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ENVIRONMENTAL PROTECTION AGENCY STAFF PAPERS PRESENTED AT THE CHESAPEAKE BAY RESEARCH CONFERENCE MARCH 1988

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and
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by
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Controlled Energy Dissipation from River Inflow as a Factor in Managing Estuarine Water Quality

by

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INTRODUCTION

River inflows into estuarine systems may carry with them large amounts of energy depending on their size and volume, and the constant tidal action provides an additional input of energy carried by the rhythmic movement of water into and within an estuary. The potential and kinetic energy of rivers is often tapped for hydroelectric power during their passages toward the sea, and where tidal range and velocity are sufficient, the energy of tidal action has been used to generate electrical power.

As the flow from a river and tidal flow from the sea meet in an estuary they usually are moving in different directions, with the kinetic energy of their passage also moving in opposite directions. Upon the meeting of two such inflows the kinetic energy is rapidly dissipated, sometimes with dramatic visible effects. For example, with an incoming tide there may be a zone of significant wave action, i.e., a "tide rip", over a limited area during a time of very light winds. Conversely, on an ebb tide there may be a zone of very smooth water with a clearly visible line of demarcation between two water bodies as the kinetic energy from river and tide move smoothly along together. In either case the kinetic energy from both sources is dissipated in an uncontrolled manner with effects generally on the mixing characteristics of the system, particularly on stratification and resuspension of sediments.

If the dissipation of this energy in the Chesapeake Bay could be controlled in some manner, it might very well be possible to control the extent of stratification in parts of the Bay so as to enhance the aquatic environment for living resources management and public health. This investigation is a preliminary investigation of the technical feasibility of doing this in the Chesapeake Bay. It does not address the broad issues of administrative feasibility, social desirability, and legal responsibility, nor the even broader scientific issue of whether or not we know enough about the physical, chemical, and biological processes of the Bay to be able to do this in a responsible manner.

The specific questions to be addressed in this study are:
1. Has a significant change in estuarine stratification and water quality resulting from a river inflow change ever been documented?
2. What physical conditions would be necessary to do this in the Chesapeake Bay and do they presently exist?
3. Can the propagation of kinetic energy and its dissipation in the Bay be demonstrated?
4. Is it feasible to develop a mathematical description of the kinetic energy flow in the Bay sufficiently quantitative to serve as the basis for regulating river discharges into the Bay?
5. Since such a management scheme would involve real time control of river flows into the Bay, what information on conditions in the Bay would be needed as a basis for using such an approach?
6. Is this approach worth pursuing further, and, if so, what must be done?

The following sections of this paper address each of these questions in order.

HAS A SIGNIFICANT CHANGE IN ESTUARINE STRATIFICATION AND WATER QUALITY RESULTING FROM A RIVER INFLOW CHANGE EVER BEEN DOCUMENTED?

In the early 1940's at Charleston,S.C., a hydroelectric dam was built near the head of tidewater on the Cooper River, the major river input to Charleston Harbor. To feed this dam, the flow of the Santee River was diverted to the power pool behind the structure. The net result was an increase in mean river flow to Charleston Harbor from 300 cfs to 15,000 cfs. The first noticeable effect of this change was an increase in the cost of dredging from $400,000 per year to $4,000,000 per year over a period of several years. even at this cost it was impossible to keep any slips perpendicular to the main channel clear.

A subsequent investigation(Wastler and Walter 1969) showed that the increased river flow had changed the main part of the Harbor from an unstratified to a salt wedge estuarine circulation system. This study concluded that reduction of the mean flow to less than 8000 cfs would break the stratification and reduce the input of sediment to the Harbor. The flow of the Santee was in

part rediverted to its original waterway to reduce the mean flow to these levels, and the stratification was indeed broken.

The Harbor at Charleston is once again a natural deepwater system with a significant reduction in annual dredging costs. Sedimentation was reduced to nearly its original levels and other aspects of water quality were also improved, in large part due to considerable improvements in waste treatment, which were also recommended because of the tremendous decrease in flushing time that would occur with the breaking of the salt wedge.

This example involved only an overall reduction of mean river input into the system, and no effort was made to control the river input in synchronization with tidal flow. However, this example does demonstrate that dramatic changes can be caused by changes in stratification resulting from changes in river flow and that these changes could be quantified and predicted, at least in the example given.

WHAT PHYSICAL CONDITIONS WOULD BE NECESSARY TO DO THIS IN THE CHESAPEAKE BAY AND DO THEY PRESENTLY EXIST?

In the Charleston Harbor situation there was a single major river inflow into the Bay and it clearly was a primary factor in controlling the extent of stratification. There was also a dam at the head of tidewater which had the capability of regulating the river discharge into the estuary. While it might be possible to visualize other physical conditions with which it might be possible to regulate river flow so as to alter stratification, is clear that these conditions are suffucient.

In the Chesapeake Bay system, the major single river inflow is the Susquehanna, which provides about 87 percent of the riverine input to the Bay above the confluence of the Potomac and 50 percent of the entire riverine input to the entire Bay. Conowingo Dam is about 9 miles above the head of tidewater and receives the flow from over 90 percent of the Susquehanna drainage basin. The pool behind Conowingo Dam is not sufficient for long-term storage but the dams operated in the Basin by the Corps of Engineers do have a total amount of storage sufficient to supply a minimum flow of about 30,000 cfs over a several month period with proper routing. Thus, the physical structure necessary to provide river flow regulation does exist at the head of tidewater.

The question of whether or not the Susquehanna does have a sufficient impact on stratification of the main stem of the Bay to allow effective regulation, particularly at low flows, must also be addressed. To do this, the results of the main stem monitoring program were examined in regard to the extent of stratification of the main stem throughout the year and how well this correlated with the annual cycle of river flow. Table I exhibits the ratios of salinity in the Main Stem of the Bay for the full period of reported data for the present monitoring program, i.e., June 1984 to July 1987, at approximate mile points closest to the routine sampling stations. The geographical area covered is from Havre de

Grace to the mouth of the Potomac, the reach of the Bay in which the Susquehanna accounts for nearly 90 percent of the river flow entering the Bay. Data are presented on a monthly basis and each value represents only one, or at the most an average of two samples.

There are several striking features of the data presented in Table I as far as the impact of the Susquehanna flow on stratification is concerned. First, the data show absolutely no evidence of any type of seaonal overturn and redevelopment. There are changes in the degree of stratification during a year, but these do not appear to be related to any kind of seasonal regime, nor to any obvious environmental scenario. Stratification in 1984 appears to be stronger than in either 1985 or 1986; however, there is less than half a year of record in 1984, so it is difficult to make an equitable comparison. Second, the change of the degree of stratification down the Bay from Havre de Grace does not appear to to be very great for a specific year or season, which suggests that whatever forces are causing the degree of stratification observed persist far down the Bay. Third, the degree of stratfication is extreme throughout the Bay and for much of the year. At salinity ratios of around 0.6, Charleston Harbor behaved as a salt wedge estuary. However, the size and structure of the Chesapeake are different from Charleston, and it is not known whether the use of salinity ratio as a surrogate for estuarine behavior is valid. Fourth, the asterisks (*) by certain values in the table indicate times and places where the Bay exhibited more than one pycnocline. These are quite common in the data set, and suggest the existence of a consistent phenomenon responsible for the condition.

Table II presents some data which may offer an insight into the reasons for the existence of multiple pycnoclines in the Bay as well as indicate a source of some of the organic material responsible for the depleted oxygen levels in the bottom layer of the water column. This Table compares water column densities in the Potomac near its mouth with those in the Bay itself above, at,and below the confluence of the Potomac with the Bay. For ease in reading the density values have been adjusted so that they appear in the table as small numbers; however, what is significant are their relative values, and these are unchanged. The data presented are for the entire year of 1985, and what is important is to note the relative densities of the Potomac water and those of the Bay at and close to the confluence.

Remember that water(e.g., Potomac water) tends to ride over other water it meets of higher density(e.g.,usually Bay bottom water) and slide under water it meets of lesser density(e.g., usually Bay surface water). The data show that in nearly all cases the surface water in the Potomac would tend to ride out over or mix with the surface waters of the Bay, although on two occasions, 3/19/85 and 5/6/85 the relative densities were such that the Potomac water would tend to slide under the surface water of the Bay. If these conditions persisted for any length of time, a significant amount of Potomac surface water could be introduced

into the bottom waters of the Bay where it would decay without benefit of reaeration, thus causing possibly severe oxygen depletion. For most of the year it appears that the surface layers of the Potomac tend to end up in the surface waters of the Bay but that a midlayer representing some kind of blending of Potomac bottom water and Bay bottom water may very well form in the Main Stem of the Bay. The small but significant differences between the densities of the bottom and midlayers in the Bay suggest the existence of such a mechanism.

This look at the salinity structure as shown by the monitoring data suggests that there is no obvious, direct relationship between the Susquehanna River flow and stratification in the Bay. A more detailed look at the salinities of the bottom layer indicated that these did tend to vary consistently with river discharge, but a quantitative relationship could not be established with the data available. Nevertheless, the wide range of values of salinity ratios in no coherent pattern over a year suggests that the Bay responds quite rapidly to changes in river discharge and that samples taken a minimum of two weeks apart are not adequate to establish the nature of the response. More detailed measurements are needed to resolve this problem.

CAN THE PROPAGATION OF KINETIC ENERGY AND ITS DISSIPATION IN THE BAY BE DEMONSTRATED?

The quantity of kinetic energy in a flowing body of water is a function of its mass and the square of its velocity. In this case there are two bodies of water involved, that associated with the river discharge into the Bay and that associated with the tidal flow. Since water behaves as an incompressible fluid, the interaction of the two bodies should be measurable as a perturbation of the observed tide height by the river discharge. The methods described by Wastler(Wastler 1969) and used by Wastler and Walter in the Charleston Harbor analysis allow the estimation of the energy present at the dominant periods exhibited in each record at each sampling point. These methods involve the calculation of cross-spectra from river discharge and tide height records and the interpretation of the results in terms of the physical parameters involved. The effect of the presence of some potential energy due to the slope of the estuary from its head toward the sea is eliminated by using only deviations from the mean tide height at each tide gage; the potential energy of the water would, however, be small in relation to the kinetic energy in any event.

For the present case records of hourly discharges from Conowingo Dam were available for the first nine months of 1983. These were run against hourly tide gage records for the same period at Havre de Grace, Matapeake, and Solomons, affording a coverage from the head of tidewater nearly to the mouth of the Potomac. A range of river discharges based on daily means from 5000 cfs to 163,000 cfs were covered by these records, thus incorporating an excellent range of flows from the Susguehanna. The river discharge records had an extremely strong diurnal component typical of hydroelectric

dam operation. The analysis showed that the propagation of energy from the river discharge was extremely rapid and consistent at all river disacharges. This amounted to 44 hours to the Solomons gage, or roughly two days from Havre de Grace to the mouth of the Potomac. This is, of course, the speed of energy propagation, not the time of water travel. It should be noted that the rapidity of energy propagation from the river flow could have some bearing on the similarity of the degree of stratification down the Bay, as shown in Table I. The diurnal component of river flow was still present at Solomons with about 25 percent of the original energy still present. Long-period components of the river discharge (i.e., those of periods greater than 7 days) were not large in the river record, and these were not distinguishable in the records at Solomons. This suggests that the long-period energy of the river discharge is dissipated in the upper part of the Bay.

These results indicate that energy from the Susquehanna flow makes its way at least halfway down the Bay in a quantifiable form, that it tends to be distributed throughout the upper part of the Bay in its presently uncontrolled regime, and that its time of propagation is quite rapid. Thus it appears that there is enough energy available to use if there is a means of controlling it.

IS IT FEASIBLE TO DEVELOP A MATHEMATICAL DESCRIPTION OF THE KINETIC ENERGY FLOW INTO THE BAY SUFFICIENTLY QUANTITATIVE TO SERVE AS THE BASIS FOR REGULATING RIVER DISCHARGES INTO THE BAY?

Kinetic energy balance equations for estuarine flow and for river discharges are standard textbook exercises, but apparently no one has ever put the two togather. As part of this investigation a synthesis of the two was attempted, not to develop an exact solution, but to establish the nature of the equation and the parameters necessary for its solution. A harmonically varying river discharge was introduced to determine its effect.

The results indicated that a mathematical description of the process is feasible, but there are some problems. First, there appear to be a large number of acceptable mathematical solutions depending on a number of unknown quantities which can only be measured in the field. That is, it would be necessary to experimentally vary the discharge from Conowingo in a preset pattern and measure what happens. Second, with opposing harmonic flow patterns, there is a real possibility of a feedback process developing with the formation of a disastrously large standing wave in the Bay. Again, this may be a mathematical artifact, and field investigation may demonstrate that such is the case.

In short, yes, it is feasible to develop a quantitative mathematical description of what would be required, but it must be based on accurate field studies in its development stage, and it must be carefully and thoroughly evaluated in the field before it is used in an operational mode.

SINCE SUCH A MANAGEMENT SCHEME WOULD INVOLVE REAL TIME CONTROL OF RIVER FLOWS INTO THE BAY, WHAT INFORMATION ON CONDITIONS IN THE BAY WOULD BE NEEDED AS A BASIS FOR USING SUCH AN APPROACH?

Before such a management scheme could be implemented, it would be necessary to have a very clear understanding of what changes in river discharge would do to the salinity structure of the Bay. It would also be necessary to understand what salinity structure was needed at specific points in the Bay to provide the suitable degree of stratification to protect and enhance the habitat for living resources and to maintain a predetermined level of water quality. With these considerations in mind, it can then be stated that high-frequency real time measurements of salinity are the only water quality information really necessary. However, the only cost effective technique to obtain this type of information presently available is automatic remote sensing buoys. The major cost of such devices is in the buoy itself and in the data transmission system, so it would be reasonable to collect other useful information such as Temperature, Dissolved Oxygen, Turbidity, pH, and other parameters of interest to management authorities and responsible investigators.

IS THIS APPROACH WORTH PURSUING FURTHER, AND,IF SO, WHAT MUST BE DONE?

This paper has emphasized the unknowns and the problems involved in developing and implementing a management approach based on flow regulation for positive control of the aquatic environment. These problems and unknowns are real in the context of this management approach certainly; but they are no less real in the context of any other management strategy.

A major problem identified has been that of establishing a firm relationship between river flow and stratification. Certainly it is necessary to know this to differentiate between "wet years" and "dry years" in any quantitative sense for any type of model. Establishing such a relationship is a first and major step in any further investigations, whether of this management approach or any other approach toward water quality control.

For example, it would be impossible to develop a complex water quality model of the Bay without accounting for the daily pattern of flows out of Conowingo, the effect of such variations in flow on water quality, the development of multiple stratification layers in the Bay, and the occasional underflow of the Potomac into these layers. The development of any such model would go far toward evaluating the feasibility of controlled river flow regulation as a management tool.

This approach should be examined further, not as a unique and separate entity, but as part of the total array of management techniques that might be feasible.

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Table I

Salinity Ratios in the Main Stem of the Chesapeake Bay
(June 1984 - July 1987)

Date	Miles from Havre de Grace								
	20	30	40	50	60	70	80	90	100
1984									
June	-	-	0.42	-	0.63	-	0.65	-	0.78
July	-	0.22	0.36	0.36	0.46	0.45	0.49*	-	0.57
Aug	-	0.22	0.33	0.38	0.63	0.62	0.41*	0.38	-
Sept	0.94	0.97	0.68*	0.70*	0.81	0.74	0.71	0.71	0.84*
Oct	-	0.94	0.82*	0.82*	0.72	0.68	0.78*	0.76	0.69
Nov	-	-	0.66	0.74*	-	-	0.74*	0.78*	0.83*
Dec	0.41	-	-	-	0.75*	0.79	-	-	-
1985									
Jan	-	-	0.83	-	-	-	0.85*	0.88	0.85
Feb	-	-	-	-	-	0.84	-	-	-
Mar	-	0.40	0.62	0.67	0.66	0.82	0.76	0.82*	0.85
Apr	-	0.20	0.52*	0.54	0.62*	0.61*	0.68*	0.74*	0.68
May	0.79	0.53	0.71*	0.68	0.67*	0.78*	0.65	0.77*	0.82
June	0.52	0.66	0.70*	0.67	0.76	0.72	0.68	0.76*	0.79
July	0.78	0.59	0.61*	0.65	0.75	0.68*	0.75*	0.70*	0.78*
Aug	0.76	-	0.67*	0.72*	0.76	0.67*	0.76	0.71*	0.76*
Sept	0.85*	0.82*	0.83*	0.82*	0.74	0.78	0.77	0.76	0.75
Oct	0.60	0.86	-	0.90	0.80	0.88	0.81	0.92	0.99
Nov	0.79	-	-	-	0.91	-	0.86	0.86	-
Dec	-	0.69	0.63	0.80	-	0.74*	-	-	0.84
1986									
Jan	0.42	-	-	-	-	-	0.84	-	-
Feb	-	0.62	0.47	0.58*	-	0.51*	-	0.68	0.73
Mar	-	0.26	0.24	0.45*	0.68	0.52	0.72	0.70	0.69*
Apr	-	0.29	0.25*	0.43*	0.41	0.57	0.55*	0.57*	0.63
May	-	0.42*	0.51	0.52*	0.67	0.56*	0.68	0.60	0.62*
June	-	0.52	0.56*	0.67*	0.61	0.76*	0.67	0.76	0.89
July	0.71	0.68	0.68	0.67	0.60	0.69	0.64	0.69	0.67*
Aug	0.92	0.69	0.66*	0.68	0.68*	0.76*	0.70	0.80	0.78
Sept	0.75	-	0.88*	0.85	0.71	0.79*	0.74	0.88	0.86
Oct	0.67*	0.70	0.67*	0.74*	0.72	0.74	0.75*	0.74	0.74
Nov	-	-	-	-	0.86	-	0.85	-	-
Dec	0.09	0.47	0.58	0.63	-	0.71	-	0.80	0.81
1987									
Jan	-	-	-	-	0.76	-	0.82	-	-
Feb	0.77	0.74	0.81	0.79	-	0.79	-	0.79	0.64
Mar	0.08	0.70	-	0.74	0.78	0.68	0.77	0.67	0.72
Apr	0.45	-	0.24	0.57	0.64*	0.51*	0.80	0.70	0.65
May	0.08	0.69	0.66	0.72	0.62	0.70	0.66*	0.71	0.73
June	0.45	0.78*	0.60	0.73	0.69*	0.79	0.69	0.75	0.78
July	-	0.62	0.72	0.63	0.71	0.64	0.78	0.68	-

Data Source: Chesapeake Bay Program monitoring data

Table II

Comparison of Water Densities at the Mouth of the Potomac River with Those in the Main Stem of the Chesapeake Bay Above and Below the Potomac Confluence*

Date		Bay above Potomac (CB5.2)	Potomac Mouth (LE2.3)	Bay at Confluence (CB5.3)	Bay below Potomac (CB5.4)
1/14/85	U	1.278	1.254	1.320	1.350
	M	-	-	-	1.429
	L	1.456	1.444	1.552	1.490
2/11/85	U	1.391	1.265	1.457	1.320
	M	-	-	-	-
	L	1.515	1.451	1.585	1.467
3/4/85	U	1.096	0.977	1.006	1.145
	M	-	1.086	-	-
	L	1.455	1.204	1.456	1.308
3/19/85	U	1.171	1.216	1.241	x
	M	1.288	-	-	x
	L	1.435	1.411	1.451	x
4/8/85	U	1.155	1.109	1.198	x
	M	1.446	1.260	1.626	x
	L	1.708	1.488	1.856	x
4/22/85	U	0.969	0.956	0.956	1.059
	M	1.119	1.090	1.227	1.305
	L	1.392	1.286	1.569	1.513
5/6/85	U	0.978	1.044	1.053	1.092
	M	1.189	1.321	1.389	1.449
	L	1.563	1.511	1.513	1.691
5/20/85	U	1.025	0.898	1.156	1.244
	M	1.150	1.068	-	1.316
	L	1.394	1.364	1.450	1.502
6/3/85	U	0.979	0.967	1.040	0.973
	M	-	-	1.365	1.333
	L	1.504	1.273	1.616	1.561
6/17/85	U	1.007	0.882	1.054	1.082
	M	1.175	1.042	-	1.248
	L	1.434	1.260	1.402	1.434

Table II (Continued)

Date		Bay above Potomac (CB5.2)	Potomac Mouth (LE2.3)	Bay at Confluence (CB5.3)	Bay below Potomac (CB5.4)
7/8/85	U	0.991	0.991	1.034	1.054
	M	1.356	-	-	-
	L	1.611	1.284	1.727	1.586
7/22/85	U	0.976	0.939	1.036	1.220
	M	1.418	-	1.102	-
	L	1.609	1.301	1.465	1.556
8/6/85	U	1.036	1.034	1.112	1.215
	M	-	-	-	-
	L	1.369	1.159	1.419	1.567
8/19/85	U	1.034	1.012	1.099	1.105
	M	1.399	-	1.256	-
	L	1.605	1.193	1.559	1.284
9/9/85	U	1.003	0.879	0.826	1.003
	M	1.240	-	1.076	-
	L	1.429	1.487	1.389	1.432
9/23/85	U	1.220	1.211	1.267	x
	M	1.429	-	-	x
	L	1.669	1.300	1.767	x
10/7/85	U	1.347	1.287	1.320	1.386
	M	-	-	-	-
	L	1.660	1.340	1.649	1.639
11/12/85	U	1.422	0.852	1.044	x
	M	-	-	1.247	x
	L	1.733	1.464	1.627	x

* Values in table are (Density x 1000) - 1000, which gives more readable numbers.

U - upper layer
M - middle layer(- means there is no middle layer)
L - bottom layer
x - no data

Role of the Organic and Inorganic Carbon Systems in the Dissolved Oxygen Regime of the Chesapeake Bay

by

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INTRODUCTION

Examination of the three years of water monitoring data on the main stem of the Chesapeake Bay(collected from June 1984 to July 1987 as part of the Chesapeake Bay Program), suggested that there is much closer coupling of the organic and inorganic carbon systems in the main stem of the Bay than had been previously suspected.

A quantifiably close coupling of the two organic systems in the Bay would allow prediction of the behavior of one system from the behavior of the other, and, since dissolved oxygen is the connecting link between the two systems, it should be possible to predict the behavior of the dissolved oxygen regime from knowledge of either the organic or inorganic carbon systems. Since the composition and dynamics of the inorganic carbon system can be determined from measurements of temperature, salinity, and pH alone, this would make it quite simple to predict periods of low dissolved oxygen from relatively straightforward measurements.

THEORETICAL BACKGROUND

The total carbon budget of any water body is composed of the sum of the components of the organic carbon system plus those of the inorganic carbon system. In freshwater the major concern is the organic carbon system, since the inorganic carbon system is driven

primarily by the solubility of gaseous carbon dioxide in fresh water. In seawater, however, the inorganic carbon system is quite complex ,as can be attested by all students of marine chemistry. The best and most detailed discussion of this subject is in Harvey's classic text "The Chemistry and Fertility of Seawater" (Harvey 1963), and only aspects of this system immediately relevant to anoxia in the Chesapeake Bay will be mentioned here.

The components of the inorganic carbon system are hydrogen ions, carbonate ions, bicarbonate ions, free carbon dioxide, and undissociated carbonic acid. The exact amounts of each component present in a specific situation are determined by the temperature, salinity, pH, and pressure. As pH values increase, the amount of free carbon dioxide decreases and the equilibrium is shifted toward the carbonate ion. Also, as algae consume free carbon dioxide, the pH tends to increase, thereby causing the amount of free carbon dioxide that can exist in the equilibrium to decrease. The kinetics of the equilibrium are not extremely rapid as far as the regeneration of free carbon dioxide from the carbonate is concerned, therefore it is possible that conditions could exist in the ocean in which algal growth is limited by the availability of free carbon dioxide even in the presence of a large excess of total inorganic carbon. It has been observed in some estuarine situations that algal growth has been inhibited by a lack of carbon dioxide at pH 9 (Kinne 1972).

In fresh water, on the other hand, a very large amount of free carbon dioxide can be dissolved in the water since the concentration of dissolved carbonate ion is quite small and does not significantly affect the chemical equilibrium. Thus, the conditions in which the availability of inorganic carbon may become a limiting factor in algal growth would not be expected in fresh water systems.

Rates of organic decay and oxygen consumption in fresh water systems have been studied in considerable detail in various water quality control and public health applications; the classic formulation by Streeter and Phelps of the oxygen sag in streams is accepted as the standard theoretical description of the natural decay processes involved(Fair and Geyer 1963). In salt water the rate of decay appears to vary with the concentration of the seawater. In concentrations of up to about 25 percent seawater, the rate constant for organic carbon decomposition is greater than in fresh water, but in higher concentrations it is depressed.

The total amount of carbon present in a water body at a specific time and place consists of the sum of those amounts present in the organic carbon system plus those present in the inorganic carbon system. Between times and places there may be exchanges between the two systems (which would not change the total amount of carbon present) but there may also be external sources or sinks of carbon in either organic or inorganic form. In examining the monitoring data on the Bay, consideration must be given to any such external sources or sinks of carbon, particularly to the sediment oxygen demand, which has been regarded as a significant factor in

development of anoxia in the Bay. Resuspension of organic matter from sediments into the water column is a potentially important source of organic carbon in the water column where water column properties such as dissolved oxygen could be affected.

The primary connecting link between the organic and inorganic carbon systems is oxygen, particularly the free oxygen dissolved in the water column. Oxygen is used up in the decay process of the dissolved organic matter with the formation of carbon dioxide; carbon dioxide is used by algae to produce organic matter with the release of oxygen. The algae themselves consume some oxygen in respiration and when algal cells die their decaying tissue uses up oxygen with the formation of carbon dioxide. Thus oxygen is an integral part of both systems and is directly involved in the transference of carbon from one system to the other.

The basis for this investigation is the application of the standard textbook concepts and relations described above to the specific problem of quantifying the coupling of the inorganic and organic carbon systems in the bottom waters of the Chesapeake Bay and gaining some insight into the quantification and prediction of the hypoxia of those waters.

STUDY METHOD

The method used in this investigation was a simple materials balance at pairs of monitoring stations occupied in the Chesapeake Bay Program monitoring effort between June 1984 and July 1987. The overall approach was to use the classical theory of the oxygen depletion in a natural body of water as enunciated in the Streeter-Phelps analysis in concert with the accepted inorganic carbon system behavior as summarized by Harvey. The parameters required for the analysis were Temperature, pH, Salinity, Dissolved Oxygen, Total Organic Carbon(TOC), Dissolved Organic Carbon(DOC), depth of the pycnocline, and the thickness of both surface and bottom layers. Particulate Organic Carbon(POC) can be calculated from TOC and DOC, and total inorganic carbon and free Carbon Dioxide can be calculated from pH, Temperature, and Salinity using the equilibrium constants and tables presented in Harvey. Thus it was possible to use only pairs of stations for which all of these parameters were available.

The major geographical area of interest is the reach of the Bay between the Bay Bridge at Annapolis and the mouth of the Potomac River. This includes the part of the Bay having the greatest depths and the most severe oxygen depletion problems. The analysis was therefore concentrated in this reach, with the analysis limited to the period between May and November 1986, which is the only period for which sufficient data were available. Individual samples were used to calculate rate processes between stations, since the two week or greater sampling period made the aggregation of samples for this purpose highly suspect.

The monitoring data show that the Bay between the Bay Bridge and the Potomac is stratified throughout the year, with the ratio of

surface to bottom salinity being generally about 0.6 to 0.8, indicative of a strongly stratified system. The monitoring data show no evidence of a seasonal overturn in the main stem of the Bay; in fact, the data show the existence of more than two layers within this reach for part of the year. The materials balance for this reach therefore reflects the existence of at least a two-layered system, with the flow in the bottom layer up the Bay, and that in the surface layer down the Bay.

Some mixing between layers is expected even in a strongly stratified system. Thus, in addition to being transferred chemically or biochemically from one carbon system to the other, there is some mechanical transfer between layers for each constituent. An overall materials balance for the Salinity, which is a conservative parameter, provides ratios of surface layer to bottom layer average velocity between the stations, as well as a ratio of the vertical transport velocity to either the surface layer or bottom layer velocity. An average effective cross-section for the two stations is assumed; the cross-sectional area above the pycnocline is assumed to be rectangular, and that below the pycnocline is assumed to be triangular with the altitude equal to the observed thickness of the bottom layer.

The first step was to compute the overall carbon balance for both layers and for each layer separately. This provided an estimate of the external sources and sinks of carbon, both organic and inorganic. Balances on each carbon system for each layer then gave estimates of the external gain or loss to each system. Next, for the bottom layer material balances were made for DOC, TOC, dissolved oxygen, and free and total carbon dioxide, assuming decay processes of the first order, following the Streeter-Phelps formulation for the decay of organic matter in the absence of reaeration. Calculations were also made in some cases using a second order rate process assumption. In the bottom layer calculations it was assumed that any change in POC between stations was the result of scouring or deposition of sediment containing adsorbed organic matter. An increase in DOC between stations in the absence of a known external point source, is assumed to be from organic matter dissolved off POC or interstitial water elutriated from the sediment bed load.

RESULTS

A first matter of concern is the extent to which the sediment load (or "bed load" in stream pollution terminology) may affect the dissolved oxygen regime in the Chesapeake Bay, particularly that part of the Bay which is subject to anoxic conditions in the bottom layer, and which is the area selected for examination in the present study. The technique of materials balancing, which considers all sources of carbon to the water column, affords an opportunity to look at external carbon sources, and in the case of the bottom layer of the water column, particularly what is put into or taken out of that layer. Table I summarizes the results of this analysis of the monitoring data. The data show a mixed

pattern of scouring and settling of Particulate Organic Carbon and also a similarly mixed pattern of addition or loss of Dissolved Organic Carbon to the water column. A significant point to note in this table is that the magnitude of addition or loss to the water column from the sediment load is of the same order of magnitude as the smallest reporting number of the data. This indicates that the quantitative extent of the impact of the sediment load is so small as to not be detectable in the monitoring data. Therefore, in analyzing the monitoring data, separate consideration of the sediment load as a factor in hypoxia in the bottom layer is not appropriate, since its impact is subsumed in the existing water column data.

Table IIa. compares the results of calculating the values of Dissolved Organic Carbon using the inorganic carbon balance with the observed values reported in the monitoring program. Table IIb. shows differences between the observed values of Dissolved Oxygen depletion between two monitoring stations and the calculated increase in Carbon Dioxide between the same two stations. The pattern of differences is again mixed, but again the differences are within the order of magnitude of the measurements. The bottom line conclusion is that there does not appear to be a basis for deciding that there is a significant difference between the two sets of data. An analysis of variance of the data in Tables IIa. and IIb. showed that in each case the data could be considered to be of the same set at the 99 percent probability level.

Table III shows the results of calculating first order deoxygenation rate constants from the organic carbon balance (Table IIIa.) and from the inorganic carbon balance (Table IIIb.) using the monitoring data. The results show a large variation in both cases. For the organic balance the mean valueis -0.28, while the inorganic balance based on free Carbon Dioxide gives a value of -0.17. In each case the mean temperature is 21.0 degrees C. The value obtained in a large number of practical cases for freshwater streams is -0.23 at 20.0 degrees C. based on BOD, not on DOC. However, it may be noted that the two values for rate constants found in this study average -0.23. It is worthy of note that at least two months of the monitoring data showed dissolved oxygen values so low as to raise questions as to their usefulness in rate constant calculations, and it was these data that resulted in the lowest rate constant values.

The calculations leading to the results presented in Table III suggested that the anoxia development in the bottom layer of the Main Stem could be explained analytically through application of a first order Streeter-Phelps deoxygenation formulation. To do this would require more reliable data on the organic carbon components of the water column and more extensive data on the course of oxygen concentration change in the bottom layer during the Spring and Fall transition periods to and from hypoxic conditions. However, the close coupling of the organic and inorganic carbon systems suggested that it might be possible to use the Streeter-Phelps formulation to calculate the Dissolved Oxygen values for 1984 and 1985, for which the organic carbon data were mis-

sing, by estimating an initial value for the Dissolved Organic Carbon from the inorganic carbon system and then applying the Streeter-Phelps approach to predict the depletion of Dissolved Oxygen in the bottom layer.

Table IV presents the results of such an analysis. This table shows a comparison between observed Dissolved Oxygen values in the bottom layer and those calculated as described above.
The calculated and observed Dissolved Oxygen values are in all cases in agreement within the limits of experimental accuracy. The results presented here use the rate constants calculated using the 1986 data to reproduce the actual Dissolved Oxygen values of 1984 and 1985, not only for the Summer months, but throughout the year where data are available. This analysis is basically an academic exercise to illustrate the close coupling between the organic and inorganic carbon systems and has no practical application in its present form. However, the method of calculation presented,e.g., a materials balance and use of the Streeter-Phelps formulation to predict the onset of low Dissolved Oxygen in the bottom layers of the Bay, could provide a reasonable basis for management decisions on combatting the anoxic conditions in the Bay at very little cost compared to large numerical models.

CONCLUSIONS

1. An extremely close coupling between the organic and inorganic carbon systems has been demonstrated; this relationship is sufficiently quantitative that the components of one system can be estimated from the components of the other.
2. Sediment load is not a significant factor in predicting the behavior of the oxygen of the bottom layer of the water column.
3. The behavior of the oxygen system of the bottom layers of the Bay can be predicted using a first-order formulation of the Streeter-Phelps type with very little expenditure for development of a practical model.
4. To develop a quantitative model based on this approach, the present monitoring program needs to be redesigned to acquire more data during the Spring and Fall when DO concentrations are in transition and to provide additional data on both carbon systems. High frequency data on pH, Temperature, Salinity, and DO in the bottom layer would minimize the need for other data, and in all likelihood make it possible to predict the onset of anoxia from these data alone. A cost effective way to do this would be to convert the existing manual monitoring program to one using consistently reliable remote sensing buoys.
5. The question of whether or not free Carbon Dioxide might be a limiting nutrient under salinity conditions existing in the Bay should be explored.

Table I

Changes in Organic Carbon in the Bottom Layer from Sediment Load

(Chesapeake Bay Mainstem between Bay Bridge and Potomac; Miles from Havre de Grace at midpoint of section between stations; all data from 1986 monitoring results)

a. Particulate Organic Carbon(mm/l)
+ - scouring
- - settling

Mile	May	June	July	August	September	October	November
20	0.07	-0.17	-0.20	-0.18	-0.11	0.15	0.01
50	0.13	-0.09	-0.17	-0.02	-0.13	0.01	-0.03
60	0.20	-0.01	-0.05	0.11	0.02	0.03	-0.03
70	0.02	-0.03	-0.07	-	-0.03	-0.30	0.50
80	0.01	-0.04	-0.09	-0.03	0.01	0.00	-
90	0.03	0.10	0.00	-0.02	0.06	0.03	0.02
100	0.03	-0.05	-	-	0.00	0.05	-

b. Dissolved Organic Carbon(mm/l)
+ - addition to the water column
- - loss from the water column

Mile	May	June	July	August	September	October	November
20	0.35	0.27	-0.55	-0.78	-0.55	-1.20	-1.26
50	0.36	0.15	0.05	0.14	-0.52	0.23	0.30
60	0.17	0.12	0.00	0.38	0.10	-0.07	-0.90
70	-0.16	-0.21	-0.38	-	-0.42	-2.02	-0.16
80	0.07	0.22	-0.44	-0.15	-0.09	-0.08	-
90	0.36	0.91	-0.14	0.06	-0.17	0.13	0.72
100	0.47	0.15	-	-	0.43	0.54	-

Table II

Coupling Between Organic and Inorganic Carbon Systems in the Bottom Layer of the Main Stem Chesapeake Bay

a. Difference Between Observed Bottom Layer Dissolved Organic Carbon Values and Those Calculated from the Inorganic Carbon Balance(mm/l)

Mile	May	June	July	August	September	October	November
20	-0.08	0.03	0.08	0.01	0.16	-1.50	-1.00
50	-0.02	0.05	0.07	-0.01	0.01	0.00	0.01
60	0.03	0.01	-0.02	-0.03	0.02	0.01	0.10
70	0.15	0.06	-0.07	-	0.16	0.17	-3.00
80	0.00	0.07	0.23	0.05	0.01	0.00	-
90	0.01	0.10	0.01	0.06	0.03	0.01	0.14
100	0.00	0.00	-	-	0.03	0.10	-

b. Difference Between Calculated Carbon Dioxide Generated and Observed Oxygen Consumed in the Bottom Layer(mm/l)

Mile	May	June	July	August	September	October	November
20	0.13	0.08	-0.22	0.23	0.21	-0.40	0.15
50	-0.07	0.05	0.04	0.08	0.28	0.06	0.20
60	0.12	0.07	0.02	-0.10	0.03	0.33	-0.30
70	0.12	0.05	0.08	-	0.12	0.54	-0.82
80	0.02	0.02	0.18	0.03	0.02	0.05	-
90	0.05	0.10	-0.10	0.05	0.10	0.05	-0.25
100	0.03	0.08	-	-	0.05	0.17	-

Table III

First Order Deoxygenation Rate Constants Calculated
from Chesapeake Bay Main Stem Monitoring Data
for the Bottom Layer Between the Bay Bridge and the Potomac

a. Rate Constants Calculated from the Organic Carbon Balance
Using the Streeter-Phelps Algorithm(per day)

Mile	May	June	July	August	September	October	November
20	-0.77	-0.71	-0.93	-0.06	-0.41	-0.45	-0.16
50	-0.40	-0.28	-0.17	-0.24	-0.24	-0.20	-0.27
60	-0.37	-0.18	-0.07	-	-0.20	-0.26	-0.32
70	-0.18	-0.40	-0.14	-	-0.28	-0.48	-0.67
80	-0.15	-0.16	-0.19	-	-	-0.19	-
90	-0.25	-0.25	-0.10	-0.03	-0.19	-0.15	-0.19
100	-0.28	-0.25	-	-	-	-0.20	-

Average(arithmetic mean) - -0.28

b. Rate Constant Based on Inorganic Carbon Balance -- Free
Carbon Dioxide Generation(per day)

Mile	May	June	July	August	September	October	November
20	-0.94	-	-0.05	-0.07	-0.02	-0.14	-
50	-0.03	-0.27	-0.28	-	-0.05	-	-
60	-0.44	-0.25	-0.01	-	-0.02	-0.32	-0.15
70	-	-	-0.02	-	-0.00	-0.32	-
80	-0.89	-	-	-	-0.01	-0.02	-
90	-0.05	-0.16	-	-	-	-	-
100	-0.28	-0.19	-	-	-	-	-

Average(arithmetic mean) - -0.17

TABLE IV

COMPARISON OF OBSERVED DISSOLVED OXYGE VALUES
IN THE BOTTOM LAYER WITH THOSE CALCULATED FROM
THE INORGANIC CARBON BALANCE
(STATIONS FROM POTOMAC TO BAY BRIDGE)

DATE	STATION	CALCULATED DO FREE CO2	CALCULATED DO TOTAL CO3	OBS.DO
RATE CONSTANT @ 20.0C -		-.23		
BOTTOM VELOCITY(FT/SEC) -			.820	
08/06/84	CB5.2	1.425	1.967	1.430
08/06/84	CB5.1	1.015	1.552	1.020
08/06/84	CB4.4	.785	1.319	.790
08/06/84	CB4.3C	-.005	.525	.000
08/06/84	CB3.3C	-.005	.522	.000
BOTTOM VELOCITY(FT/SEC) -			.180	
09/10/84	CB5.2	-.114	-.195	.120
09/10/84	CB5.1	-.173	-.250	.050
09/10/84	CB4.4	-.199	-.275	.020
09/10/84	CB4.3C	-.196	-.271	.020
09/10/84	CB3.3C	-.033	-.107	.180
BOTTOM VELOCITY(FT/SEC) -			.479	
09/24/84	CB5.2	2.571	2.986	2.570
09/24/84	CB5.1	2.091	2.506	2.090
09/24/84	CB4.4	1.731	2.142	1.730
09/24/84	CB4.3C	.001	.407	.000
09/24/84	CB3.3C	2.661	3.055	2.660
BOTTOM VELOCITY(FT/SEC) -			.691	
10/22/84	CB5.2	4.518	5.182	5.020
10/22/84	CB5.1	4.355	5.023	4.860
10/22/84	CB4.4	4.032	4.704	4.540
10/22/84	CB4.3C	2.330	3.005	2.840
10/22/84	CB3.3C	1.573	2.257	2.090
BOTTOM VELOCITY(FT/SEC) -			.459	
12/10/84	CB5.2	8.250	8.339	8.300
12/10/84	CB5.1	8.526	8.622	8.580
12/10/84	CB4.4	7.754	7.853	7.810
12/10/84	CB4.3C	8.412	8.515	8.470
12/10/84	CB3.3C	8.677	8.789	8.740
BOTTOM VELOCITY(FT/SEC) -			.154	
04/22/85	CB5.2	7.982	7.850	8.100
04/22/85	CB5.1	4.941	4.796	5.070
04/22/85	CB4.4	3.837	3.688	3.970
04/22/85	CB4.3C	2.943	2.789	3.080
04/22/85	CB3.3C	1.788	1.629	1.930

TABLE IV (CONTINUED)

DATE	STATION	CALCULATED DO FREE CO2	CALCULATED DO TOTAL CO3	OBS.DO
BOTTOM VELOCITY(FT/SEC) -			.258	
05/06/85	CB5.2	4.234	5.121	4.540
05/06/85	CB5.1	3.508	4.440	3.830
05/06/85	CB4.4	4.094	5.038	4.420
05/06/85	CB4.3C	3.405	4.375	3.740
05/06/85	CB3.3C	2.981	3.992	3.330
BOTTOM VELOCITY(FT/SEC) -			.246	
05/20/85	CB5.2	3.443	3.918	3.600
05/20/85	CB5.1	1.851	2.331	2.010
05/20/85	CB4.4	1.879	2.365	2.040
05/20/85	CB4.3C	.418	.907	.580
05/20/85	CB3.3C	-.165	.335	.000
BOTTOM VELOCITY(FT/SEC) -			.520	
06/03/85	CB5.2	3.903	4.475	3.860
06/03/85	CB5.1	3.144	3.718	3.100
06/03/85	CB4.4	1.593	2.161	1.550
06/03/85	CB4.3C	.073	.645	.030
06/03/85	CB3.3C	.044	.618	.000
BOTTOM VELOCITY(FT/SEC) -			.287	
06/17/85	CB5.2	.856	1.355	1.260
06/17/85	CB5.1	.546	1.033	.940
06/17/85	CB4.4	.328	.812	.720
06/17/85	CB4.3C	.220	.702	.610
06/17/85	CB3.3C	.764	1.241	1.150
BOTTOM VELOCITY(FT/SEC) -			.934	
07/08/85	CB5.2	3.099	5.439	2.880
07/08/85	CB5.1	2.796	5.107	2.580
07/08/85	CB4.4	2.675	4.968	2.460
07/08/85	CB4.3C	2.043	4.315	1.830
07/08/85	CB3.3C	.405	2.601	.200
BOTTOM VELOCITY(FT/SEC) -			.495	
07/22/85	CB5.2	.171	.095	.610
07/22/85	CB5.1	-.109	-.182	.310
07/22/85	CB4.4	-.103	-.174	.310
07/22/85	CB4.3C	.025	-.045	.430
07/22/85	CB3.3C	-.320	-.388	.070

TABLE IV (CONTINUED)

DATE	STATION	CALCULATED DO FREE CO2	CALCULATED DO TOTAL CO3	OBS.DO
BOTTOM VELOCITY(FT/SEC) -			.444	
08/06/85	CB5.2	.514	.856	1.250
08/06/85	CB5.1	-.609	-.280	.100
08/06/85	CB4.4	-.276	.047	.420
08/06/85	CB4.3C	-.525	-.212	.150
08/06/85	CB3.3C	.098	.400	.750
BOTTOM VELOCITY(FT/SEC) -			.680	
09/09/85	CB5.2	2.038	2.283	2.810
09/09/85	CB5.1	.103	.337	.840
09/09/85	CB4.4	-.236	-.005	.490
09/09/85	CB4.3C	-.453	-.226	.260
09/09/85	CB3.3C	-.612	-.395	.070
BOTTOM VELOCITY(FT/SEC) -			.697	
10/07/85	CB5.2	3.135	4.010	3.720
10/07/85	CB5.1	3.371	4.237	3.950
10/07/85	CB4.4	3.305	4.165	3.880
10/07/85	CB4.3C	2.919	3.773	3.490
10/07/85	CB3.3C	1.599	2.438	2.160
BOTTOM VELOCITY(FT/SEC) -			.747	
11/12/85	CB5.2	6.056	6.302	6.130
11/12/85	CB5.1	6.155	6.405	6.230
11/12/85	CB4.4	5.924	6.176	6.000
11/12/85	GB4.3C	6.604	6.856	6.680
11/12/85	CB3.3C	7.272	7.532	7.350
BOTTOM VELOCITY(FT/SEC) -			.145	
12/09/85	CB5.2	7.963	8.184	8.110
12/09/85	CB5.1	7.727	7.971	7.890
12/09/85	CB4.4	7.591	7.844	7.760
12/09/85	CB4.3C	7.686	7.947	7.860
12/09/85	CB3.3C	7.469	7.740	7.650

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CHESAPEAKE BAY SEDIMENT MONITORING FOR WATER QUALITY MODEL DEVELOPMENT

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INTRODUCTION

The Chesapeake Bay Program has a comprehensive modeling strategy consisting of three mathematical water quality models: the Watershed Model, the Steady State Eutrophication Model, and the Time Variable Eutrophication Model. The Watershed Model covers the entire 64,000 square miles of the Bay drainage basin and simulates pollutant loads delivered to the Bay from various land use, population, and point source treatment scenarios. Essentially complete, the Watershed Model is undergoing a series of refinements which will be concluded in 1989. The Steady State Model is designed to give an initial estimate of the relationships among nutrients, eutrophication, and anoxia, and to provide an initial evaluation of proposed nutrient control strategies. The Steady State Model was completed in the spring of 1987 and is fully successful in its application. The Time Variable Model will improve nutrient control strategy evaluation by projecting the degree and timing of the Bay response to control actions. The Time Variable Model will be capable of short-term simulations of critical episodic events (e.g. pycnocline tilting) and long-term simulations of about 30 years. Work was initiated on the Time Variable Model in October 1987, and will be completed in 1991.

Steady State Model results provided important guidance for the development of the Time Variable Model (HydroQual, 1987). Among the findings of the Steady State Model:

- The decline in dissolved oxygen (DO) in the bottom waters of the Bay

- Only management actions that reduce SOD and sediment nutrient flux improve Bay water quality to any significant degree.

The strong linkage between Bay sediments and water quality requires a multilayer sediment model to be incorporated into the Time Variable Model framework.

The sediment submodel will have three components: *net deposition* of particulate organic matter (POM), *diagenesis* of POM to dissolved inorganic components within the sediment, and *nutrient flux*, the movement of the dissolved inorganic nutrients from the sediment to the water column (Figure 1). Detailed modeling of Bay vertical processes requires an intensive sediment monitoring program to provide necessary data for model formulation, calibration, and verification. The sediment monitoring program described below will begin in April 1988 and continue for one year.

The sediment monitoring program is a cooperative effort of the Chesapeake Bay Program's Modeling and Monitoring Subcommittees with expert assistance from HydroQual, Inc., the participants of the Sediment Processes and Sediment Modeling Workshop, and the U.S. Army Corps of Engineers. The generous cooperation of the principle investigators participating in the sediment monitoring program is gratefully acknowledged. They are: Walter Boynton, Michael Kemp, Johnathan Garber, Peter Sampou, and Jeff Cornwell, University of Maryland; Richard Wetzel, Larry Hass, and Bruce Neilson, Virginia Institute of Marine Science; David Burdige, Old Dominion University; and Grace Brush, Johns Hopkins University.

EXISTING SEDIMENT DATA

Since 1984, the Maryland Department of the Environment (then the Office of Environmental Programs) has supported an integrated sediment, water column, and phytoplankton monitoring program called SONE (Sediment Oxygen and Nutrient Exchange) (Boynton et al., 1987). SONE focuses on the exchange of material between sediment, water column, and phytoplankton in the upper and mid-Bay (Figure 2). Incubated sediment cores are used to measure SOD and nutrient flux, and vertical arrays of sediment traps are used to measure movement of material between the sediment and water column. This ongoing study is foundational to the sediment monitoring program. The SONE study allows the use of an existing data set for a large portion of the Bay, and informed judgment as a guide for the sediment monitoring program. Other important studies of Bay vertical processes can be found in an excellent review and synthesis by Garber (1987).

REQUIRED SEDIMENT DATA

The close linkage between data collection and model development requires correct anticipation of data needs. Effort is concentrated on the mainbay and on lower estuary sites of major tributaries.

Sediment station locations

The sampling plan has a total of 25 stations located along the mainstem channel axis and in the lower tributaries (Figure 2). Stations 2, 4, 5, 7, 9, 10, 11, 12, 13, and 14 are existing SONE program stations (Boynton et al., 1987). Stations 6, 7, and 8 are the mid-Bay transect stations and include shallow lateral stations and a deep water station of a previous study (Malone et al., 1986). Stations 20, 21, and 22 are lower Bay transect stations.

Lateral transect stations are used in the mid-Bay and lower Bay to capture aspects of the vertical exchanges between nutrient generating deep waters and adjacent biologically

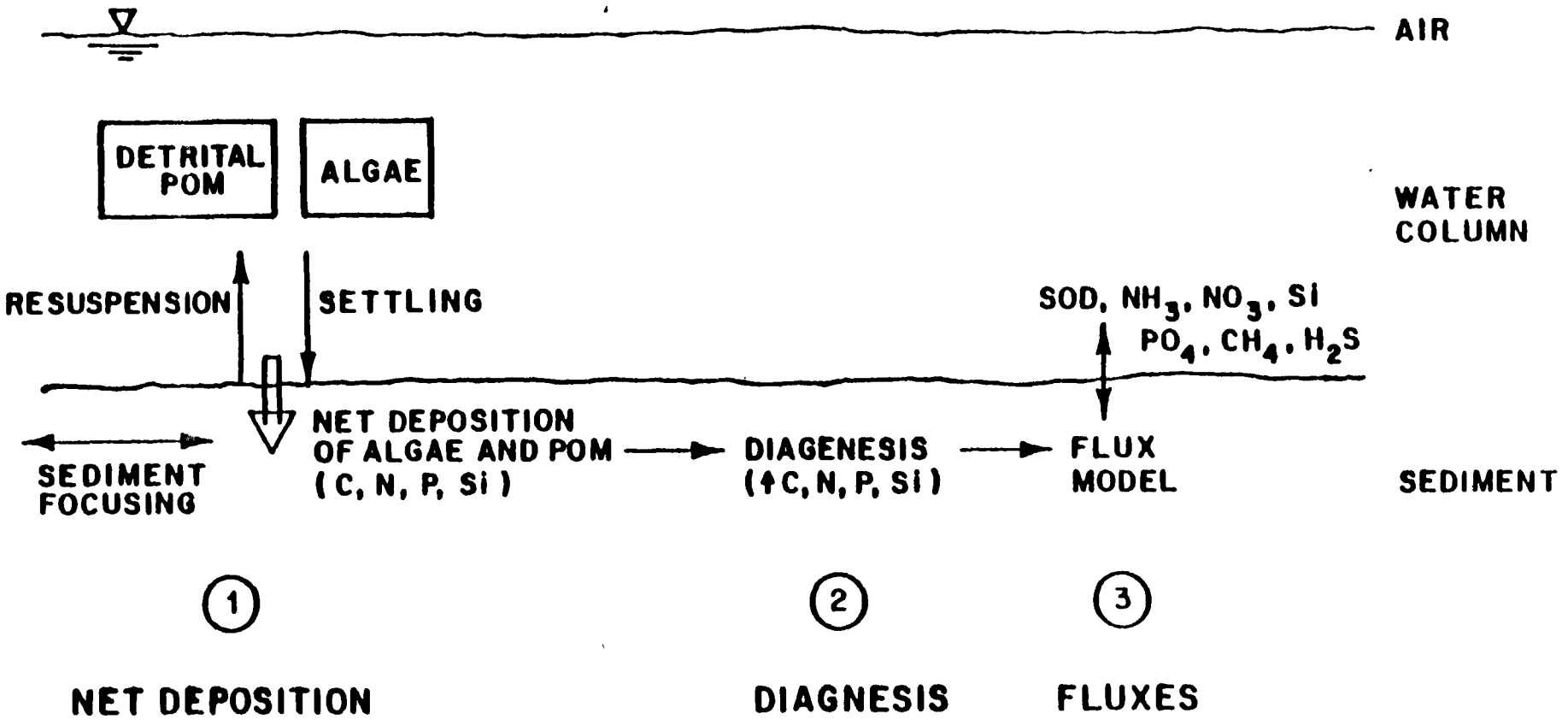

Figure 1. Principal Components of the Sediment Model (From HYDROQUAL, 1988)

productive shallow shelves (Malone et al., 1986). The mid-Bay transect stations are approximately centered in the area of summer anoxia. Southern transect stations are in an area of summer hypoxia (U.S. EPA, 1983).

There is very little existing information on lower Bay sediment/water column exchanges. Lower Bay stations 20, 21, 22, 24, and 26 are located according to homogeneous sediment types identified by Wright et al. (1986) as: estuary mouth shoals and spits, Bay-stem plains, Bay-stem channels, lower estuary or shallow bay muddy beds, and Bay mouth shoals respectively. The remaining eight stations are located relative to the 2-D model segmentation scheme. All sediment stations are located close to CBP water column monitoring stations.

Net Deposition

The net deposition component of the model simulates the input of inorganic and organic matter to the sediments. Remineralization of organic matter within the sediment provides material which contributes to SOD and dissolved nutrient fluxes. Inorganic matter contributes to sedimentation and advects organic inputs down into the sediments. Measurements of the net input of organic and inorganic matter to the bottom will be used to calibrate net deposition.

Two vertical array sediment traps will measure the net flux of solids across various water column layers and to the bottom. Each vertical array trap has three pairs of particle collectors located above the pycnocline, just below the pycnocline, and just above the sediment, as in the SONE program (Boynton et al., 1987). Particulate organic nutrients (particulate organic carbon [POC], particulate organic phosphorus [POP], particulate organic nitrogen [PON], particulate organic silica [POSi]), total solids, BOD, chlorophyll *a* and pheophytin, and general algal classification will be determined from the collected particulates. Algal identification of three broad functional groups (diatoms, non-diatom eucaryotes, and picoplankton), will provide information on phytoplankton occurrence, cell size, and settling rate. Sediment traps are located at two stations: station 7, an existing SONE station in the mid-Bay transect, and station 21 in the lower Bay transect. The sampling schedule results in 25 sampling periods a year with intensive weekly sampling in summer months between July and mid-September. Data from the sediment traps will be used for calibration of net deposition.

Sediment particulate organic profiles will be measured to determine the particulate organic material (POC, POP, PON, POSi, particulate organic sulfur [POS]) deposited over the years modeled in the long term simulations. The vertical distribution of bulk POM will be matched with coincident determinations of average sedimentation rates in duplicate cores. Radionuclide (^{14}C) and pollen dating techniques (Brush, 1984; Brush et al., 1982) will be used to determine average sedimentation rates, with particular emphasis on the profile between 1950 and the present. From this, long term average net deposition of refractory organic matter will be determined. The vertical profiles of particulate organics will be determined at all stations. Vertical profiles of pore water concentration will further characterize sediment composition at most sediment stations (Table 1).

Diagenesis

The diagenesis component of the model simulates the transformation of POM inputs to dissolved inorganic nutrients. There are three fractions of diagenic material: a labile fraction, a refractory fraction, and an inert fraction. These are empirical classifications based on a fraction that is fast reacting (labile) and is in thermodynamic equilibrium, a

Figure 2. Location of sediment monitoring stations.

■ = SONE station. Sone stations have four flux sample periods, except for stations 2 and 7.

● = CBP station. CBP stations have two flux sample periods,except for station 21.

□ = High frequency stations; six flux sample periods.

△ = Sediment trap.

fraction that is slow reacting (refractory) but has kinetic rates that are important to the model, and a fraction (inert) that does not react within the time frames of the model. The kinetic rates for these three fractions must be determined for the diagenesis component of the model. Organic material is transformed at rates which are a function of temperature, the degree of anoxia, the amount of organic matter present, the presence of other chemical constituents, and the sedimentation rate.

The sediment in the Bay contains a considerable amount of the refractory organic fraction that has not undergone complete decomposition. This heterogeneously distributed organic portion of the sediment continues to exert considerable SOD and contribute to nutrient flux from the sediments. Long term diagenesis will be determined from main-Bay and southern Bay tributary stations. Diagenesis rates will be determined by long-term sulfate depletion studies on sediment slurries from three sediment depths: a surface (0-2 cm) sample of recently deposited material, sediment of a "medium" age collected at a depth between 6 and 8 cm, and older deep sediments collected between 12 and 14 cm. Sediments will be incubated for 50 to 250 days. Total carbon dioxide (ΣCO_2), NH_4, NO_2, NO_3, PO_4, Si, SO_3, ΣH_2S, CH_4, and pH will be measured over time.

Nutrient flux

The flux component of the model completes the sediment cycle by returning inorganic nutrients to the water column. Model calibration requires extensive temporal and spatial coverage of SOD and nutrient flux measured under ambient bottom water conditions. Shipboard measurements of incubated intact cores will be the primary method of data collection. Cores will be collected and maintained at ambient conditions. Some observational measurement will be made of the effects of bioturbation. Fluxes of O_2, NH_4, NO2, NO_3, PO_4, Si, SO_3, ΣCO_2, CH_4, and ΣH_2S will be measured. Hydrogen sulfide flux will be measured at stations with overlying water DO < 1.0 mg/L.

Nutrient flux studies have three sampling frequencies. Eight stations (4, 5, 9, 10, 11, 12, 13, 14; the SONE stations) have a sampling frequency of four periods. Three stations (2, 7, 21) have a high sampling frequency of six sampling periods. The remaining stations (3, 4a, 6, 7a, 8, 16, 17, 18, 20, 22, 23, 24, 25, 26) have a low sampling frequency of two periods. Denitrification measurements have the same sampling frequencies as nutrient flux measurements but are limited to nineteen main Bay stations.

The sampling scheme is not systematic with respect to time; rather, sampling periods are established to coincide with an annual cycle of benthic processes. Sampling frequencies are based on the experience gained from the SONE program. All sediment stations with four or more sampling periods are sampled according to the following SONE sampling periods: "(1) a period (April-May) when the early spring phytoplankton bloom occurs, and nutrients (particularly nitrate) are high in the water column , (2) a period influenced by the presence of a large macrofaunal community (spring-early summer), (3) a period during which macrofaunal biomass is low but water temperature and water-column metabolic activity are high and anoxia is prevalent in deeper waters (August), and (4) a period in the fall when anoxia is not present and the macrofaunal community biomass is low but reestablishing" (Boynton et al.,1987).

To improve temporal coverage of sediment nutrient flux and denitrification, three high frequency stations have two additional sampling periods. Additional sampling periods include: (1) a period in the winter (December - early March) when anoxia is not present, metabolic activity is low and nitrate is increasing in the water column, and, (2) a period in the early fall (September) just prior to the break-up of anoxia .

The majority of the stations (14 out of a total of 25 stations) have a low sampling frequency of two periods; one sample period when anoxia is prevalent in deeper waters (August), and one when anoxia is not present (spring-early summer). Location and frequency of the various field and laboratory measurements are described in Table 1.

Paired with the measurement of ambient nutrient flux are measurements of anoxic fluxes. Anoxic fluxes are measured at all stations with the same frequency as ambient fluxes. Anoxic fluxes are designed to measure rapid, short-term changes in flux due to the die-off of benthic infauna and the rapid chemical changes caused by decreasing oxygen concentrations and oxidation-reduction potential. Pore water concentrations of NH_4, NO_2, NO_3, PO_4, Si, SO_3, ΣCO_2, CH_4, ΣH_2S, Fe, Mn, and pH will be measured with short-term anoxic incubations of surficial (0-2 cm) sediment.

Denitrification is a major sink for nitrogen in the Bay and has seasonal and region-specific properties which must be delineated for successful sediment model development (Twilley and Kemp, 1985). Three methods for measuring denitrification will be used. At all stations sampled, an acetate inhibition method will be used with the same sampling frequency as for nutrient flux . Acetate inhibits the final reaction (with N_2 the product) of the denitrification reaction path. This allows the analysis of an intermediate product concentration without the problem of background contamination. Nitrification potential will be measured at a limited number of stations by N-serve treated control sediment slurries, an inhibition technique that prevents nitrification in the treated slurries. Control slurries will be compared to untreated test cores. Calibration of the denitrification and nitrification measurements will be with ^{15}N labeled nitrate and ammonia respectively. Details of sampling periods and times are in Table 1.

CONCLUSION

Work on acquiring data for calibrating the sediment submodel will be initiated in April, 1988. The sediment monitoring program is based on the consensus recommendations of the expert panel of the Sediment Processes Workshop, and the Modeling and Monitoring Subcommittees of the Chesapeake Bay Program. The sediment data outlined above are essential to the Time Variable Model of the Chesapeake Bay.

TABLE 1. SEDIMENT PLAN ACTIVITIES, STATIONS, AND SAMPLING FREQUENCY.

ACTIVITY	METHODS	STATIONS	SAMPLING FREQUENCY
Nutrient Flux	ambient shipboard	3,4a,6,7a,8,16,17,18, 20,22,23,24,25,26	2 periods
		4*,5*,9*,10*,11*,12*, 13*,14*	4 periods
		2*,7*,2	6 periods
	anaerobic	same as above	same as above
	transitive	2*,7*,21	6 periods
Denitrification	acetlyene blockage	3,4a,7a,16,18,20,21	2 periods
		4,14	4 periods
		2,7,21	6 periods
	nitrification (N-serve)	2,7,21	6 periods
	^{15}N	2 stations^	2 periods
Sediment Traps	vertical array particle traps	7*,21	25 periods
Particulate Organic Profile	depth profile of POM	all stations	1 period
Pore Water Profile	pore water concentration	2,3,4,4a,5,6,7,7a,8,9,10, 11,12,14,16,18,20,21,22	1 period
Recent Rates of Sedimentation	pollen dating	all stations	1 period
Long Term Diagenesis	sulfate depletion	2,3,4,4a,6,7,7a,8, 14, 16,17**,18,20,21,22, 23**,24**,25**,26**	1 period

* SONE stations
^ Not Determined
** two sample periods for surface sediments

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IMPLICATIONS OF TOXIC MATERIALS ACCUMULATING IN THE SURFACE MICROLAYER IN CHESAPEAKE BAY

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ABSTRACT

The aquatic surface microlayer is subject to the spontaneous, thermodynamically driven enrichment of naturally occurring surface active molecules (surfactants). These in turn may serve as ligands or solvents for toxic metals, chlorinated, saturate and polyaromatic hydrocarbons. Toxic substances that are themselves surface active, such as TBT (tributyltin) and its derivatives will readily accumulate in the microlayer as well. There is a growing body of evidence that shows enrichment ratios of one to several orders of magnitude in samples of toxic substances taken from estuarine, coastal, and lake waters on a worldwide scale. In Chesapeake Bay, limited sampling has shown elevated levels of metals and hydrocarbons (alkanes and polyaromatics) on the upper tidal Potomac River and at three northern Bay stations (Susquehanna, Elk, and Patapsco Rivers). Sources implicated appear to include aerial deposition and surface run-off. Additional sampling during autumn 1987, i.e. a time least likely to suggest pesticide presence, nevertheless showed detectable levels of 23 different toxic organics and pesticides in upper bay microlayer samples. Bulk water samples, by comparison, rarely had detectable levels of the same compounds. Only few investigators have sought to demonstrate the susceptibility of exposure to and effects on aquatic organisms that are neustonic in at least part of their life cycle. Preliminary work by others in the North Sea, southern California coastal waters, and by one of us on Puget sound and in

Gulf Stream waters near the straits of Florida showed selectively high mortality to some indicator organisms. We have tentatively identified eggs and larvae of the bay anchovy (Anchoa mitchilli), atlantic silverside (Menidia menidia), hogchoker (Trinectes maculatus) as well as the copepod (Acartia tonsa) as candidate vertebrate and invertebrate indicator organisms for bioassays. Suitable bioassay techniques for in-situ and laboratory testing of microlayer water need to be developed in order to assess both the biological significance of toxic materials enrichment and the degree to which this enrichment is found. The resultant information should provide further assessment relating the role of microlayer contamination to the decline of Chesapeake Bay living resources. If significant effects can be traced to sources contributing toxics to the surface microlayer and if these concentrations affect the living resources, their control measure should be explored, evaluated and implemented.

INTRODUCTION

The boundary between the atmosphere and the aquatic environment is an important biological habitat and a collection point for pollutants. The eggs and larvae of many fish and shellfish species float on, or come in contact with, the water surface throughout their early development. The aquatic surface microlayer (surface microlayer), operationally defined as 50-100 um thick, serves as a concentration point for metal and organic contaminants that have low water solubility or are associated with floatable particles. Recent studies have linked aquatic surface contamination with negative biological impacts. In Puget Sound (Hardy et al., 1988a), Southern California (Cross et al., 1987), and the North Sea (Kocan et al., 1987), fish eggs exposed to contaminated surface microlayer exhibited reduced viability.

We report here work on the evaluation of microlayer samples taken from six stations in the upper Chesapeake Bay (Figure 1) and identify living resources potentially impacted. Additionally, we explore necessary features of a plan for assessing the biological significance of the observed enrichment of toxic contaminants.

Organic molecules, collectively called surfactants (surface-active agents), are thermodynamically driven to remain at the interface because they lower the surface free energy. Dominant molecules appear to be biogenic, are long-chain, and of high molecular weight. The enriched interface is capable of trapping other molecules, both dissolved and particulate, toxic and benign. This matrix forms a substrate for bacterial growth as seen by 1 to 4 orders of magnitude higher bacterial counts in the microlayer than the subsurface water. In estuarine waters the time scale for such initial enrichment is short, a time frame of hours or tens of hours is likely (Gucinski, 1985, 1986; Olson, 1983; Crow et al., 1975; Sieburth, 1982; Hartwig and Herr, 1984).

Many taxa, known to be neustonic offshore, also occur in the Bay. Table 1 is a listing of taxa clearly found to be neustonic in marine waters of the Mid-Atlantic Bight (Grant, 1979). Comparing his findings to species lists published by Wass (1972) and Lippson et al. (1979), one can identify those species occurring in the Bay, and putatively classify them as neuston as well. Exceptions do exist. For example, the larval stage of the blue crab (Callinectes sapidus) is neustonic in offshore waters, but once transported into Bay waters, its zoeae and megalopae are pelagic (Provenzano et al., 1982; 1983). Marine neuston studies cannot be extrapolated to the low salinity zones of the Bay. Clearly, a need exists to characterize the neuston of Chesapeake Bay.

On the mid-Atlantic shelf the larvae of commercially valuable species such as menhaden, hake, cod, bluefish, lobster, and blue crab, occur in greater concentrations in surface water than in subsurface water (Grant, 1979, Castagna, 1977). The copepod Acartia tonsa, various species of amphipod, the bay anchovy, atlantic silverside and hogchocher, are all known to have egg, larval, or adult stages that contact the surface microlayer. The exposure risks that these selected species undergo when toxic materials are enriched or resident in the microlayer need to be assessed. In addition, to estimate exposure duration, information is needed on the residence times in the microlayer for zooplankton, larvae, and eggs.

Studies conducted in the spring of 1986 (Hardy et al., 1987) suggested that serious surface microlayer contamination, consisting of complex mixtures of chemicals, occurs in Chesapeake Bay. Contamination levels suggested the presence of three major zones —the upper Bay, with high levels of contamination; the Potomac River with moderate to high levels; and the southern and eastern shore, with a different chemical mixture and generally low levels of contamination. Our data result from autumn sampling and cover an extended range of contaminants identified as potentially harmful by the EPA.

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METHODS

Neuston collections were limited to waters of the upper Chesapeake Bay (entirely within the state of Maryland) and did not include night-time tows. The rectangular mouth (0.46 x 0.3 m) zooplankton net, had a mesh size of 100 um and sampled a surface-water volume of 9 cubic meters when immersed to an average depth of 0.05 m. In a few cases the volume sampled was estimated

at about 4 cubic meters rather than calculated. Samples were preserved in buffered 5% formalin for later analysis.

Samples of the sea-surface microlayer were collected from 6 sites in Chesapeake Bay between September 10 and 12, 1987 using the rotating teflon drum microlayer sampler and repeated during October, 1987. because of the loss of some samples in shipment. Methods followed in general those described previously for similar samples (Hardy et al., 1986; Hardy et al., 1987; Hardy et al., 1988). In addition, samples were analyzed for concentrations of tributyl tin by hydride generation and atomic absorption detection. Analyses performed at EPA contract labs followed standard methodology as specified by the Office of Water Regulation and Standards.

Microlayer sampler collection efficiency determinations were made with Lycopodium spores, a hydrophobic particulate representative of aerially deposited materials and radio-labelled 14C DDT dissolved in oleyl alcohol, used to represent a dry surfactant that forms monolayers.

Table 2 summarizes the results for particulate recovery, done on seawater, except for a single run on tapwater at 0 mN/m spreading pressure.

Counts of 14C DDT indicated a recovery which increased with increasing film pressure and was 78.7, 80.3 and 88.2%, for mean surface pressures of 4.6, 19.6 and >23 mN/m, respectively. Collection efficiencies of 2 microlayer samplers tested here are less than 100%, and vary both as to the substance recovered as well as with the presence of slick forming molecules, as determined by surface pressure or surface tension measurement. Within slicks, such efficiencies exceed 85%, and are much less, from 60% to nearly 80% when surface waters are "free" of organized, coherent films.

RESULTS

Our analysis of neuston tows collected this summer and autumn from four sites in Chesapeake Bay indicates the presence of at least 20 abundant taxa dominated by the copepod Acartia tonsa (Table 3). All tows were conducted in daytime and only zooplankters were collected. The limited scope of the study (no replicate sampling) and late season of the sampling could not comprehensively represent overall neuston abundance and diversity. Results indicate that surface-dwelling organisms occur at very high densities in the areas sampled with a mean density > 7000 individuals/cubic meter. For comparison, densities of total zooneuston in Puget Sound collected with a similar net were about 100 to 400 individuals/cubic meter with copepods dominating the community. In Chesapeake Bay, the copepod Acartia tonsa could represent an important prey item for surface feeding fish or other organisms.

Samples collected during September and October 1987 (Station locations are shown in Figure 1) showed elevated concentrations of metals in the microlayer. Of particular interest, in terms of potential toxicity, were the concentrations and/or enrichments (microlayer/bulkwater concentrations) of silver (indicative of sewage inputs) at Stations 7, 8, and 12, copper at all stations, and arsenic, lead and zinc at Station 3. The total microlayer concentrations of Ag+Cu+Cd+Pb+Zn ranged from 59 to 642 ug/l. We are presently reconfirming the results of these measurements.

Pesticides and other organic compounds were enriched in the microlayer compared to the bulkwater samples at several sites. Microlayer concentrations are shown in Table 4. Enriched microlayer concentrations occurred at two or more stations for the following: carbophenothion, demeton, diazinon, di-butyl phthalate, EPN, ethion, famphur, fensulfothion, and kepone. In general, Stations 7 and 8 were most contaminated; e.g. the microlayer at Station 8 was particularly enriched in dieldrin. Dieldrin has been found in urban run-off at concentration from 8 to 100 ng/l, values in the same range as seen here (EPA, 1982).

October 1987 samples (Table 5), analyzed by Battelle, had high concentrations of organic contaminants in the microlayer at most stations. Concentrations of organotin ranged from 30 to 349 ng/l in the microlayer and 60 to 90 ng/l in the two bulkwater samples analyzed.

Concentrations of total aromatic hydrocarbons in the surface microlayer of Chesapeake Bay ranged from 0 to 20 ug/l. Table 6 lists concentrations at or above the detectable level. Spike recovery measurements on the samples using surrogate aromatic hydrocarbons suggested that only 35 to 100% (mean 69%) of the aromatic hydrocarbons were recovered, i.e. our reported values represent roughly 69% of the actual concentrations present in the sample. Aromatic hydrocarbon concentrations were low or below detection at Stations 2 and 3, significant (potentially toxic) at Stations 7, 11, and 12, and very high at station 8 (Susquehanna River). Concentrations of total saturate hydrocarbons ranged from 3.8 to 66.5 (mean 21.3) ug/l (Table 7). Highest concentrations occurred at Stations 3, 7 and 8.

Pesticide and chlorinated organic compounds were largely undetected in surface microlayer samples taken in October, with the exception of dieldrin. Dieldrin occurred in concentrations of 1 to 18 ng/l except at Station 2, where it was absent. As was the case with most of the other contaminants, high concentrations of pesticides were found at Stations 3, 7, and 8.

DISCUSSION

In recent years, toxicity tests of sediment contamination have involved the development of an environmental quality triad (chemical, bioassay and infauna) to determine environmental impact

(Long and Chapman, 1985). Chapman and Long (1983) argued that for accurate evaluation of sediment quality, at least three categories of measurement must be evaluated. These are: (1) concentrations of toxic chemicals, (2) toxicity of the environmental samples (bioassay), and (3) evidence of modified resident biota, - particularly the infauna.

The same approach can be used for determining the effects of surface microlayer contamination on neustonic eggs and larvae. Thus, accurate evaluation of surface waters may involve at least five categories of sampling, testing, and evaluation. These are: 1) collection of surface microlayer samples, 2) determination of concentrations of toxic chemicals, 3) collection and enumeration of representative resident neuston species populations, 4) toxicity tests (laboratory bioassays), and 5) toxicity to representative neustonic organisms (field bioassays). Certainly all five measures are necessary to get an accurate picture of the physical, biological and chemical parameters that contribute to aquatic surface quality. At this stage, the toxicity evaluation should involve both laboratory bioassays with standard organisms, such as the sea urchin, and selected field bioassays, with important resident organisms such as the bay anchovy or crab larvae. To determine the toxicity of surface microlayer in Chesapeake Bay a dual approach, consisting of both in-situ and laboratory bioassays may be useful. In-situ studies simulate natural conditions, but are often unable to determine controlling variables (e.g., temperature and salinity). Organisms for field bioassays are not always available which leads to unproductive field time. If no difference exists in the results of toxicity tests using fresh versus frozen surface microlayer samples, the samples could be collected throughout the year and tested on seasonal spawning species when eggs are available. Nevertheless, laboratory studies are needed because they allow better control and provide a basis for accurate hypothesis testing. Both types of bioassay are necessary.

Floating pelagic fish eggs are particularly suitable for studies of aquatic surface toxicity. For example, eggs of anchovy (Hunter, 1981), sole (Hardy, 1987), and mackerel (Longwell, 1976, 1980) have been used successfully as sensitive indicators of toxicity. The eggs of such pelagic spawners are often distributed in extremely patchy but dense concentrations. Eggs are frequently present in only 5% of the neuston net trawls, but when found, are often in densities of 17 to 31 eggs/L. This is the equivalent of up to 46,000 eggs per 10 square meters of water surface. These patches originate from intensive spawning activity and gradually disperse (Hunter, 1981).

Two fish, in particular, are important ecosystem components and produce floating eggs in large numbers. Neuston net tows at the South Island of the Chesapeake Bay Bridge-Tunnel indicated maximum egg densities in mid-June and mid-July for the hogchoker (Trinectes maculatus) and the bay anchovy (Anchoa mitchilli), (Birdsong, pers. comm.). Blue crab (Callinectes sapidus) occurred

in high densities near the surface in mid-July to mid-August. Also, the zoeal larvae of the blue crab concentrate at the surface and can be cultured in the laboratory. We recommend the bay anchovy and the blue crab larva as appropriate species for assessing surface microlayer toxicity in Chesapeake Bay. The anchovy egg is a representative pelagic fish egg that contacts the surface during an approximately four-day period during development, is widespread and euryhaline, and can be collected in large numbers during the summer using neuston net tows. Blue crab larvae represent the reproductive stage of an important commercial shellfish resource. They are typical of crustacean neuston that probably feed on the high densities of microorganisms at the water surface (Zaitsev, 1971). Also, they can be cultured and used for toxicity tests in the laboratory.

Our research in Puget Sound suggested that toxicity to pelagic fish eggs and other organisms resulted from a complex mixture of contaminants, with no single compound or group of compounds responsible for the overall toxicity (Hardy et al., 1988). We do not yet have toxicity measurements for Chesapeake Bay microlayer. To obtain a relative measure of toxicity one may enter the data from this study into our microlayer toxicity model (Hardy et al., 1988). The results of the cumulative impact suggest surface contamination in Chesapeake Bay may be responsible for a reduction in the survival of neuston, including the hatching success of pelagic fish eggs. Predicted toxicity would be highest at stations 3, 7 and 8. This estimate, based on a limited data set, is uncertain, but is probably conservative because it does not take into account the possible effects of the organotin found in our samples. A comprehensive study, including simultaneous measurements of toxicity and concentrations of contaminants, should be conducted in Chesapeake Bay.

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TABLE 1: PELAGIC ZOOPLANKTON FOUND IN NEUSTON TOWS AND ALSO FOUND IN LOWER CHESAPEAKE BAY (after Grant, 1979 , Wass, 1972 , Lippson et al, 1979)

CNIDERIA (comb jellies)
- *Liriope* sp.

COPEPODA (zooplankton)
- *Acartia* sp.
- *Acartia tonsa*
- *Calanus finmarchicus*
- *Candacia armata*
- *Centropages furcatus*
- *Centropages hamatus*
- *Centropages typicus*
- *Corycaeus* sp.
- *Eucalanus* sp.
- *Eucalanus pileatus*
- *Labidocra* sp.
- *Labidocera aestiva*
- *Oithona* spp.
- *Oncaea venusta*
- *Paracalanus parvus*
- *Paracalanus quasimodo*
- *Pontella* sp.
- *Pontella meadii*
- *Pseudocalanus* sp.
- *Temora longicornis*
- *Temora stylifera*
- *Temora turbinata*

ECHINODERMATA (including seastars, etc.)
- unident. ophiuroids

CHAETOGNATHA (arrow worms)
- *Sagitta elegans*
- *Sagitta enflata*
- *Sagitta hispida*
- *Sagitta tenuis*
- unident. chaetognaths

MOLLUSCA (mussels & clams)
- *Dosinia discus*
- *Melampus bidentatus*
- *Spisula solidisima*

MYSIDACEA (mysid shrimp)
- *Heteromysis formosa*
- *Mysidopsis bigelowi*
- *Neomysis americana*

CUMACEA
- *Oxyurostylis*

ISOPODA
- *Edotea triloba*
- *Idotea metallica*

AMPHIPODA
- *Corophium* sp.
- *Stenothoe* sp.

DECAPODA (principally crabs)
- *Callianassa* sp.
- *Callinectes* sp.
- *Crangon septemspinosa*
- *Homarus americanus*
- *Homola barbata*
- *Latreutes fucorum*
- *Leptochela* sp.
- *Libinia* sp.
- *Munida* sp.
- *Pinnixa cylindrica*
- *Portunus* sp.

PISCES (fish, eggs, larvae)
- *Astroscopus guttatus*
- *Cynoscion regalis*
- *Hippocampus* sp.
- *Menidia menidia*
- *Scomberesox saurus*
- *Scophthalmus aquosus*
- *Sphoeroides* sp.
- *Syngnathus* sp.
- *Syngnathus fuscus*
- *Urophyics* sp.
- *Urophycis regius*

TABLE 2: LYCOPODIUM SPORE SAMPLING EFFICIENCY AS A FUNCTION OF SPREADING PRESSURE

	Surface Spreading Pressure mN/m			
Sampler	0	<0.82	4.4	18.8
Drum	64.4	18.4*	74.3	89.8
Glass plate	31.1	55.2	37.6	99

*Visual observation showed spores pushed away from sampler, suggesting some surfactant contamination that produced its own spreading pressure.

TABLE 5: BUTYL TIN (ng/l AS INORGANIC TIN) IN CHESAPEAKE BAY MICROLAYER AND BULKWATER SAMPLES COLLECTED OCTOBER, 1987

	Butyl Tin (ng/l)	
Location	Microlayer	Bulkwater
*Baltimore Harbor	80	
Susquehanna River	299	
Upper Potomac	349	
*Matapeake	200	90
*Choptank River	70	60
*Point Lookout	30	

*Data requires additional verification.

TABLE 3: NEUSTON CONCENTRATIONS IN THE UPPER CHESAPEAKE BAY

Choptank River

TAXA:	Number per m^3	Percent Total
Acartia tonsa	9362	76.71
copepod nauplii	942	7.72
barnacle nauplii	626	5.13
Bosmina longirostris	312	2.56
Camptocercus rectirostris	208	1.70
insecta	184	1.50
shrimp larvae	184	1.50
Moina micrura	184	1.50
Podon polyphemoides	92	0.75
Chydorus sp.	92	0.75

Matapeake

TAXA:	Number per m^3	Percent Total
Acartia tonsa	2167	50.01
copepod nauplii	557	12.85
Bosmina longirostris	1052	24.28
Podon polyphemoides	62	1.43
Cyclops vernalis	62	1.43
Cyclops bicuspidatus	371	8.57
Diaphanosoma	62	1.43

Elk River

TAXA:	Number per m^3	Percent Total
Acartia tonsa	495	95.38
Diaphanosoma	21	4.05

Susquehanna River

TAXA:	Number per m^3	Percent Total
Acartia tonsa	4800	38.46
insecta	1280	10.26
Chydorus sp.	320	2.56
Cyclops vernalis	2880	23.08
unident'd, dmaged	3200	25.64

TABLE 4: CONCENTRATIONS (ug/l) OF PESTICIDES AND ORGANIC COMPOUNDS IN THE SURFACE MICROLAYER AND BULKWATER OF CHESAPEAKE BAY SEPTEMBER, 1987. ANALYZED BY EPA. M=MICROLAYER, B=BULK SEA WATER

Compound:	Station: Baltimore Harbor		Susquehanna River		Matapeake		Potomac River	
	M	B	M	B	M	B	M	B
azinphos ethyl	nd*	nd	<1!	nd	nd	nd	nd	nd
azinphos methyl	nd	nd	<1	nd	nd	nd	nd	nd
captafol	<0.2	<0.2	nd	nd	<0.25	nd	nd	nd
carbophenothion	nd	nd	<0.5	nd	<0.5	nd	nd	nd
chlorfevinphos	<0.5	<0.5	<0.5	nd	nd	nd	<nd	<0.5
coumaphos	nd	nd	nd	nd	nd	nd	<2	nd
crotoxyphos	<1	<1	<1	nd	nd	nd	nd	nd
demeton	nd	nd	<1	<1	nd	nd	nd	<1
diazinon	<0.5	nd	nd	nd	nd	nd	<0.5	<0.5
dichlorvos	nd	nd	<0.5	nd	nd	nd	nd	nd
m-di-butyl-phthalate	50	nd	nd	nd	nd	nd	nd	nd
EPN	<0.5	nd	nd	nd	nd	nd	<0.5	nd
ethion	<2	nd	<2	nd	nd	nd	nd	nd
famphur	nd	nd	<0.5	nd	<0.5	nd	<0.5	<0.5
fensulfothion	<1	nd	<1	nd	nd	nd	nd	nd
kepone	nd	nd	<0.12	nd	<0.12	nd	nd	nd
leptophos	nd	nd	<0.5	nd	nd	nd	nd	nd
monocrotophos	nd	nd	nd	nd	nd	nd	<5	nd
phosmet	<1	nd	nd	nd	nd	nd	nd	nd
terbufos	nd	<1.2	<1.2	<1.2	<1.2	nd	nd	nd
tetrachlorovinphos	nd	nd	nd	nd	nd	nd	<0.5	<0.5
thio-bis-methane	nd	nd	nd	nd	nd	nd	nd	nd
trichlorofon	nd	nd	<1	nd	nd	nd	nd	nd

*Not detectable. ! Present at or below detection limit

TABLE 6: AROMATIC HYDROCARBONS IN THE SURFACE MICROLAYER OF CHESAPEAKE BAY, OCTOBER, 1987

Compound:	Baltimore (ug/l)	Choptank (ug/l)	Matapeake (ug/l)
Phenanthrene	0.1	0.0	0.0
Fluoranthene	0.2	0.1	0.0
Pyrene	0.1	X	0.0
Benz(a)anthracene	0.0	X	0.0
Crysene	0.0	X	0.0
Benzo(k)fluoranthene	0.0	X	0.0
Benzo(e)pyrene	0.0	X	0.0
Total (all compounds)	0.4	0.1	0.0

X = Present, but below detection limit of 0.05 ug/l.

Compound:	Susquehanna River (ug/l)
Acenaphthene	0.1
Fluorene	0.4
Phenanthrene	3.4
Anthracene	1.0
C1-P/C1-A	1.0
Dibenzothiophene	0.1
Fluoranthene	5.3
Pyrene	3.5
Benz(a)anthracene	1.4
Chrysene	2.8
Benzo(b)fluoranthene	0.8
Benzo(k)fluoranthene	0.1
Benzo(e)pyrene	0.2
Benzo(a)pyrene	0.1
Total (all compounds)	20.2

AROMATIC HYDROCARBONS SCANNED BUT NOT DETECTED:

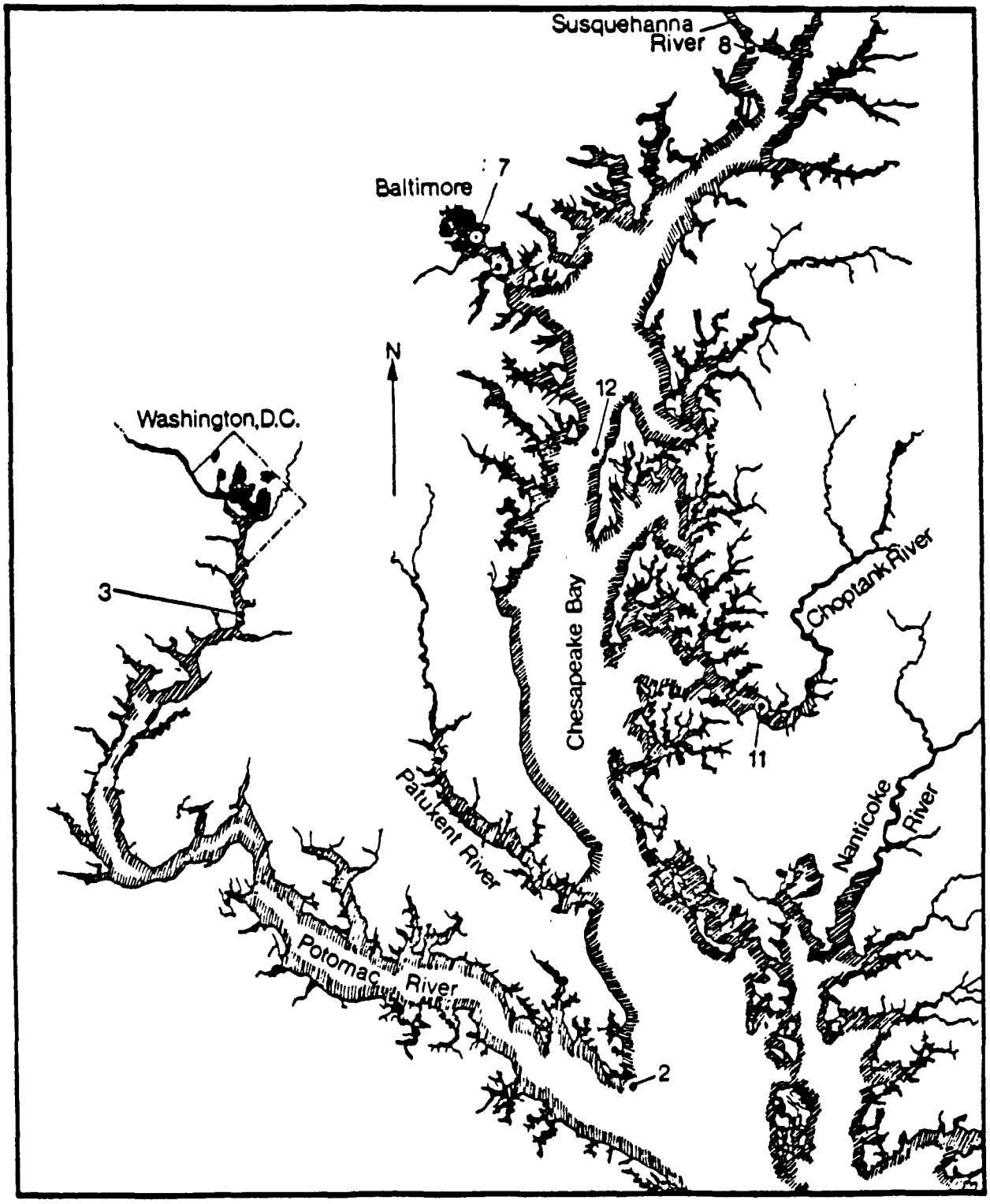
Naphthalene	C1-Fluorene	C1-D
C1-Naphthalene	C2-Fluorene	C2-D
C2-Naphthalene	C3-Fluorene	C3-D
C3-Naphthalene	C4-Fluorene	C4-D
C4-Naphthalene	C2-P/C2-A	Benzo(g,h,i)perylene
Acenaphthylene	C3-P/C3-A	Perylene
Biphenyl	C4-P/C4-A	Indeno(1,2,3-CD)pyrene
Dibenz(a,h)anthracene		

TABLE 7: CONCENTRATIONS (ug/l) OF SATURATE HYDROCARBONS IN SURFACE MICROLAYER OF CHESAPEAKE BAY, OCTOBER 1987

	Station:							
Compound:	2	3	7	8	11	12A	12B	Blank
Heptadecane	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0
Pristane	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0
Octadecane	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0
Phytane	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nonadecane	0.0	0.0	1.2	0.0	0.1	0.0	0.0	0.0
Eicosane	0.0	0.0	1.1	0.0	0.3	0.0	0.0	0.0
Henicosane	0.0	0.0	2.3	0.3	0.4	0.0	0.0	0.0
Docosane	0.0	0.0	5.5	0.6	0.4	0.0	0.0	0.0
Tricosane	0.0	0.3	9.8	1.9	0.2	0.0	0.1	0.0
Tetracosane	0.0	0.2	11.8	2.0	0.0	0.0	0.0	0.0
Pentacosane	0.0	0.7	10.7	4.2	0.0	1.8	0.4	0.0
Hexacosane	0.0	0.3	7.8	2.1	0.3	0.0	0.4	0.0
Heptacosane	0.9	1.9	4.8	6.3	0.5	0.9	1.0	0.0
Octacosane	0.1	0.2	3.2	1.6	0.2	0.0	0.0	0.0
Nonacosane	2.1	4.5	3.4	18.3	1.1	0.0	1.1	0.0
Triacontane	0.0	0.3	1.2	1.5	0.5	0.0	0.0	0.0
Hentriacontane	0.8	2.4	1.8	10.5	0.9	2.9	0.7	0.0
Dotriacontane	0.0	0.0	0.2	0.4	0.0	0.0	0.0	0.0
Tritriacontane	0.0	0.2	0.7	1.7	0.2	0.0	0.0	0.0
Tetratriacontane	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0
OTP	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
Isopronoid 1380	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
Farnesane 1470	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0
Isopronoid 1650	0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.0
Total	3.9	11.0	66.5	51.2	5.8	7.0	3.8	0.0
Mean for All Stations		24.0						

Station #	Location:	Scanned but not detected:	
2	Point Lookout	Decane	Undecane
3	Upper Potomac	Dodecane	Tridecane
7	Baltimore Harbor	Tetradecane	Pentadecane
8	Susquehanna River	Hexadecane	
11	Choptank River		
12a	Matapeake		
12b	Matapeake, Bulkwater		

Figure 1. Station Locations



POTENTIAL BIOLOGICAL EFFECTS OF MODELED WATER QUALITY IMPROVEMENTS RESULTING FROM TWO POLLUTANT REDUCTION SCENARIOS

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INTRODUCTION

The use of water quality models to make environmental projections is popular in contemporary management circles. It is important, however, to relate model output in practical terms to the attainment of beneficial uses; especially restoration of both a balanced ecological system and the recreational and commercial resources which make man eager to pay the bill for his transgressions. This paper is an approach to satisfy that immediate need in Chesapeake Bay and is companion to App and Fitzpatrick (1988) elsewhere in this volume.

In an effort to efficiently allocate limited resources, the Environmental Protection Agency (EPA), together with a cooperative state/federal management structure called the Chesapeake Bay Program (CBP), sponsored development of a state of the art steady-state coupled hydrodynamic/water quality model (HydroQual, Inc. 1987). This computer model, known as the Chesapeake Bay Program Steady-State Model, is used as a management tool to calculate and project water quality conditions based on a number of management scenarios. The model develops estimates of selected environmental conditions in the Bay and presents them as an average summer "steady-state" condition (defined as the 62-day period covering July and August). Many users and reviewers of the voluminous and complex model output have had difficulty relating this product to meaningful improvements in the Bay's living resources.

In this paper, we present some of the mechanisms by which the estuarine ecosystem and principally the benthos could respond to water quality improvement as restoration of Chesapeake Bay progresses. The proposed approach is general rather than scenario-specific, and can be applied to future runs of this and other water quality models. The results of these processes, as they might be expressed in the natural system, are based on the best available estimates made by qualified scientists in the areas of their expertise.

ACKNOWLEDGEMENTS:

The estimation of living resource effects which have resulted or might result from changing water quality in Chesapeake Bay is not a unique idea. In particular, relating

hypoxic volume and the exposure of benthic habitat to low oxygen was explored by Taft and others (USEPA 1983a). In this paper, we emphasize that with respect to the model and habitat change estimates we are working with the excellent data of other researchers. Our contribution is to bring this work together to create a better understanding of potential environmental response. Any errors attendant to this process are ours alone and should not reflect on the primary investigators who have shared their ideas with us.

In particular, we wish to acknowledge detailed discussions with the following scientists and researchers: Fred Holland and Mary Tyler, Versar/ESM; Denise Breitburg and Kevin Sellner, Academy of Natural Sciences of Philadelphia; Jonathan Garber, Roger Newell and Larry Sanford, University of Maryland; Jim Fitzpatrick, HydroQual, Inc.; Steve Jordan, Maryland Department of Natural Resources; Charles App, Lewis Linker, Alan Beck and David Hanson, U.S. Environmental Protection Agency. Charles Spooner, EPA/CBP Director and Technical Coordinator Edward Stigall, generously supported us in accomplishing this work. Nina Fisher, Computer Sciences Corporation., produced the graphics and layout.

THE PROBLEM AND OUR STRATEGY

Two principal nutrients, nitrogen and phosphorus, have been strongly associated with the decline of environmental quality in Chesapeake Bay; specifically with eutrophication and the resulting expansion of deep water hypoxia (USEPA , 1983b). Despite efforts to reduce nutrient loads to the estuary, there is a continuing need for controls on both point and non-point nutrient sources, a need recognized in the 1987 Chesapeake Bay Agreement which commits to a Baywide 40% reduction of both nitrogen and phosphorus by the year 2000. The escalating cost for such controls, especially at a time when federal deficits demand fiscal restraint, make decisions on the allocation of limited resources extremely painful. For analysis, we have chosen to use one of the model calibration years --1984. This was a relatively high freshwater flow year which resulted in strong vertical water column stratification and severe deep water oxygen depletion (Seliger et al. 1985). Over the years, the distribution and duration of severe hypoxia has defined the survival and habitat range for many ecologically important species in the Bay (N. Mountford et al. 1977).

We will evaluate potential effects on habitat conditions which could result from implementing either of two control scenarios of primary interest; a 40% reduction in total phosphorus loadings basin-wide (TP) and a 40% reduction in both total nitrogen and total phosphorus (TN+TP). Phosphorus and nitrogen removal alternatives are, of course, separated by a substantial incremental cost to basin taxpayers. At the same time, the improvements suggested by the model output are perceived by many agency decisionmakers to be relatively modest. For this reason, in evaluating the scenarios we will portray potential environmental results in terms of the incremental benefit from upgrading treatment from TP to TN+TP.

Whenever we seek to "force" the system a small distance further along the path of restoration, the apparent resistance is relatively great and the apparent benefits not so dramatic as the previous increment. We suggest that biological mechanisms inherent in the Bay may loop back and enhance the relatively modest changes implied by the model results, producing improvements of real environmental significance.

We will deal with two principles, both of which will be expanded in turn:

1. The model indicates oxygen distribution with depth in the water column would improve as the proposed pollution controls are implemented, resulting in increased habitat suitable for living resources.

2. Such improvements in habitat with depth could result in pycnocline tilting events impacting less benthic habitat than occurs under present conditions in the Bay.

HABITAT EXPANSION FROM IMPROVEMENTS IN OXYGEN DISTRIBUTION AT DEPTH

The model estimates dissolved oxygen (DO) for each segment and at different depths in the water column. For current conditions, these estimates represent quite well the observed summer oxygen at different depths in Chesapeake Bay. For various pollution control scenarios these projections change, generally producing higher oxygen values which penetrate deeper in the water column. This pattern is shown schematically (Figure 1) for two hypothetical conditions. When acceptable DO conditions extend to an increased depth according to model calculations, a strip or perimeter of bottom previously exposed to unfavorable conditions could become suitable habitat.

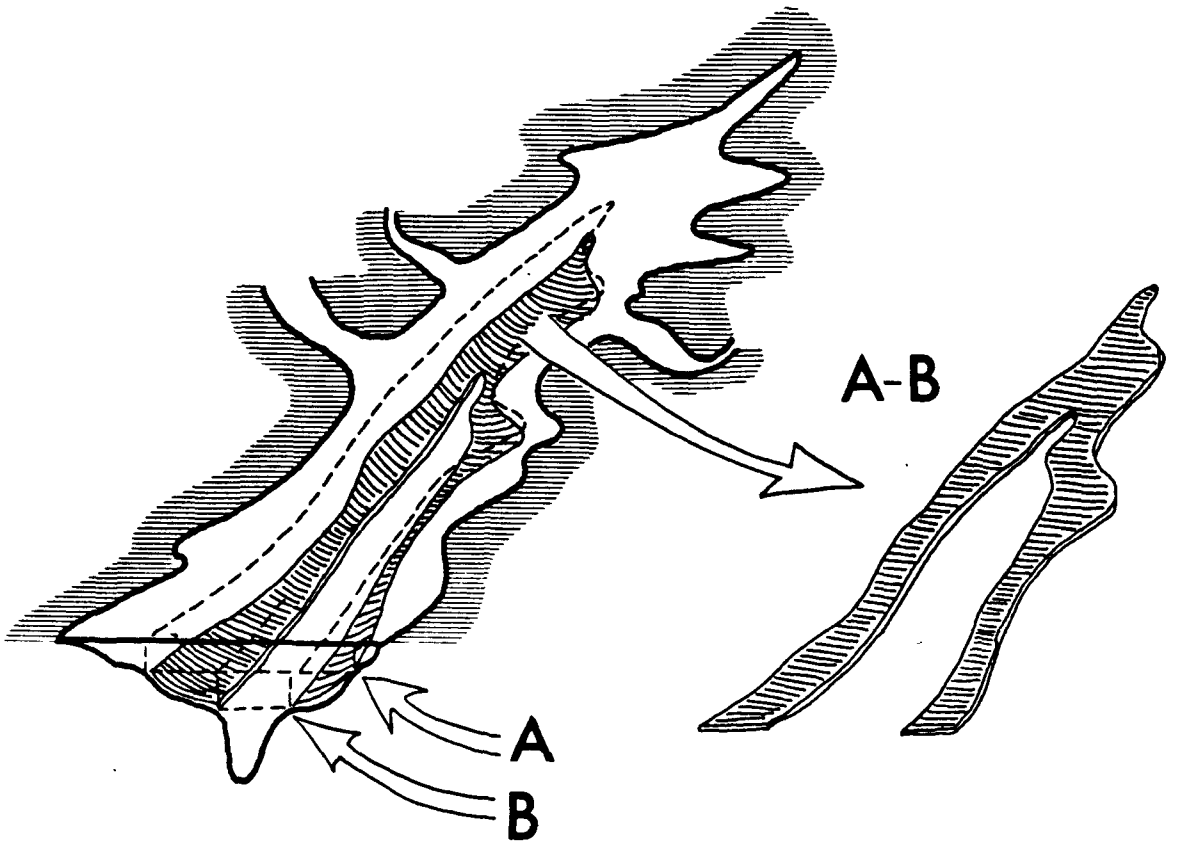
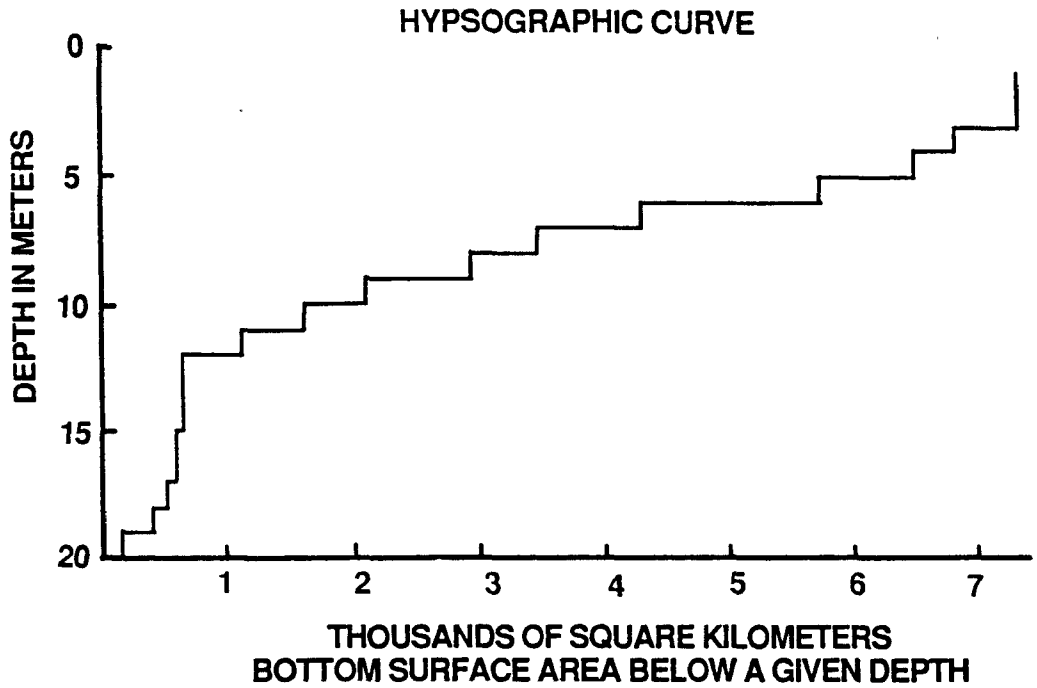


Figure 1: Schematic representation of incremental habitat improvement. "A" scenario: unfavorable conditions extend high in the water column. "B" scenario: unfavorable conditions are reduced and are found only at greater depth. "A-B" differential: represents an increment of Bay bottom which is now potential new habitat.

Figure 2 is a hypsographic curve summarizing the bottom area of several hundred modeled segments in Chesapeake Bay. Starting with segments having a modeled depth of 20 m, the total additional bottom area encountered by rising up each meter in the water column can be read along the abcissa. At right is the entire modeled area of the estuary, 7500 km^2. The "shape" of this curve is typical for water bodies with a relatively constrained deep region and rapid rise to a broad "shelf" region of substantially shallower depth.

The modeled structure of Chesapeake Bay superimposes "boxes" of varying dimension on the Bay's natural contours. This scheme does not capture accurately the extremely deep holes and narrow trench areas which exceed 20 m and even 30 m in a number of areas. The estuary is, however, modeled (HydroQual, Inc. 1987) to contain the same volume as



Chesapeake Bay. Cronin (1971) gives the accumulated area of the Bay as 6495 km^2 but when, as in the model, Tangier Sound (771 km^2) and Pocomoke Sound (379 km^2) are added total area is 7645 km^2, within 1.9% of the model value (7500 km^2). The model, therefore, fairly represents the overall distribution of potential benthic habitat area at successive depths in the real estuary.

For this paper, output was structured to present bottom area at each of 73 unique depths which occur in the model structure, and summer average oxygen exposure was calculated for each spatial habitat element. The matrix for 1984 and the two management scenarios under consideration has about 2000 entries. We have therefore used Occam's razor and summarized the essential elements as Table 1. This table presents summer average exposure of benthic habitat to varying degrees of hypoxia and DO levels equal to the state water quality standards for both the calibration year and scenarios of 40% load reduction in TP and TP+TN. The expected instantaneous minimum DO can be estimated (HydroQual, Inc. 1987) from any summer average value using the relationship:

$$DO_{(min)} = 0.887\ DO_{(obs\text{-}mean)} - 0.937$$

Since this regression accounts for 92% of observed variance in the source data from Chesapeake Bay monitoring (cf. Mountford and Mackiernan 1986), we believe summer averages (represented by the specified ranges) are a reasonable depiction of habitat conditions for the living resources of concern.

Model output can also be interpreted as water volumes with summer average DO equal to the specified range. These volumes can be compared among scenarios to estimate incremental improvements in habitat available to pelagic organisms (e.g. rockfish and bay anchovy which swim free in the water column). Such an analysis is not included in the present paper but may be of substantial environmental significance (Coutant, personal communication).

Vertical distribution of oxygen exposure

Data from Table 1 have been grouped in accordance with the ranges proposed above and are graphically displayed as a case comparison in Figures 3 A, B, and C (see page). These

histograms represent aggregation into 1 m depth increments and are shaded to depict the severity of hypoxia. The denser the shading, the greater the environmental stress at a given depth and the lighter the shading, the greater the improvement. The abcissa reads in square kilometers of bottom surface area exposed to the indicated condition.

The 1984 calibration year used for this comparison clearly has the least favorable condition. Simple inspection shows the reduction in extreme hypoxia forecasted to result from imposition of the 40% TP reduction scenario. In the 40% TP+TN scenario for the summer average condition elimination of extreme hypoxia is projected. Other categories of reduced oxygen are proportionately improved.

Table 1
Chesapeake Bay bottom (km^2) exposed to varying degrees of hypoxia for 3 scenarios run on the summer-averaged Steady-State model

	Range of Summer Average Dissolved Oxygen Exposure (mg/l) *							
Management Scenario	0-0.5	0.5-1.0	1.0-1.5	1.5-2.0	2.0-3.0	3.0-4.0	4.0-5.0	>5.0
1984 Calib.	367.36	133.37	39.83	148.65	117.98	586.58	997.98	5107.73
40% TP	48.80	299.17	141.01	86.74	188.48	481.24	1105.69	5223.33
40% TP+TN	0.00	136.37	230.99	90.81	231.04	368.27	972.53	5469.47

* The DO ranges displayed are chosen to represent approximate "break-point" levels significant to the biological community. These are arbitrary but convenient. Each is supported by a brief justification:

0.0-0.5 mg/l	Inhospitable to living organisms other than sulfur bacteria
0.5-1.0 mg/l	Some benthic organisms can tolerate limited (a few days) exposure
1.0-1.5 mg/l	Region in which hypoxic benthic phosphorus flux may decrease by 85%
1.5-2.0 mg/l	Probable acceptable exposure for demersal (sinking) fish eggs
2.0-3.0 mg/l	Stressful, especially if prolonged (> 7 days), but probably non-lethal
3.0-4.0 mg/l	Below the instantaneous water quality standard
4.0-5.0 mg/l	Meeting the instantaneous water quality standard on average
> 5.0 mg/l	Exceeding the water quality standard on average

Is the expectation of summer bottom DO above 1.0 mg/l a plausible one? So far as we know, the earliest survey work for the Chesapeake and lower Potomac estuaries was done in 1912 (Sale and Skinner 1917). Their worst case condition, in the lower Potomac towards Point Lookout (a region today also characterized by summer anoxia) was described for 21 and 22 September, 1912, with the mean for six stations at 3.40 mg/l and a range of 1.57-4.57 (converted from $ml \cdot l^{-1}$ to $mg \cdot l^{-1}$ with the ratio 1.4286 after Barnes, 1959). Sale and Skinner "...expected the maximum reduction in the DO content of the denser bottom layer would occur at this season of the year, since the decomposable organic matter from plankton form, vegetation, etc., is greater during the summer months, and bacteria which assist in the breaking down of organic matter are most active at this season, because of the higher temperature." In this they anticipated aptly our current understanding of the process, if not the severity experienced today.

Lowest bottom DO in the mainstem Chesapeake (55% saturation) was observed 10 miles off the mouth of the Rappahannock on 20 September, 1912. This translates to about 4.3 mg/l (Fair and Geyer 1963). Off Annapolis they found about 5.2 mg/l, albeit on 3 October. These values were both substantially lower than the surface values. It is interesting the authors here also attributed the reduction in bottom DO to: "Broken-down submerged plants, leaves,

elutriated soil and other debris which are carried into the estuary and bay by numerous tributaries...."

LIVING RESOURCE IMPLICATIONS

Except for the overharvesting of oysters, the living resources of this turn-of-the-century Chesapeake were in good shape. If Sale and Skinner's data reveal the true condition of bottom oxygen in 1912, it is heartening to consider what re-establishing these concentrations could mean to a future Chesapeake.

In order to investigate the living resource implications of the model projections for benthic communities in the Bay, it is necessary to interpret the water column model results in terms of bottom surface area impacted. In this paper, we concentrate on those portions of the Bay bottom currently exposed to varying degrees of hypoxia (< 3 mg/l DO--summer average). The discussion of improvements focuses on these areas.

Holland and Shaughnessy (personal communication) have interpreted data from thousands of benthic samples collected in Chesapeake Bay and related those data to habitat conditions observed at their sampling stations. They provided the provisional data used to construct Table 2. These data are based on samples for 1985 but Holland indicated that 1984 was a substantially more productive year for the benthos and recommended that the original values be increased by a factor of 2.0 for use in a 1984 scenario.

Table 2.
Provisional Numbers Characterizing Benthic Habitat Ecology in Chesapeake Bay†

Habitat Condition	Benthic Organism Biomass (g/m^2)	Annual Benthic Production (g/m^2)	* Potential Capacity for Particulate Removal 6-20 mg/g dry wt/day
Severe Hypoxia or Prolonged Anoxia	0.3-1.0	1.0-2.0 (mean 1.67)	11.25-37.52 mg/g/d (mean 8.9 g/yr)*
Less severe exposure <30 days (cf: today's 30-40' contour)	1.7-2.4	8.56	57.01-190.03 mg/g/d (mean 45.1 g/yr)
Consistent 1-2 mg/l bottom DO	4.7-11.2	24.9	165.83-552.78 mg/g/d (mean 131.2 g/yr)

* "Potential" does not mean that the entire water column would be "cleared" of particles. Holland estimates that shallower water columns (4-5 m) might be 60% cleared, while at 30-40 m depth only 10% of the water column might be cleared.
† Source: Holland and Shaughnessy, personal communication, 1987.

Raising summer average bottom DO in substantial areas of the deep estuary (or in the case of TP +TN, the entire estuary) to a level above 0.5 mg/l and frequently above 1.5 mg/l would permit the colonization of these areas by benthic infauna (worms, clams, etc.). This colonization presently begins each year when a larval "set" occurs following the seasonal water column mixing which re-establishes oxygen in bottom waters. Organisms from this set are detected by sampling in less than a month. They are extirpated, however, late each spring when hypoxia first occurs.

Improvement in "long-lived" benthos

The areas subject to varying degrees of hypoxia shown in Figure 3 have been aggregated into three categories to correspond roughly to the habitat groupings Holland and Shaughnessy considered in their analysis of actual benthic community data. These are shown in Table 3.

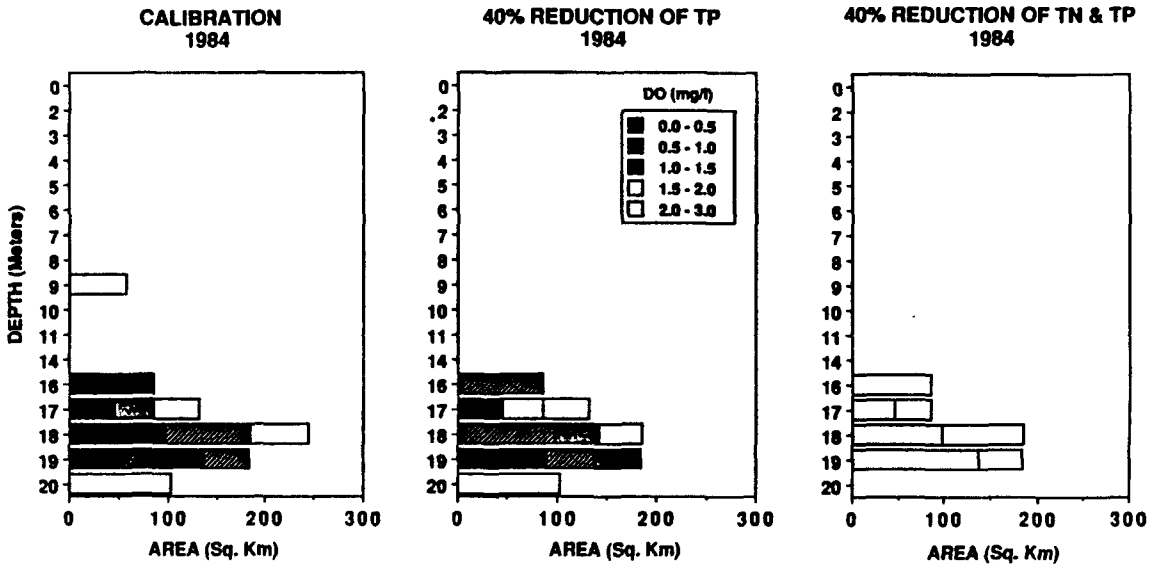


Table 3.
Aggregation of modeled benthic hypoxia exposure into habitat categories*

Model Scenario	Severe Hypoxia or Prolonged Anoxia (0.0-1.0 mg/l)	Less Severe Exposure (cf: 30-40' contour) (1.0-2.0 mg/l)	Consistent Bottom DO > mg/l (2.0-4.0 mg/l)
1984 Calib.	501.03	188.48	704.56
40% TP Removal	272.97	227.75	669.72
40% Removal TN + TP	136.37	351.80	599.31

* Areas are based on data in Table 1 and are expressed as square kilometers bottom area.

The coefficients of Holland and Shaughnessy (Table 2) are combined with the aggregated areas (Table 3) to produce Table 4, an overall benthic community interpretation for the hypoxic regions in Chesapeake Bay and potential habitat improvements projected by the model for this sub-region of the entire Bay.

Areas with DO remaining above 1 mg/l could begin to sustain "long-lived benthos," persisting for a year or more. As organisms grow in size their filtering capacity increases markedly. This capability assists in the aggregation of flocculents into larger effective particle sizes ("pelletizing"), reducing resuspension through the production of mucilaginous pseudofeces. The filtering potential of this expanded benthos under the 40% TP scenario could remove 7.4 x 10^4 metric tons of particulate material;, 24% more than benthos under the 40% TP scenario in the 1984 calibration year. For the 40% TP+TN scenario, this potential could rise to 9.4 x 10^4 metric tons, 40% more than the calibration year. While Holland's caveat in Table 2 is fully appropriate, there is good precedent to cite the importance of benthos as a contributor to particulate removal.

Newell (personal communication) has calculated that the overharvesting of oysters in the Maryland Chesapeake has dramatically reduced particulate clearing capacity. He suggests that in the 1880s filtering turnover time for the Bay could have been 3.8 days; in the mid-1970s this increased to 97.2 days and with the 1987 oyster population estimated by the Maryland Department of Natural Resources to 486.1 days.

Table 4
Benthic Community Interpretation Summary for Hypoxic Regions of Chesapeake Bay in the 1984 Model Scenario and two Projected Environmental Management Strategies

Management Strategy	Summer Ave. Bottom DO	Total km^2 Bottom Area	Benthic Organism Biomass		Annual Benthic Production		Potential Removal Capacity for Particulates		Benthic Organisms Consumed by Food Chain in Region
			g/m^2	Metric Tons	g/m^2/yr	Metric Tons	g/m^2	Metric Tons	M. tons/yr. k=0.33*
1984 Calibration Year	0.0 - 1.0	501.03	1.3	651.3	3.3	1653.4	17.8	8923.1	——
	1.0 - 2.0	188.48	4.2	791.6	17.1	3223.0	90.2	17,000.9	——
	2.0 - 3.0	117.98	16.0	1887.7	49.8	5875.4	262.	30,910.8	——
	——	——	—	3330.6	—	10,751.8	—	56,834.8	3583.6
40% TP Removal Scenario	0.0 - 1.0	272.97	1.3	354.9	3.3	900.8	17.8	4858.9	——
	1.0 - 2.0	227.75	4.2	956.6	17.1	4664.0	90.2	20,543.1	——
	2.0 - 3.0	188.48	16.0	3015.7	49.8	9386.3	262.	49,381.8	——
	——	——	—	4327.2 (23.3%)	—	14,951.1 (28.1%)	—	74,783.8 (24.0%)	4983.2 (28.1%)
40% TP+TN Removal Scenario	0.0 - 1.0	136.37	1.3	177.3	3.3	450.0	17.8	2427.4	——
	1.0 - 2.0	351.80	4.2	1477.6	17.1	6015.8	90.2	31,732.4	——
	2.0 - 3.0	231.04	16.0	3996.6	49.8	11,505.8	262.	60,532.5	——
	——	——	—	5351.5 (37.8%)	—	17,971.6 (40.2%)	—	94,692.3 (40.0%)	5989.9 (40.2%)

* K = 0.33 represents rounding to the mid-point of the "grazing range" 30-35%, see text.

Forecast biomass improvements

Relative to the calibration year, habitat improvements are projected by the model which would permit extension of increased benthic populations into regions of the Bay covering some hundreds of square kilometers. The augmented biomass of such a benthos for the 40% TP scenario is estimated at 4.3 x 10^3 metric tons, an increase of 23.3% over the calibration. For the 40% TP+TN scenario, biomass is projected at 5.4 x 10^3 metric tons, 37.8% above the 1984 calibration year.

Production of the benthic community

Calculated production for benthic communities accounts for losses by mortality and the grazing of predators. This estimate includes the new living tissue accumulating and the amount passed up the food chain to higher organisms, presumably including harvestable fish and crabs. In Table 4, we have developed production estimates. For the 40% TP case, production could rise 28% over the deep region value for 1984. For the 40% TP+TN case, production could rise to 1.8 x 10^4 tons, 40.2% greater than the calibration year.

Potential pass through to the food chain

Holland estimates that from 30-35% of benthic biomass is grazed by predator organisms in the estuarine system, so that (assuming 33%) the 40% TP scenario could result in 5.0 x 10^3 metric tons moving up the food chain in Chesapeake Bay. This quantity is 28.1% more than this region might have produced in the calibration year. For the 40% TP+TN scenario, potential increment to the food chain is 6.0 x 10^3 tons, an increase over the 1984 calibration of 40.2%.

Improvement in shellfish habitat

Mature and healthy oysters, especially in low temperature conditions, can tolerate more than a week with closed shells by respiring anaerobically (Beck and Hanson, personal communication, 1987). Chesapeake Bay oysters, however, encounter low oxygen at high temperatures (near 25° C) and usually just after or during their spawning period. At a time when their metabolic rate and nutritional need is greatest and their energy reserves are least,

exposure to hypoxic conditions is very stressful (Newell 1987). Over the past decades, such repeated exposures have systematically eliminated the deeper oyster beds. The absence of these beds correlates well with intrusions of severe hypoxia measured today. It also correlates with the distribution of higher salinity and the occurrence of the major oyster diseases MSX and "Dermo." But, it would surprise no one to find all these prejudicial factors are synergistic. The fact remains that deeper oyster beds, once viable, are no more.

Newell (personal communication, 1987) and others believe that deeper shellfish habitats, once present in the pristine Chesapeake, cannot become re-established until they are insulated from repeated (even though episodic) insults from low oxygen intrusion. This is particularly important for juvenile oysters since swimming veliger oyster larvae appear to die upon exposure to zero oxygen after about 18 hours. Severely hypoxic regions might also be a "black hole" for oyster larvae spending any time there. At 20% oxygen saturation, swimming larvae exhibit an avoidance response, swimming upwards in the water column where they may be swept away from suitable attaching substrate and lost. (Mann et al. 1987 and Newell, personal communication).

Using the database which supports Table 1, we can estimate the aggregate bottom areas impacted by DO levels $\leq$ 4 mg/l . If we consider this as the unacceptable long-term average condition for successful shellfish beds, the implication is that this marginal habitat (1394 km^2 during the 1984 calibration year) decreases to only 1170 km^2 with 40% TP removal, an improvement of 223 km^2 (55,186 acres). For the 40% TP+TN scenario, the region below 4 mg/l summer average decreases to 1057 km^2, an increase in the region with potential for oyster and clam production of 336 km^2 (83,099 acres). These incremental acreages are depicted as a histogram in Figure 4 in which the "worst case" (1984) is zero and is not shown.

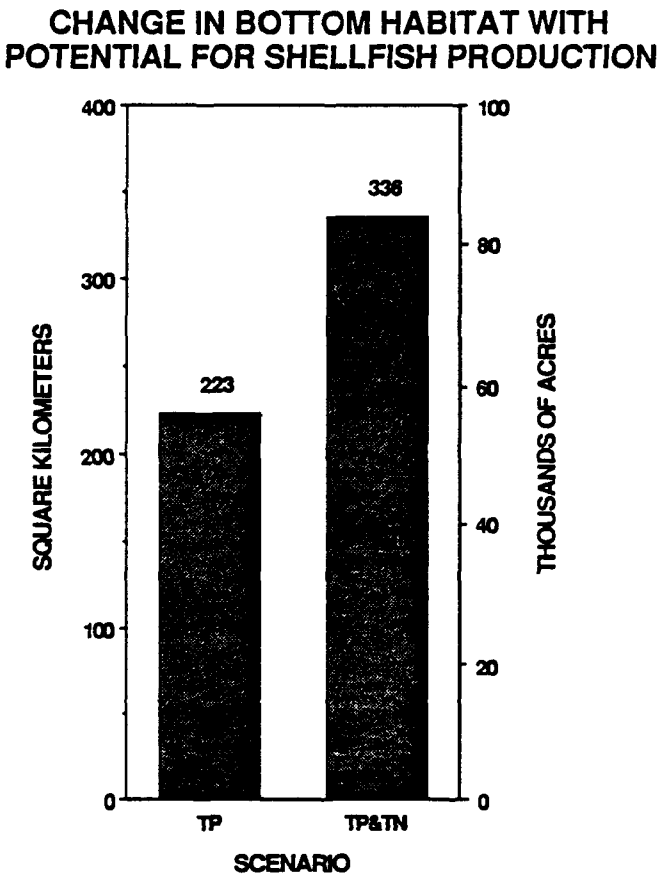


Figure 4:
Increases in bottom area with potential for shellfish production projected for two management scenarios; 40% reduction in total phosphorus loading (TP) and 40% reduction in both total nitrogen and total phosphorus (TP+TN).

Reduction in benthic phosphorus release

Where DO could be raised to 1.0-1.5 mg/l, this would substantially interrupt the chemical reactions that release phosphorus from the sediments. The release of phosphorus is never completely interdicted. As calculated in this model, however, rates drop from about 8.0 $mg \cdot m^{-2} \cdot d^{-1}$ in the presence of severe summer hypoxia to about 1.2 $mg \cdot m^{-2} \cdot d$ in shallower waters not experiencing these conditions (HydroQual, Inc. 1987). Projected changes following implementation of the two management scenarios are detailed in Table 5 and represent substantial reductions in phosphorus release to the water column.

Table 5
Changes projected in benthic phosphorus release from bottom areas in Chesapeake Bay with summer average DO below 1.0-1.5 mg/l converted to a higher oxygen regime as a result of two pollution control scenarios

Scenario	km^2	km^2 Area Change	% Change kg/day *	Incremental % Change for TP+TN	Phosphorus Release (metric tons)	P Release % Change
1984 Calib.	501.03	-	-	-	4.008×10^3	-
TP 40% Rem.	272.97	228.06	-54.5	-	2.458×10^3	38.7%
TP+TN 40%	136.37	364.66	-72.8	-18.3	1.529×10^3	61.9%

* Releases in kg/d represent the *differential* of hypoxia mediated release at 8 mg over the "baseline" level of 1.2 mg for mid-Bay lateral regions above the pycnocline.

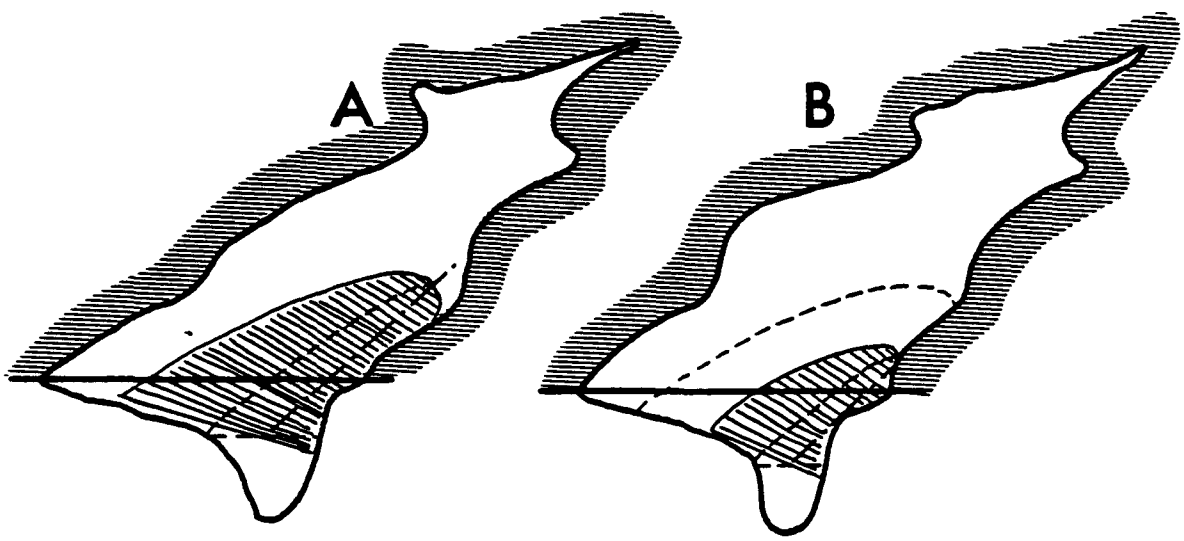


Figure 5: Schematic representation for pycnocline tilting events in Chesapeake Bay; "A" scenario in which unfavorable conditions reach high in the water column and tilting exposes large "shelf" areas to low oxygen; "B" scenario, following an incremental improvement, shows unfavorable conditions are lower in the water column and a tilt of the same magnitude results in less impacted bottom area.

HABITAT IMPROVEMENTS DURING PYCNOCLINE TILTING EPISODES

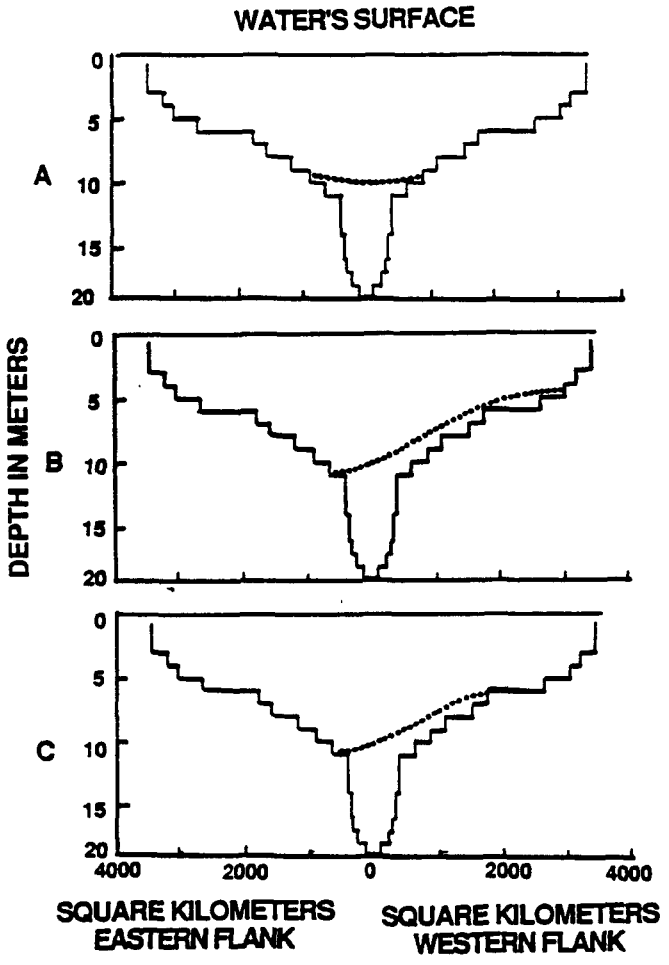
A second major environmental process, one not modeled by our current tools, is the phenomenon of pycnocline tilting. This process is shown schematically in Figure 5, version (A) representing tilt to some fixed angle with unfavorable conditions existing higher in the water column and version (B) with the *same* angle of tilt following an improvement scenario where unfavorable conditions are only encountered at a lower depth and a substantial region of Bay bottom is spared the transient exposure.

The pycnocline, and the horizons of varying oxygen concentration below it, are highly dynamic zones. The term "seiche" has been loosely used to describe the transport of sub-

pycnocline waters up onto the shallower shelf lateral to the main channel. In fact, the term "tilt" is better applied, and such conditions usually arise when a prolonged wind set "piles" water up against one shore and deeper water upwells into the shallows of the opposite, windward shore. In summer, this condition frequently occurs with a southwest wind, which sets surface water east and establishes the return circulation to upwell water with low DO onto the western shore *(towards)* the wind.

Explanation for curve of bottom half-areas

The result is that approximately half the Bay's shallow bottom is affected by reduced oxygen concentrations. In order to depict this, the bottom *area* data in Figure 2 has been re-plotted symmetrically in Figure 6 A, so that half the incremental area at each successive depth is on the eastern and half on the western flank. The abcissa measures in square kilometers, both left *and* right from the center. These are curves of bottom *areas* totalled for all the model segments, and are not a cross section of the Bay.



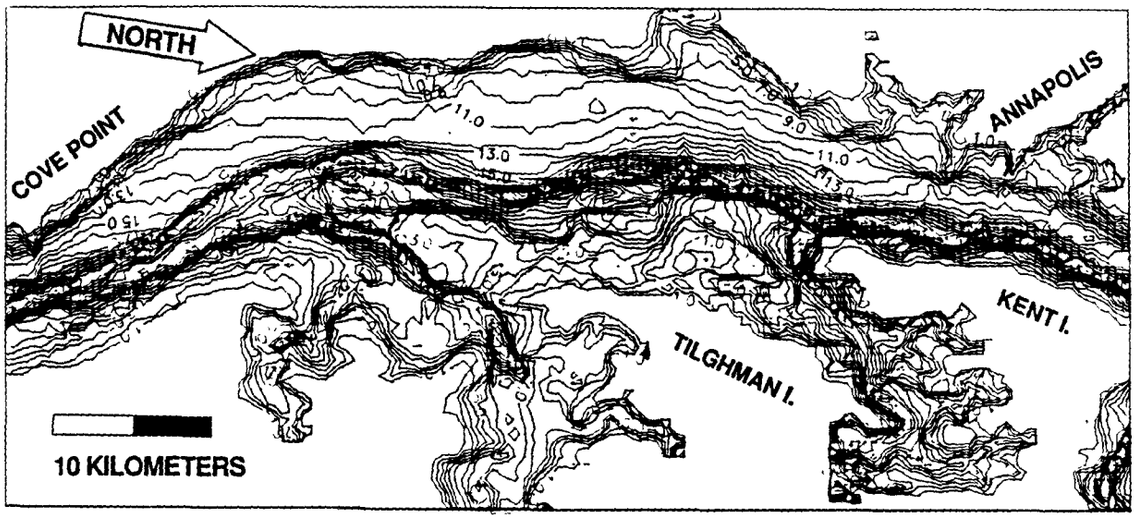
Monitoring data and research funded by the Chesapeake Bay Program, Maryland DNR (Sanford and Sellner 1987) and NOAA Sea Grant indicate that the amplitude of these tilting events can easily advect oxygen-poor waters inshore to depths of 3-4 meters. Documented events have brought these conditions wholly to the shoreline (personal observations, 1982, 1987; and Breitburg, 1987, personal communication). This latter circumstance is a condition usually described as a "crab jubilee" when hundreds or thousands of organisms attempt to leave the water and often many suffocate.

Figure 6:
Curves of half bottom areas for Chesapeake Bay: "A" with pycnocline at rest around 10 m; "B" with pycnocline tilted westerly up to 4 m depth, and "C" with pycnocline tilted westerly up to 6 m depth, exposing different areas to sub-pycnocline conditions.

This condition may hold for as long as the wind field persists, a period which can last a matter of hours or several days (Sanford et al. 1987). When the wind forcing ceases, the Bay rocks laterally back and forth like a tub of water, with a calculable period. This wave is a "seiche" and the period for a full cycle of the pattern, the seiche period. The holding pattern is usually the problem, not the seiche. Long exposure to oxygen deprivation can readily exceed the survival times of key organisms.

Habitat improvement during pycnocline tilting events

The bottom habitat areas potentially affected by such conditions are shown in Figures 6 B, and 6 C. The areas impinged range from 875 km^2 for an intrusion of 6 m to 2180 km^2 for an intrusion of 4 m depth. These are tremendous areas, but inspection of Figure 7 (generated from the database of Goldsmith and Hutton 1977) showing detailed bathymetry for the



1 METER BATHYMETRY CONTOURS FOR MD - CHESAPEAKE BAY

Figure 7: Middle (Maryland) reach of Chesapeake Bay with 1 m contours shown (Data Source: Goldsmith and Hutton 1977).

middle 88 km reach of Chesapeake Bay indicates large areas can be covered by a lateral pycnocline tilt of a few meters.

The frequency of such intrusion events has not been well documented in the past, since they require continuous recording equipment deployed in a hostile environment. During several recent survey periods, including deployments of a month or more (Sanford and Sellner, 1987; Breitburg, personal communication), enough events have been observed that one can estimate in an "average" year that perhaps six events occur; four of intermediate intensity (DO $\leq$ 2 mg/l) and possibly two of severe intensity (DO $\leq$ 0.2 mg/l). Both classes of events can persist sufficiently long for environmental damage. Much more work needs to be done with respect to such continuous data series since the frequency and duration of events determines magnitude of impact.

Currently proposed management strategies are not directed at changing these intrusion frequencies or duration. The improvement in DO conditions to greater depths in the water column, however, would quickly impact their severity. Achieving the TP+TN scenario is projected to result, on the average, in virtually no extremely hypoxic water ($\leq$ 0.5 mg/l) available for intrusion onto the Bay's shelf regions.

CAVEATS TO THESE PROJECTIONS

Models simplify reality in order to deal with the complexity that confronts us when we try to interpret the natural world. This simplification process is inherently hazardous, as workers found when the Potomac Estuary Model was unable to reproduce observed algal concentrations during a major 1983 bloom (Mutman and Masse, 1985). Subsequent research suggested that a previously unrecognized feedback mechanism between the sediments and water column had operated to mediate the bloom (Seitzinger, 1985). Such mechanisms could easily be encountered again in Chesapeake Bay.

Some workers (Magnien, personal communication) expect that reductions in nutrient loadings would not necessarily result in incremental improvements of DO with depth, but rather in a gradual overall rise in average sub-pycnocline oxygen. We have no way of directly evaluating this possibility but note that the model calculates oxygen distribution as we have analyzed it.

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