonglochidial and released as fully formed larvae from adults in the spawning season, and a sediment bioassay using the larvae has been developed (Phelps and Clark 1988). *Corbicula* is currently abundant in the Potomac estuary near Washington, DC (Phelps 1987) where it may have ecosystem importance (Phelps in press).

Pesticides in sediments can have significant sublethal effects on populations of molluscs other than toxicity. For example, Pridmore et al. (1991) found low levels of chlordane added to sediment (20 - 80 ng/cm<sup>2</sup>) reduced mollusc populations in marine sediments primarily because of drift movement of juveniles away from enriched sites. Bivalves were the most affected group and the three most common species showed declines of 31-56% (Pridmore et al. 1992). *Corbicula* juveniles are also known to disperse via floatation (Prezant and Chalermwat 1984). Belanger et al. (1993) reported

the most sensitive bioindicator for cationic surfactant in fresh water was its effect on *Corbicula* juvenile colonization. Behavioral reactions to low-level atrazine or AAtrex® in sediment might also result in population changes in freshwater molluscs. The effects of sediment-sorbed atrazine and 40W AAtrex® on the burrowing ability of adult *Corbicula* were also studied (Phelps 1994). The ability to burrow has significant adaptive importance in avoiding predation and in avoiding transport away from a favorable location.

#### METHODOLOGY

##### *Corbicula* Larva Sediment Bioassays

Sediment was collected from the freshwater estuarine Potomac River near Fort Washington and from the nearby Anacostia River. For larval toxicity exposure to sorbed atrazine, the sediment was sieved (110  $\mu$  Nytex) and stirred with a saturated solution of technical-grade atrazine (Dow Chemical) for 12 hours, centrifuged, and rinsed twice with distilled water. A trace amount of radioactive  $C^{14}$  atrazine (Sigma Chemical Co.) was added to the atrazine solution for determination of sorbed atrazine by subtraction. One ml of centrifuged sediment was placed in a 2.5 ml tissue culture well (Falcon Plastics) as described by Phelps and Clark (1988) and Phelps and Warner (1990). *Corbicula* larvae (20 or 30) that were recently released from adult *Corbicula* were placed in the well and layered with antibiotic-containing distilled water (neomycin, penicillin, streptomycin 1:100 (Sigma Chemical)). Each bioassay was conducted in triplicate. Tissue culture plates were kept at room temperature (18°C) for 96 hours, the well contents filtered (110  $\mu$  Nytex) and larvae scored as live (clear, moving) or dead (opaque, nonmotile, with gaping shell). As *Corbicula* shells are fragile and often not recovered if a larva died early in the bioassay period, the mortality average was calculated from living larvae recovered. The adjusted mortality percentage was calculated by subtracting average control mortality, which ranged from 9% to 17%.

For experiments with 40W AAtrex®, filtered centrifuged sediment was mixed with an equivalent (w:v 1:1) of 40W AAtrex® diluted to 0.01, 0.001 and 0.0001X with antibiotic-containing distilled water. The final concentration of atrazine in sediment was estimated from the AAtrex® formulation. One sediment sample with 0.01, AAtrex® 40W was rinsed twice at 1:10 (w:v) before bioassay.

##### *Corbicula* Burrowing Bioassays

The Asiatic clam burrowing bioassay was as described for estuarine and marine clams (Phelps et al. 1983, Phelps 1989). Adult *Corbicula* were collected from the Potomac River. Shell height influences burrowing speed (Phelps 1989), and ten clams of a single size class ( $12.9 \pm 1.96$  mm shell height) were used in each sediment mesocosm. The number completely burrowed with no projecting shell was noted at intervals (2, 5, 10, 15, 20, 40, 60 minutes). Probit analysis was used to determine the EI50 for 50% of clams to bury, with 95% fiducial limits.

Bioassay sediment was either sandy, from the Potomac River, or mud-type, from the Anacostia River. Water content was determined by drying and organic carbon content was estimated by loss on ignition (LOI, 550°C for one hour). Sand sediment had 16.9% water and 1.39% LOI; mud sediment had 74.6% water and 13.7% LOI. Sediment was stored in plastic containers at 8°C up to 3 weeks before use. Mesocosms were 200 g or 300 g sediment (WW) in 10 x 10 x 6 cm plastic boxes with 100 ml spring water (80  $\mu$ mos) overlay. Water quality is critical in the burrowing bioassay, and distilled or tap water inhibits burrowing behavior. To spike the 200 g or 300 g wet sediment for mesocosms, 0.01 AAtrex® was added to drained sediment and stirred for 10.0 minutes and then gently overlaid with spring water. The final concentration of atrazine in sediment was estimated. In one experiment, sand sediment spiked with 163  $\mu$ g atrazine/g (DW) (est.) was rinsed two times (1:1) with water and re-assayed.

#### RESULTS

##### *Corbicula* Larva Sediment Bioassays

Maximum sorption of the technical-grade atrazine to the < 110  $\mu$  fraction was 1,270  $\mu$ g/g dry sediment, and mortality of *Corbicula* larvae was only 20%. Addition of AAtrex® 40W to sediment gave 11,600  $\mu$ g atrazine/g sediment (DW), which had 80% mortality of larvae and established an LC50 of 2,665  $\mu$ g atrazine/g sediment (Phelps 1994). Atrazine LC50 of 6,300 to 9,900  $\mu$ g/L has been reported for adults of other freshwater molluscs (Huber 1993). Toxicity of the rinsed sediment was 81% of unrinsed sediment (Phelps 1994).

## Corbicula Burrowing Bioassays

Average ET50 was approximately 5 minutes for sand and 8 minutes for mud sediment. Addition of 0.01 AAtrex® to either sand or mud sediment caused a logarithmic increase in burrowing speed ET50. The lowest estimated concentration of atrazine in sand sediment causing statistically significant burrowing delay was 72.4 ug atrazine/g sediment (DW).

Rinsing the sand sediment that had 163 ug AAtrex® atrazine/g reduced the ET50 from 4.3 hrs to 0.106 hrs (statistically equivalent to control ET50). This suggested the bioactive form of AAtrex® was not sorbed to sand sediment.

AAtrex® added to mud sediment did not cause significant burrowing inhibition in *Corbicula* until a 4-20 greater concentration than in sand sediment. The mud sediment had 9.9 times the organic content of sand sediment. The lower reactivity of AAtrex® atrazine in mud sediment suggested that the organic material sorbed free atrazine.

This organic material had a lower dissociation constant than AAtrex® and resulted in a lower bioeffective (pore-water) concentration than the same amount of AAtrex® in sand sediment. Uptake in fine-grained mud-type sediment reduced the effectiveness of AAtrex® atrazine by 80%, while the rinsing experiment suggested that sand sediment did not sorb or reduce the effectiveness of AAtrex® atrazine.

A recent summary of atrazine in aquatic systems (Huber 1993) found atrazine could have a half-life of over 300 days in ecosystems, particularly freshwater sediment. The present study found the behavioral bioassay with adult *Corbicula* was at least ten times more sensitive to concentrations of atrazine in sediment than larval mortality (72.4 and 300-1,400 versus 2,655 ug/g). However, even this burrowing bioassay is not sufficiently sensitive to detect ecosystem effects of atrazine at reported environmental concentrations in Chesapeake Bay of a maximum of 0.8 ug/g in freshwater sediment (Beane 1977).

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DISSOLVED ORGANIC CARBON (DOC) CYCLING IN ESTUARINE SEDIMENTS: DOES THE  
MACROBENTHOS HAVE ANY EFFECT?

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*Abstract:* Studies of dissolved organic carbon (DOC) cycling in Chesapeake Bay sediments suggest that the presence of benthic macrofauna has a significant, though perhaps counterintuitive, effect on these processes. At a sulfidic site in the mid-Bay (site M), only seasonally (spring) colonized by small polychaetes and bivalves, DOC fluxes from the sediments corresponded to ~4 to 13% of the integrated benthic carbon remineralization rate (YOCR) and were likely controlled by molecular diffusion. At a much more heavily, and year-round, bioturbated site in the southern Bay (site S), the activity of macrobenthos did not appear to enhance DOC fluxes from these sediments, in that measured benthic DOC fluxes were lower than those at site M. The ratio of DOC fluxes to YOCR at site S was also slightly smaller than that observed at site M. Consistent with these observations, pore-water DOC concentrations at site S were generally lower than those at site M, and down-core concentration gradients were also smaller. The causes of these differences are not well understood and may be related to several factors. Macrofaunal activity may change sediment redox conditions, causing differences in the relative rates of DOC production and consumption by either microbial or abiotic reactions. Direct utilization of pore-water DOC by benthic macrofauna may also cause these differences. These possibilities also have important implications for the effects of benthic macrofauna on the cycling of sedimentary pollutants. These include organic pollutants, redox-sensitive metal pollutants, and other metals pollutants for which organic complexation may play an important role in affecting their estuarine cycling.